POLITECNICO DI TORINO

Corso di Laurea in Ingegneria per l'Ambiente e il Territorio

Tesi di Laurea Magistrale

Assessment of the effect of digestion technology on the dewaterability of waste activated sludge

Relatrice:
Prof.ssa Silvia Fiore

Co-relatore:
Prof. Pavel Jenicek, VSCHT University of Prague

Candidata:
Chiara Marchi

Ottobre 2019

Summary

Abstract 3
1. Introduction 4
2. State of the art 6
2.1. Conditioning of the sludge 10
2.2. Dewatering process 13
2.3. Final disposal 20
3. Materials and Methods 26
3.1. Collection of samples 26
3.2. Characterization of the samples 26
3.3. Flocculant choice 26
3.4. Dewaterability 29
3.5 Analysis of dewatering performances 31
3.6. Statistical Analysis 32
4. Result and discussion 34
4.1 Flocculant choice 34
4.2.1. Sludge from the Post Aeration Pilot Plant 34
4.2.2 Sludge from Lab-Scale Anaerobic Digesters 36
4.3 Statistical evaluation 38
4.4 Technical and economical assessment of the scale up 39
5. Conclusions 46
6. Acknowledgment 48
7. References 49

Abstract

Over 11 M tons of waste activated sludge (WAS) are produced every year by wastewater treatment plants in Europe, often with a dry substance content below 20-25 % w/w. It is crucial to find the best solution to treat WAS in order to reduce its amount, and thus the disposal costs, and to explore reuse perspectives that could imply the lowest possible costs and environmental impacts. WAS management is a common topic nowadays and its disposal usually involves landfilling, incineration, composting and other applications related to agricultural use. The main goal of the optimization of WAS management should be to reduce the amounts landfilled and/or destined to agricultural applications and to increase the fraction sent to thermal valorization. There are many techniques that can be aplied to reduce water content in WAS and one of these, which has been improved these last years, is dewatering. This process helps to separate water from the solid particles and it plays a key role within any sludge disposal perspective.

This thesis is aimed at the investigation of the optimization of dewaterability through the analysis of different types of WAS obtained from different anaerobic digestion processes (mesophilic and thermophilic). The considered WAS samples were obtained from Central Prague wastewater treatment plant and from pilot-scale reactors. For each type of WAS different process parameters (applied pressure, dose and type of flocculant) have been investigated in order to assess the performances of the dewatering process through the amount of residual total solids after press-filtering at lab scale. The most challenging phase of the study was related to the optimization of the coagulation phase, particularly because the efficiency of the flocculant is highly unstable because of several factors that can change during the process and affect flocs structure and amount of water that could be extracted. The results of the experimental activity showed that different WAS samples exhibited different behaviors towards the overall dewaterability process. An economic preliminary assessment of the scale-up of the dewaterability process concluded the study.

The experimental activity of this thesis was performed at VSCHT University in Prague along 5 months in the period February – June 2019. I gratefully acknowledge ERASMUS+ program for the economic support of a 10 months stay at VSCHT University in Prague in the period September 2018-June 2019.

1. Introduction

These days 11 M tons of waste activated sludge (WAS) are produced every year by wastewater treatment plants in Europe (Wójcik & Stachowicz, 2019), often with a dry substance content below 20-25 % w/w. It is crucial to find the best solution to treat WAS in order to reduce its amount, and thus the disposal costs, and to explore reuse perspectives that could imply the lowest possible costs and environmental impacts. There are many techniques that can be applied to reduce water content in WAS and one of these, which is has been improved these last years, is dewatering. This process helps to separate water from the solid particles in order to reduce the disposal costs of the sludge; it can also help to promote its agricultural reuse.

WAS management is a common topic nowadays and disposal usually involves landfilling, incineration, composting and other applications related to agricultural reuse (Figure 1). The data about WAS management in Europe (Eurostat, 2018) (Figure 1) are not available for all countries and those which are available are referred only up to 2017. The main goal of the optimization of WAS management should be to reduce the amounts landfilled and/or destined to agricultural applications and to increase the fraction sent to thermal valorization (Panepinto et al., 2016). In this framework, the optimization of dewatering processes, aimed to reduce WAS volume and mass and therefore also transport and energetic costs through the reduction of moisture content and the separation of solid and liquid phases (Wei et al., 2018), is highly necessary.

This thesis is aimed at the investigation of the optimization of dewaterability, which plays a key role within any sludge disposal perspective, through the analysis of two different types of WAS obtained from different processes of anaerobic digestion (mesophilic and thermophilic). For each type of WAS different process parameters (applied pressure, dose and type of flocculant) have been investigated in order to assess the performances of the dewatering process through the amount of residual total solids. The experimental activity of this thesis was performed in VSCHT University in Prague, within the research group coordinated by prof. Pavel Jenicek, along 5 months in the period February – June 2019. I gratefully acknowledge ERASMUS+ program for the economic support of a 10 months stay at VSCHT University in Prague in the period September 2018 - June 2019.

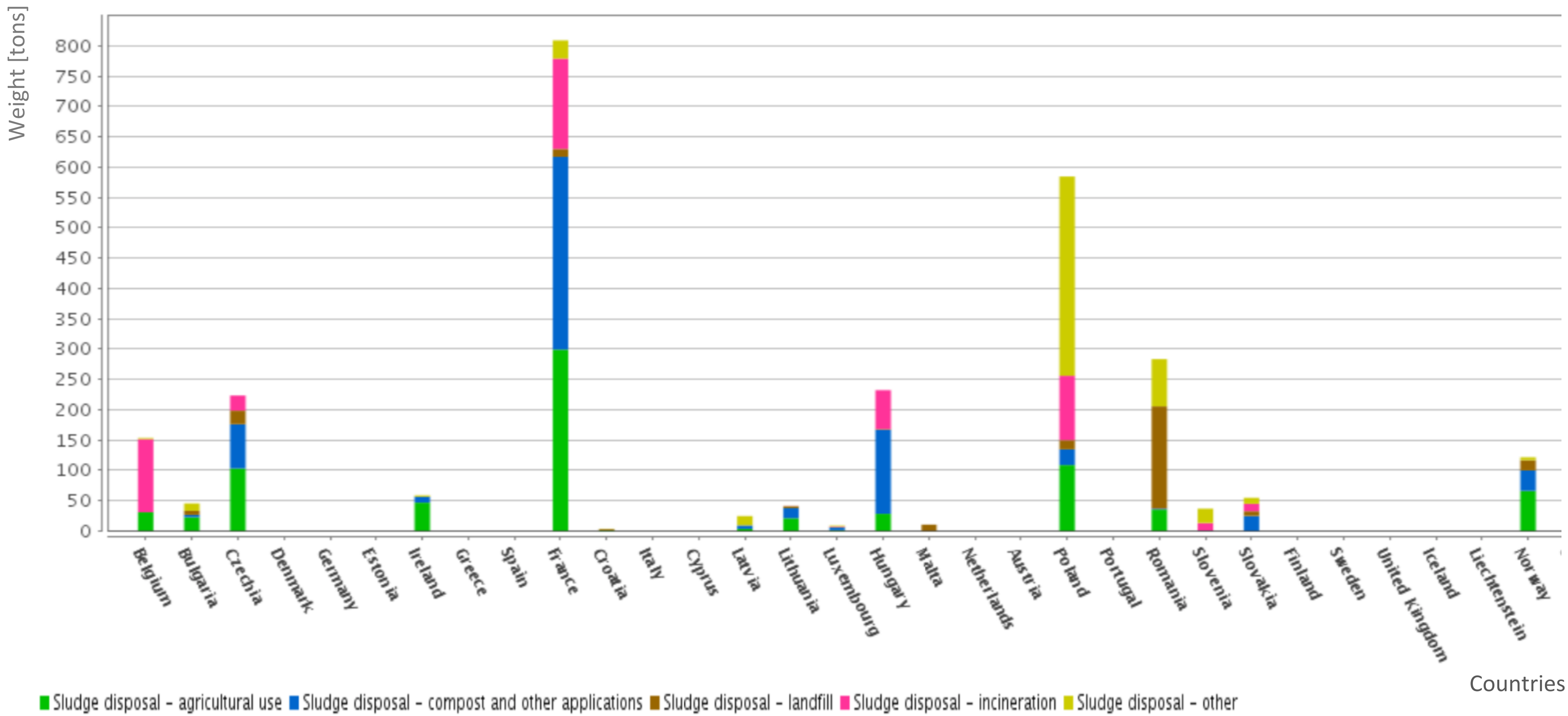

*Figure 1.1_* Disposal options for waste activated sludge in different countries in Europe (Eurostat, 2018)

2. State of the art

The principal product that comes out from wastewater biological treatment processes is waste activated sludge (WAS), that is a semi-solid waste flow that has to be treated before disposal (Fang et al., 2019). WAS is also considered as a critical biologically-active mixture of water, organic matter, dead and alive microorganisms, and inorganic and organic toxic contaminants that contains 97-98 % of water; moreover in order to be used as a biosolid, it has to be treated properly (Kacprzak et al., 2017).
Recent studies evidenced that the annual production of sewage sludge in Europe is around 11 million tons (Wójcik & Stachowicz, 2019). Urban Waste Water Treatment Directive 91/271/EC considerably improved wastewater treatment in EU in the last decades, having as a direct consequence an overall increase of the produced amount of WAS; afterwards Directive 1999/31/EC banned direct landfill disposal of WAS. WAS production of a specific area is strictly related to several main factors: 1. the presence of a sewer system and a wastewater treatment facility; 2. the population density and flow rate of water distributed for human consumption and other uses (e.g. commercial and industrial users connected to the sewer system), which afterwards becomes wastewater; 3. the requirements of EU and local regulations, becoming more and more strict through the years about the mandatory removal of nutrients from wastewater. The highest WAS production is related to Germany, whereas Cyprus and Malta are the countries with the lowest production per year (Wójcik & Stachowicz, 2019).
In a wastewater treatment plant (WWTP) there are different steps used to treat the sewage sludge: preliminary treatment, primary thickening, liquid sludge stabilization, secondary thickening, conditioning, dewatering, final treatment, storage, transportation and final destination (Kacprzak et al., 2017) (Figure 2.1). The two steps on which it is important to focus more attention are conditioning and dewatering, because they enhance the separation of solid and liquid phases with the consequent decrease of the water content of the digestate, and therefore of its transport and disposal costs (Guilayn et al., 2019). Comparing conditioning (i.e. the addition of a chemical to improve flocs aggregation) and dewatering, it was evaluated that the most expensive is the second; for example in the US the annual cost of sludge dewatering was estimated around 5 billion dollars (Wójcik & Stachowicz, 2019)
The study of disposal and recycling of WAS has become very important in the last decades due to the negative impacts of pollutants and greenhouse gas emissions related to sludge management. These emissions are considered a potential risk during the treatments and the disposals of WAS (Fang et al., 2019). Sludge stabilization is one of the critical steps in sludge management because it affects both its amount and the quality; so, it is very important to

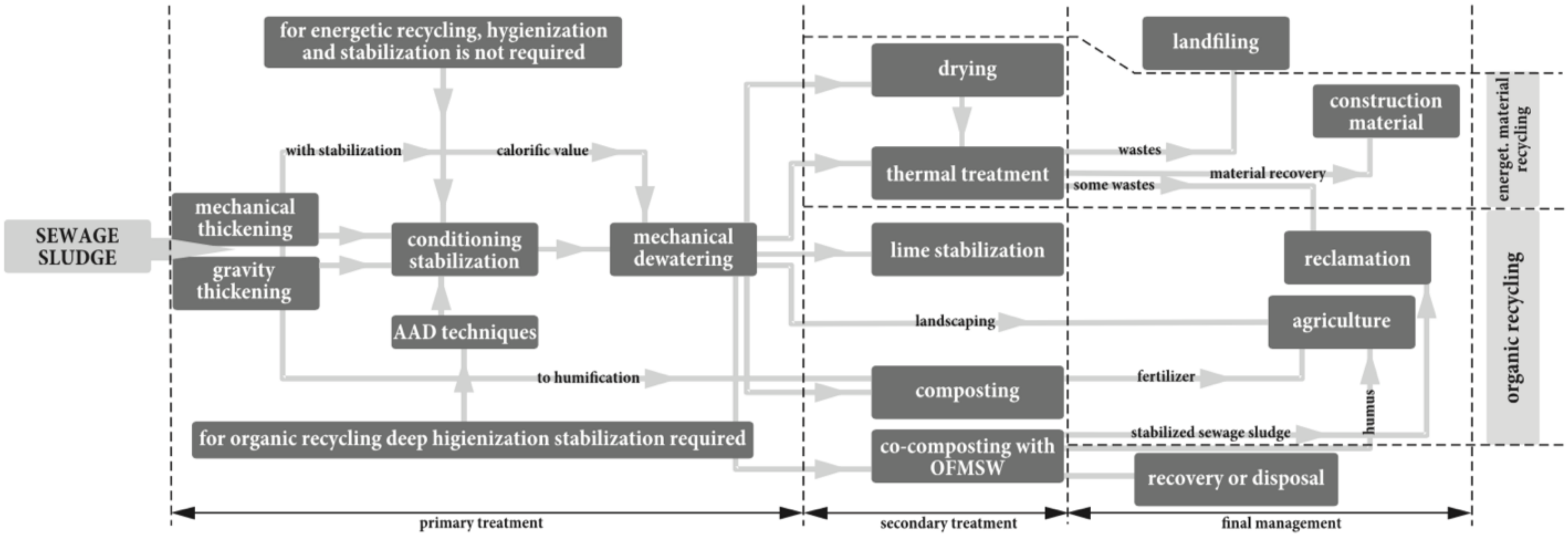

Figure 2.1_ Sewage sludge treatment outline (source: Kacprak et al.,2017)

perform an efficient stabilization in order to reduce the solid fraction and reduce the toxic pollutants, pathogens and viruses (Tomei & Carozza, 2015). A careful management of WAS is also required from wastewater treatment operations during the treatment processes and after the removal from the facilities, which should be consistent with the main concept of European Commission “reduce, reuse, recycle” (Kacprzak et al., 2017).

Through anaerobic digestion and biogas production the anthropogenic organic residues can be converted into energy and organic fertilizers; this way there will be less waste and these two processes are considered the fulcrum for the circular economy implemented into wastewater treatment (Guilayn et al., 2019). Nowadays the study of the efficiency of mechanical dewatering of the digestate, which is the final product obtained from anaerobic digestion, is increasing in order to evaluate which one could be the best solution in order to reduce costs and pollutant emissions (Guilayn et al., 2019). Furthermore anaerobic digestion is also considered one of the most prevalent treatment approaches for the reduction of mass and volume of WAS and also for the mitigation of greenhouse gas (GHG) emissions (Fang et al., 2019).

Among different treatment solutions, anaerobic digestion is considered a good technology to solve the problem of sludge management and to reduce plant costs due to the production of methane (Braguglia et al., 2015). Anaerobic digestion of organic matter occurs when different bacterial trophic groups act cooperating sequentially in order to obtain degradation of different substrates (Braguglia et al., 2014). Anaerobic digestion can be mesophilic, using 35 °C as operating temperature, or thermophilic (55-60 °C). Between these two types of processes, mesophilic ones are more common because they require less energy and they lead to a higher stability of the sludge, but thermophilic processes are the best option to reduce organic matter and to produce methane (Braguglia et al., 2014). Mesophilic processes are less sensitive to toxic compounds but they are characterized by lower organic loading rates (ORLs) and poor inactivation of pathogens; whereas thermophilic processes present faster reaction rates, which allow higher decomposition of volatile solids (VS), production of biogas and inactivation of pathogens, require more energy for heating, have higher odor potential, poorer stability and dewaterability of the digestate (Kevbrina et al., 2011).

Anaerobic digestion, although more complicated to manage and control than aerobic WAS treatment, is the best process to optimize the energy balance of a WWTP due to the production of biogas. Nevertheless, aerobic WAS treatment, even though it would need additional energy for aeration, seems to be more efficient in removing the toxic pollutants in the sludge (Tomei and Carozza, 2015). For large WWTPs the anaerobic stabilization should be considered the optimal process and the quality of the sludge should be improved by adding a post-aeration process (Vojtiskova et al., in preparation). It was demonstrated that a possible solution to enhance the sludge stabilization, leading to the improvement of dewaterability, is the combination of anaerobic-aerobic sludge digestion processes (Tomei et al., 2016). It was discovered that this combination increased the stabilization of secondary (WAS) and mixed sludges and it makes easier to adjust the parameters during the process. This aspect is very relevant because every digested sludge requires different process conditions according to the different reuse and disposal perspectives (Tomei and Carozza, 2015). Indeed the dual-stage anaerobic/aerobic digestion it is a promising solution because of the different reaction systems (Braguglia et al., 2015). It was also evaluated (Parravicini et al., 2008) that post-aeration costs increase the total annual ones of only 0.84 % and that the organic solids content in the digestate could decrease by 20 % after post-aeration. The addition of an aerobic post-treatment can increase WAS stabilization and dewatering, but it doesn't improve biogas production (Braguglia et al., 2014). Another positive effect of post-aeration is total ammonia nitrogen (TAN) removal in the supernatant stream, which can decrease the nitrogen load entering the WWTP through the supernatant back-flow (Tomei and Carozza, 2015). Post-aeration also positively affects dissolved chemical oxygen demand (COD) removal and digestate dewaterability (Vojtiskova et al., in preparation?).

Another possible technical solution that is appropriate for large-size WWTPs could be the separation of primary and secondary sludge (e.g. WAS) due to their different characteristics about biodegradability and dewatering. WAS is less polluted than primary sludge and it can be reused, after stabilization and dewatering, in agriculture; primary sludge can be thickened, digested and dewatered better than WAS, resulting in better energy performances of the overall system (Tomei and Carozza, 2015).

Stability is generally defined for sludge as the point at which the organic matter is no longer available for rapid microbial activity, and this property is also associated with the concept of putrescibility and with odor generation. Stability could be quantitatively assessed through the volatile/total solids ratio (VS/TS) and/or the percentage of VS removed after sludge stabilization treatment (Braguglia et al., 2014).

In many WWTPs the nitrogen removal from anaerobically digested sludge has been studied in order to reduce the overload of nitrogen in the water line due to the return of the supernatant of the digested sludge to the head of the plant (Morras et al., 2015).

The addition of pretreatment steps before the digestion or the modification of the digestion temperature up to thermophilic conditions could improve the anaerobic stabilization process increasing hydrolysis rate and the quality of the digestate (Braguglia et al., 2015).

Another technology that can improve the digestate quality is the micro-aeration (i.e. the addition of very small amount of air). This method does not improve the reduction in total or volatile solids, but it improves the dewaterability of the sludge and reduces the accumulation of soluble COD and of foaming potential (Diak et al., 2013).

2.1. Conditioning of the sludge

The conditioning phase is essential because it can enhance the efficiency of dewatering, but at the same time it is critical due to the choice of the flocculant, of its dose and mixing conditions (Olivier et al., 2018). Acceptable dewatering rates are obtained through the use of conditioners, i.e. synthetic organic polymers or metal ions, especially iron, which make the dewatering faster without alterating the cake solids (Novak, 2006; Wei et al., 2018).
Dewatering can be improved through three different conditioning methods: chemical, physical and biological, or it can also be improved by the use of microwaves, ultrasounds or thermal methods (Wójcik & Stachowicz, 2019) (Figure 2.1.1).
Physical conditioning consists in changing the physico-chemical properties of the sludge through the addition of skeleton builders, the use of microwaves or ultrasounds; chemical conditioning relies on the addition of acids, surfactants or coagulants/flocculants (Wei et al., 2018).

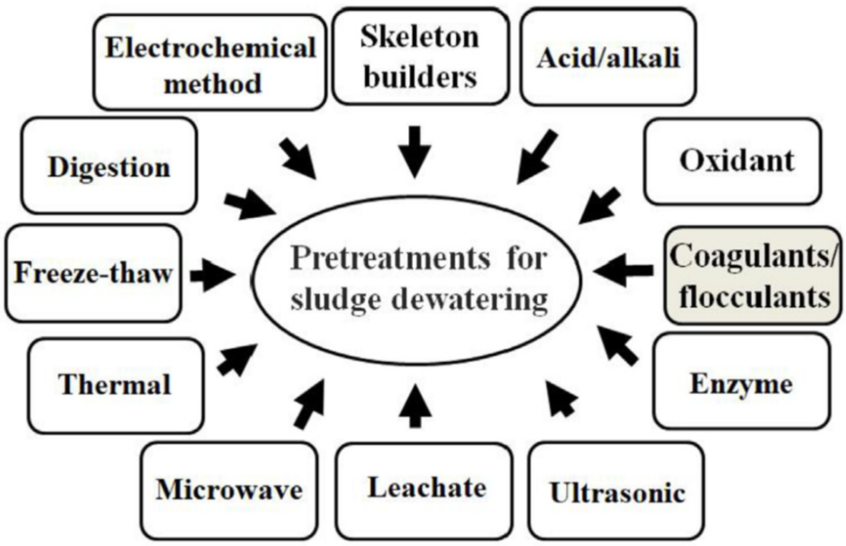

Figure 2.1.1_ Various conditioning methods for sludge dewatering (Wei et al,l 2018)

Regarding the physical conditioner, which is a neutral material, it was studied that it does not improve sludge dewatering significantly, but it only influences the mechanical strength of the flocs and the permeability of the sludge during filtration (Wójcik & Stachowicz, 2019). The use of chemicals is well known to obtain solid-liquid separation through the support of the aggregation of colloids into flocs; coagulation processes depend on the colloid characteristics such as charge, chemical composition and particle size. It was found that WAS particles are typically negatively charged, but it can also be possible to find both negatively and positively charged particles; in order to increase the efficiency of the dewatering it is essential adopt combinations of cationic and anionic polymers (Novak, 2010). The capability of chemical conditioning is linked to the transformation of the sludge structure, which increases the spaces among the particles in order to allow the release of more water during dewatering (Wójcik & Stachowicz, 2019). During the coagulation/flocculation process small colloidal particles form large “flocs” in order to improve the dewatering efficiency (figure 2.1.2). Coagulants are able to compensate the charge of WAS particles and thus to support their aggregation, also including suspended solids; once these particles are destabilized, they aggregate and settle down. The selection of the coagulant drives the efficiency of dewatering because each one has different structural characteristics (e.g. charge characteristics, ionic properties, functional groups and molecular weight) (Wei et al., 2018).

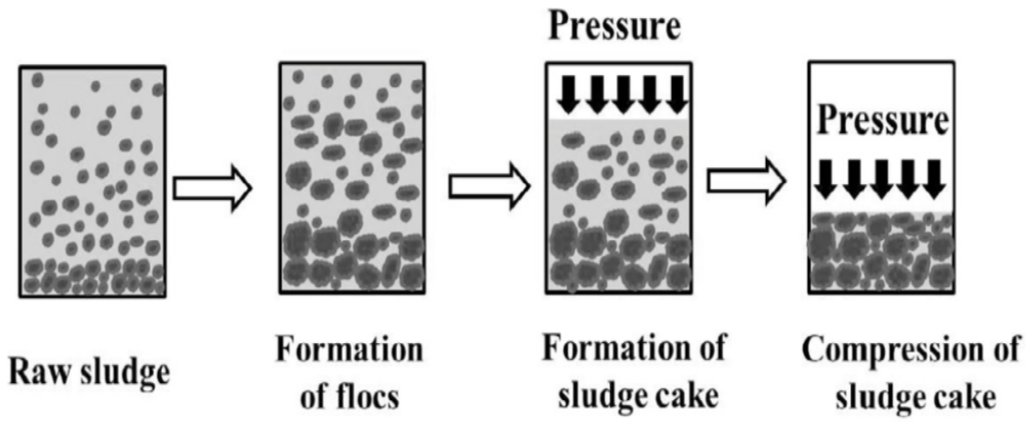

Figure 2.1.2_ Coagulation/flocculation and dewatering process (Wei et al., 2018)

Moreover, the mixing conditions between sludge and chemicals are an important aspect that should not be overlooked (Olivier et al., 2018). It was demonstrated that polymer dosing and mixing could affect floc size distribution (Ginisty et al., 2014). Synthetic organic polymers are frequently utilized as conditioners WAS due to their cationic charge, which reduces the repulsion among polymer molecules and biosludge particles, thus forming strong and large flocs; at the same time, the toxicity to aquatic systems of these polymers was discovered. An alternative solution could be to use surfactants and proteins even if until now the properties and mechanisms that procure the enhancement are little-known (Bonilla & Allen, 2018).

The use of cationic polyelectrolytes as conditioners is rather common in WWTPs; their addition neutralizes the charge of WAS particles and support their aggregation into larger agglomerates. However the sludge results more compact during filtration, thus influencing the further dewatering. Another negative aspect is the use of high doses of reagents and the high cost of the conditioners (Wójcik & Stachowicz, 2019).

Sometimes, in order to improve the efficiency of conditioning, chemical and physical agents are applied together. It was demonstrated that the dual conditioning improved dewatering performances due to the reduction of the dose of polyelectrolytes, such as polyacrylamide, which is partially replaced by biomass ash, and it also reduce the cost of dewatering (Wójcik & Stachowicz, 2019). The addition of biomass ash helps to form a rigid and permeable structure within the sludge, thus the sludge cake can remain permeable under pressure and

the water is easily removed. The dual conditioning is based on charge neutralization, bridging of particles and improvement of WAS structure. Therefore, the dual conditioning might be considered as a promising alternative in WAS management (Wójcik & Stachowicz, 2019).
Inorganic coagulants and synthetic polymeric flocculants are the most common and used conditioners, such as aluminum and ferric salts and poly-acrylamide (PAM) derivatives. Even if inorganic coagulants are cheaper and they increase the efficiency of dewatering, they also have negative aspects because of the high dose needed, their sensitivity to pH and the biological toxicity due to the release of residual metal ions. It is also true for synthetic polymeric flocculants: their positive aspects are related to the dose, floc size and dewatering performances, but their cost is high and they also contribute to spread secondary pollution and health risks because of their residual monomers (Wei et al., 2018).
The optimal coagulant dose is the one that corresponds to the lowest capillary suction time (CST) needed to evaluate the sludge dewaterability. The selection of the coagulant, its dose and the way of adding is usually based on empiric procedures (Olivier et al., 2018).
The calculation of the drainage index is also used to evaluate and quantify which flocculant parameters lead to the highest dewatering efficiency. Drainage index (Eg) was studied to optimize sludge thickening and sludge dewatering in a filter press. The index was evaluated considering the following parameters: capture rate (C_{GD}), final sludge dryness (T_{GD}) and drainage kinetics (K_{GD}):

$$E_g = \ln\left(\frac{T_{GD}}{k_{GD}^3 \times C_{GD}^{0.25}}\right)$$

The drainage index can be easily measured and it can be employed to set the operating conditions of the flocculation (Olivier et al., 2018) (figure 2.1.3).

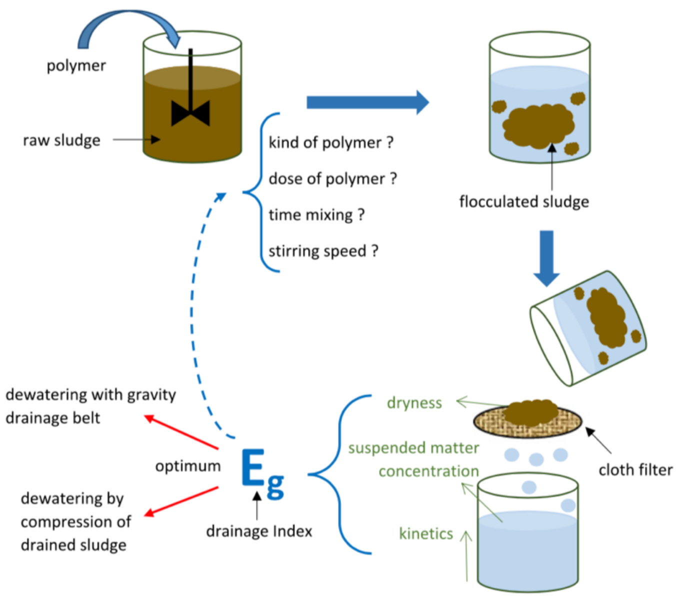

Figure 2.1.3_ Procedure for the investigation of the operating conditions of a coagulation/flocculation process (Olivier et al., 2018)

Different parameters, such as pH and temperature, can also improve conditioning performances. About pH, the surface charges of colloids and coagulants/flocculants vary according to the different pH values. The charge neutralization effect is affected by pH and usually increasing its value enhances the negative charges of colloidal particles. Decreasing temperature, the optimal doses of coagulants should increase. This is due to the fact that decreasing temperature the viscosity of the liquid increases, inhibiting the Brownian movements of colloids and consequently decreasing also their agglomeration (Wei et al., 2018). On the other hand, high temperature can improve the solubility and activity of polymeric flocculants (Wei et al., 2018).

It was found that the most difficult type of sludge to dewater is WAS, which is composed by a colloidal suspension of microbial aggregates and by a gel-like matrix of extracellular polymeric substances that hampers and makes harder the dewatering (Bonilla & Allen, 2018). In order to improve and facilitate WAS dewaterability, different experiments were conducted to analyze different type of flocculants and the optimal doses required. Total solids (TS), total suspended solids (TSS), total volatile solids (TVS) and volatile suspended solids (VSS) were considered in the experiments (Novak, 2010).

2.2. Dewatering process

In order to evaluate the critical issues of dewatering, an important parameter that should be taken into account is the moisture distribution in the sludge (Vaxelaire & Cézac, 2004). It was found that the volume and quality of sludge with a 80 % moisture content (Mc) is two times smaller than the one of a sludge with 90 % Mc, and also that a sludge with a high Mc is more difficult to transport, put in a landfill, dry, incinerate or recycle because of the high costs (Rao et al., 2019). Dewatering is necessary in order to reduce the volume of digestate and the cost of the following processes (transportation, storage and other post-treatments) (Lü et al., 2015). It was also found that the Mc decreased if the dosage of conditioner increased so the two parameters are inversely proportional (Wójcik & Stachowicz, 2019). Another aspect is that the solid - liquid separation can reduce up to 60 % of transport costs and 25 % of the drying step (Lü et al., 2015).

There are different types of sludge and it was studied that the hardest to dewater is WAS; the first thing to evaluate is how the water is distributed within activated sludge (Vaxelaire & Cézac, 2004). Water within WAS is commonly divided in four fractions:

- Free water: it can be removed by gravity settling or by mechanical stress;
- Interstitial water: it is trapped between particles and inside the flocs and it can be removed by mechanical dewatering (it is similar to the water retained in a sponge);
- Surface water: is that one contained near surfaces, this kind of water is difficult to remove and it can be eliminated only drying;

- Bound water: is present in lowest quantity and it can be removed only with high temperatures, it can be removed by dewatering but not completely. (Novak, 2006)

All the properties of the water such as vapor pressure, enthalpy, entropy, viscosity and density, depend on the presence of solids in the sludge. That is the reason why during dewatering the behavior of a single molecule of water is dependent on how much it is near to the solid particles. The free water is not affected by the presence of solid particles whereas the properties of the bound water are influenced by that. Furthermore, it was studied that the bound water can be detected and distinguished from the free water because it remains unfrozen at temperatures below the freezing point (Vaxelaire & Cézac, 2004). It was also found that one of the factors that mostly influence the dewatering performances is the content of bound water, but some researchers think that bound water cannot reflect sludge dewaterability as the free water because it is present in small portion in the sludge and also because of its strong binding characteristics (Wei et al., 2018). Moreover, with the addition of some chemical conditioners to the sludge, the amount of free water increases, but in case of overdose the bound water increased (instead of the free one) and it was assumed that the reason could be the absorption of moisture inside the polymer segments (Vaxelaire & Cézac, 2004).

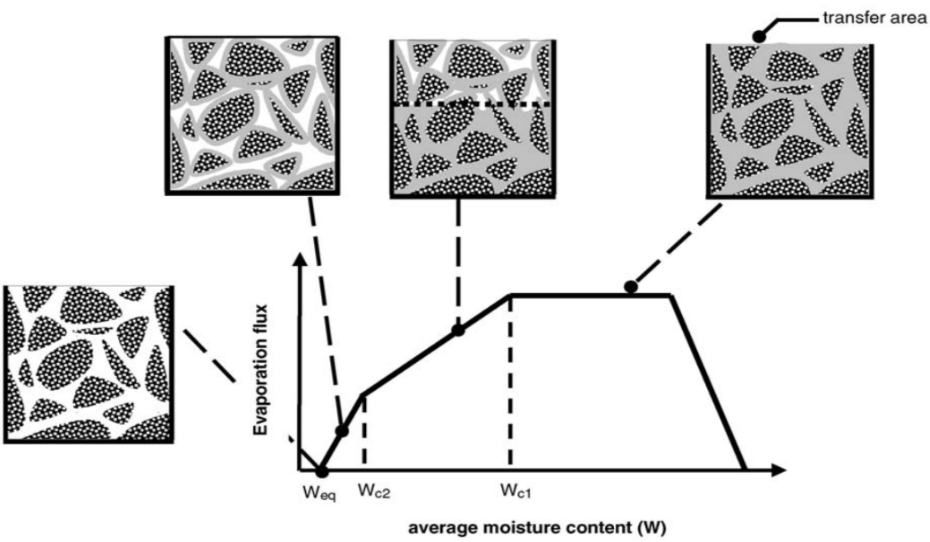

Figure 2.2.1_ classical drying curve (Vaxelaire et al., 2004)

It is difficult to have a clear idea of the distribution of water in WAS with drying test, however the rate of evaporation of the water (Figure 2.2.1) depends on the type of bond between water and the solid particles and it is important in order to study sludge conditioning or to predict its dewaterability, although it is not always of practical use. Even if the distribution of water in the activated sludge should be constantly and precisely described, the drying curve cannot be used as a single index to predict dewatering performances because the

solid-liquid separation is due to different and numerous phenomena. That is the reason why one single index is not enough to describe the entire operation (Vaxelaire & Cézac, 2004).
Another aspect that influence dewatering performances is the presence of solids inside the sludge, which is different according to the different pollutants contained in the wastewater plants or to the different characteristics that these plants present (Wei et al., 2018). It was investigated also the influence on dewatering of proteins and polysaccharides in order to determine which size of the particles and aggregates were better to obtain. It was demonstrated that small particles would not be retained by the filters and they would not contribute to the specific resistance (Novak, 2010).
Dewatering processes are usually implemented through mechanical devices. Three important elements should be taken into account in dewatering; the first is the time at which the liquid is removed from the sludge; the second is the amount of solids in the dewatered material, and the third is the quantity and the type of chemicals added during the conditioning phase (Novak, 2010). Dewatering consists of two different phases: filtration and expression. In the first one the water is removed until when sludge particles begin to be in contact, resulting in solid pressure. During the second phase, once the normalized liquid pressure decreased below 1, with the continued application of pressure, the particles are deformed and the water is squeezed from both the sludge particle and the particles and consist. The amount of water removed by filtration alone is not sufficient to obtain a dewatered cake; dry cakes are achieved through the expression (Novak, 2006).
One of the first questions that should be asked is why the sludges could not be dewatered so easily; the answer is ascribable to the presence in the sludge of extra cellular polymeric substances (EPS). The term EPS describes the cell exudates and includes a range of compounds such as proteins, polysaccharides, nucleic acids and phospholipids (Skinner et al., 2015). EPS are mainly high-molecular-weight secretions from microorganisms, products of cellular lysis and hydrolytes of macromolecules, all exhibiting a critical role in holding huge quantities of bound water by chemical hydration, crystallization, adsorption and osmotic effect (Zhang et al., 2018). EPS keep microbial aggregates together in a three-dimensional matrix and affect biodegradability, hydrophilicity/hydrophobicity and fluidity of WAS (Miryahyaei et al., 2019). In fact, the macromolecular components in EPS can decrease the digestate dewaterability (Zhang et al., 2018). Dewatering is therefore more difficult because after the formation of an extremely compressible and impermeable layer the normal mode of diffusion of solid material into the filter cake changes significantly (Skinner et al., 2015). After the increase of the sludge solid concentrations, the EPS polymeric structure become stronger, the viscosity and the fluidity of the sludge are changed and several consequences could happen such as short-circuiting the substrate in the reactor, formation of stagnant regions and reduction of the effective reactor volume or also contact between substrate and active microorganisms. All these issues can decrease the mass transfer and the heat transfer rate, not allowing the sludge to reach the operating temperature of the anaerobic digester

and impacting at the end also the bioreaction rate. Furthermore the high EPS content makes more difficult the dewatering process increasing the viscosity of the sludge (Miryahyaei et al., 2019).

As EPS have many charged functional groups, they promote physico-chemical interactions such as electrostatic, hydrophobic, Van deer Waals forces, hydrogen-bonding and adsorption. During anaerobic digestion, EPS transfers from particles into the bulk fluid and makes the flow easier increasing the functional group and surface charge of the bulk fluid (Miryahyaei et al., 2019). EPS is considered a key factor in dewatering and it is also related to different physico-chemical properties of WAS such as surface charge, moisture content, compressibility, hydrophobicity and viscosity (Liu et al., 2019).

In order to improve the dewaterability, pretreatments could be based on thermal hydrolysis processes and/or chemical ones in order to reduce or destroy EPS (Skinner et al., 2015). The dewaterability of digestate increased with an enhancement of EPS solubilization, which achieved the release of partially-bound water (Zhang et al., 2018). The variations of sludge dewaterability have been explained by differences in composition and distribution of EPS, but the results of performed studies showed contradictions among different processes (Zhang et al., 2018), and the performances of dewaterability cannot be foreseen only from the EPS content (Zhang et al., 2018).

Among physical conditioning processes, thermal pretreatment reduces the viscosity of the sludge and releases EPS, thus improving the dewaterability. It was also found that thermal pretreatment could deteriorate the dewaterability in the range of temperature between 30° and 170°C because of the increase of EPS content. (Liu et al., 2019)

Dryness limit, corresponding to the cake dryness obtained after an infinite operating time, is one of the parameters that can assess dewatering processes efficiency, which is affected by the properties of conditioned sludge and by operating conditions. The method used to evaluate this parameter is a filtration-compression test at lab-scale and a standard method is being studied in France in order to improve the controlled procedure. Dryness limit depends on final cake thickness, expression pressure and compression time. Recently a new simple procedure has been studied in order to perform the test in less time (4 hours) with a lower pressure (4 bar) and a measurable cake thickness (5mm) and also in order to have a reference value (Ginisty et al., 2014).

Due to the fact that there is a lack of quantitative and fast laboratory techniques that characterize dewaterability, many laboratories turned to empirical testing (Skinner et al., 2015). Capillary suction time (CST) and specific resistance to filtration (SRF) are two important tests that can evaluate sludge dewaterability, but they are empirical and not very accurate (Wei et al., 2018), and they are used to understand the trends in the dewaterability of sludges that depend especially on the addition of flocculants (Skinner et al., 2015). The CST was also used to evaluate the conditioners effect on WAS dewaterability; it is based on the use of two electrodes: when the water reaches the first, after it travelled through a filter

paper, a timer counts the seconds used to reach the second one and then the timer stops. This time used to travel from the first to the second electrode it the CST and a lower CST means good dewaterability (Bonilla & Allen, 2018a). This test is one of the most applied and it is known to be sensitive to changes in temperature, for this reason with higher temperatures the CST tends to become lower due to the decrease of the viscosity of the sludge and due to the flocculation properties of the sludge. Indeed, the higher temperatures influence the density and chemical composition of the suspended particles and also alter setteability and desorptivity (Sawalha & Scholz, 2012). An example of CST values could be less than 20 s for a readily dewaterable sludge, whereas for poorly dewatering sludges the CST will be higher and may exceed 50000 s (Novak, 2006).
Specific resistance of filtration (SRF) is another criterion used to characterize sludge dewaterability. Its measurement requires recording the amount of filtrate removed over time at constant pressure and it can be carried out with a Buchner funnel under vacuum or with a filtration/compression cell. With the optimal dose it will be obtained the lowest SRF (Olivier et al., 2018). SRF results to be more complex, expensive and time consuming compared with CST test (simpler and inexpensive), so recently it was developed a prediction methodology in order to obtain SRF test results from the data generated by CST tests under different experimental conditions. Around 2007 it was discovered a relationship between the CST test results and the SRF test findings based on SRF theory:

$$\mathrm{CST} = c_1 \times \mathrm{SRF} \times \mu \times W + c_2 \times \mu$$

where c_1 and c_2 are empirical coefficients related to CST; μ is the viscosity [Pas] of the filtrate and W is the solid content per unit volume of the filtrate. This is the first model based on empirical data sets which predict SRF results from CST data (Sawalha & Scholz, 2010).
In order to carry out sludge dewatering in an economic way the use of mechanical devices such as filters or centrifuges is common because of their low power consumption (Olivier et al., 2018). The process efficiency is mostly affected by sludge properties and linked to organic content, state of water and rheological properties (Ginisty et al., 2014). However each sludge has a different ability to retain water and this influences the quantity of water that can be removed not considering which dewatering device is used. Therefore the nature of the sludge should be taken into account in order to select the best dewatering process to use (Novak, 2006).
One of the most prevalent dewatering devices used nowadays are the filter presses which involve two different dewatering phases: filtration and compaction of the produced cake. The conditioned sludge is inserted, during the first step, inside the filter chambers with the use of a volumetric pump; during the second step the membrane is compressed applying a pressure over 8 bars and the cake is produced. The second step increases the sludge dryness, but its effectiveness decreases if the filtration pressure increases. This is due to the higher extraction of free water in the first step because of the higher filtration pressure; the

compacting step is less efficient because a large part of free water has already been extracted. There are many operating parameters that have to be optimized such as the filtration pression, the pressure increase rate, the pressure holding time and the thickness of the cake (Tosoni, Baudez, & Girault, 2015). It was also studied that increasing the pressure the filtration rate doesn't increase; the explanation could be that some water molecules are more difficult to remove then others and so it will be more difficult also to dewater it (Skinner et al., 2015).

The wastewater treatment technology, the type of sewage sludge, the composition of sludge and the method of dewatering are different factors on which the efficiency of sewage sludge dewatering depends on (Wójcik & Stachowicz, 2019). A parameter which strongly interferes with the final dryness during the mechanical dewatering and which improves the kinetics is the cake thickness. This parameter is the most sensitive to the applied pressure for waste activated sludge dewatering in filter press. After many studies it wasn't observed a direct correlation between the pressure gradient and dryness of the cakes (Tosoni et al., 2015).

Another potential and possible method used to characterize the sludge structure and to understand the mechanisms of dewatering after anaerobic digestion is the rheological characterization of the sludge (Zhang et al., 2018). The rheological behavior of WAS also has an important role in heat and mass transfer during the anaerobic digestion (Miryahyaei et al., 2019). It was also studied that rheological characteristics of digested sludge are strongly dependent on changes in total solid concentration, temperature and polymer dose (Yeneneh et al., 2016). The rheological behavior of the sludge deeply affects the degree of decomposition of the flocs and it can improve or decrease the level of released organic matter. The higher fluidity of the sludge can improve biogas production and dewaterability because it can help to release macromolecular components and bound water during anaerobic digestion. Another parameter that can predict the dewaterability performances is the yield stress of sludge before entering the digester; as an example, a reduction of 12 % in yield stress leads to a 12 % dewaterability increase (Miryahyaei et al., 2019).

The rheological characteristics of WAS depend on many factors: WAS origin, total solid concentration (which affects the most the rheology), temperature (which results in change of the shape and size of flocculated particles), polymer dose (which affects the dewaterability and the operational cost of dewatering), and sludge treatment method. The effect of total solid concentration results to be stronger than the temperature because of the increase in total solid content and size of particles in suspension that lead to a stronger interparticle interactions and consequently result in higher apparent viscosity and yield stress (Yeneneh et al., 2016). Considering the polymer dose, it is important to choose the right one in order to obtain an acceptable separation of water and a good dewatering capacity (and obviously to optimize the costs) and not a high dose which could lead to an overdose condition and a not realistic representation of the flocculation, due to the formation of big flocs structure which negatively affect dewaterability (Yeneneh et al., 2016).

Higher temperature enhances dewatering performance for unconditioned pre-sheared digested sludge, but in case of polymer-conditioned digestate at 35 °C the optimum dewaterability can be achieved. Whereas considering the total solid concentration it was found that the smaller concentration increases the dewatering efficiency. So, in order to improve the dewatering, it should be important enhance the performance of anaerobic digesters in order to reduce the total solid and volatile solid (Yeneneh et al., 2016).

It was also found that the dewaterability of sludge for the low-solids anaerobic digestion system was deteriorating during the passing of time and that the level of dewaterability was not stable: improved and deteriorated during the time or maintained roughly constant. In order to improve and to increase the digestion efficiency, the dewaterability and the fluidity within digesters, different pretreatments were applied: thermal hydrolysis, ultrasound, enzymatic, electrochemical, microwave or Fenton and many others (Zhang et al., 2018).

Another parameter that can change the sludge dewaterability is the temperature; it was demonstrated that the dewaterability can be deteriorated by a thermal pretreatment in the range between 20 and 170 °C, but above a temperature threshold of 120 and 150 °C it was found that the deterioration decreased. This is possible because the factors that affected dewaterability in the two temperature ranges (20-105 °C and 105-170 °C) were different. It was evaluated through CST tests that the dewaterability decreased in the range between 20 and 135 °C and then increase from 135 until 170°C. This means that different effects on dewaterability are linked to different treatment temperatures and that around 135°C there was an extreme in which the poorest dewaterability was achieved (Liu et al., 2019). This is due to the fact that increasing the temperature also the ionization of organic compounds (protein-like substances, humic acids (HA) and PS) increases. Therefore, the negative charge and electrostatic repulsion between the sludge particles increased with the decrease of the zeta potential, resulting in the deterioration of the dewaterability. Whereas with the increase of the temperature to 170 °C some anionic groups are destabilized and lead to combine ionic groups and then the ionization of anionic functional groups is weakened. Therefore the negative charge on the flocs surface and the electrostatic repulsion between the particles decreased and the zeta potential increased leading to the improvement of the dewaterability. It was discovered that the dewaterability in the two temperature ranges was affected by different factors such as EPS content, proteins, polysaccharides and humic acids content (Liu et al., 2019).

Moreover, another parameter affecting dewaterability is the particle size of flocs; it was found an important correlation between the mean particle size and the CST. "Supracolloidal" particles (1-100 µm) had the highest effect on the dewaterability of the flocs, which was determined by SRF expressing the compact-ability of sludge flocs (SHAO, et al., 2009).

2.3. Final disposal

Improper handling and disposal of organic solid waste heavily affects air, land, water, human health and climate change due to the release of methane. That is the reason why management of digestate is obtaining great importance from an economic and environmental point of view recently (Zeshan & Visvanathan, 2014). The sustainable management of sewage sludge is a complicated challenge for both governmental and private organizations (Braguglia et al., 2015).

The sustainable management of sewage sludge should be improved through the development of innovative technologies and management systems, also based on energetic valorization perspectives (Panepinto et al., 2016). Also policy actions oriented towards climate change mitigation and renewable energy use can also influence sludge management (Kacprzak et al., 2017).

The sludge dewatering is essential in the sludge disposal due to the efficient separation of liquid and solid in sludge and also due to the effective reduction of the final processing costs (treatment, transportation and final disposal) (Wei et al., 2018). As a matter of fact the cost of sludge treatment and disposal operations are deeply affected by the volume that has to be treated and the water content, that is the reason why 30 - 50 % of annual treatment costs in WWTPs are related to the sludge dewatering process (Ginisty et al., 2014) . The water contained in the sludge is about 99 % by weight; since the sludge in the liquid state is difficult to handle, and the transport is expensive, the removal of water through dewatering will help to store and transport the sludge (Tosoni et al., 2015).

It is known that dewatered sewage sludge contains on average 50 - 70 % of organic matter, 30 – 50 % mineral components, 3.4 - 4.0 % N, 0.5-2.5% P and significant amounts of other nutrients and micronutrients depending on stabilization processes (Kacprzak et al., 2017). Due to the content of heavy metals, organic compounds and microorganisms, sewage sludge should be properly treated (Wójcik & Stachowicz, 2019). It is also important to remember that is necessary a proper management strategy because from digestate there are different kinds of emissions, such as methane, which can contribute to the climate change, ammonia, carbon dioxide and nitrous oxide, which cause locally or globally pollution (Zeshan & Visvanathan, 2014; Masuda et al., 2018)) There are different option to reuse the sludge as a nutrient-rich fertilizer, after dewatering and composting (Fang, Li, Zhang, & Xie, 2019), as a supplement to composting or as a feedstock to energy production (Skinner et al., 2015). However digestate can contain pathogens that can affect human health and can lead to accumulation of toxic substances in the soil (Braguglia et al., 2015). That is the reason why the European Union (EU) sets limits for several heavy metals in a sludge directive (86/278/EEC, CEC 1986).

Figure 2.3.1 shows the different amount of GHG emissions produced by each step of AD processes. It was also found that among different kinds of digestate, that one with the highest GHG emission potential was the raw digestate, whereas for the stored one it is about 10 % less and for the stored-cured one 42 % less; this decrease of GHG potential is due to the treatment and management of the digestate. Moreover, with the decrease of the total solids in a dry anaerobic digester also the volatile solids decrease and this implies that also GHG potential decreases because it is directly linked to its volatile solids content (Zeshan & Visvanathan, 2014). It was also found that GHG emissions of sludge treated with AD were lower than without it (Fang et al., 2019). An important fact that should not be understimated is that CH_4 and N_2O can be produced from treatment units both directly and indirectly as shown in table 2.3.1 (Masuda et al., 2018). Direct GHG emissions are produced during wastewater and sewage sludge treatment, whereas the indirect ones are related to the consumption of electricity, burning fossil fuel for transport, the use of chemical or the disposal of the sludge. (Parravicini et al., 2016). Methane is emitted in the WWTP, especially in the grit chamber, in the primary sedimentation tank and in the oxidation tank. Moreover, CH_4 and N_2O characteristics are different for each plant and depend on the characteristics of the STP. N_2O emissions can be gaseous (from the oxidation tank) or dissolved through the effluent,and the indirect emission of N_2O has to be considered as the main source of N_2O from STPs (Masuda et al., 2018). The impact of indirect CO_2e emission caused by electricity consumption of the WWTP is considered much lower compared to the direct N_2O emission (Parravicini et al., 2016).

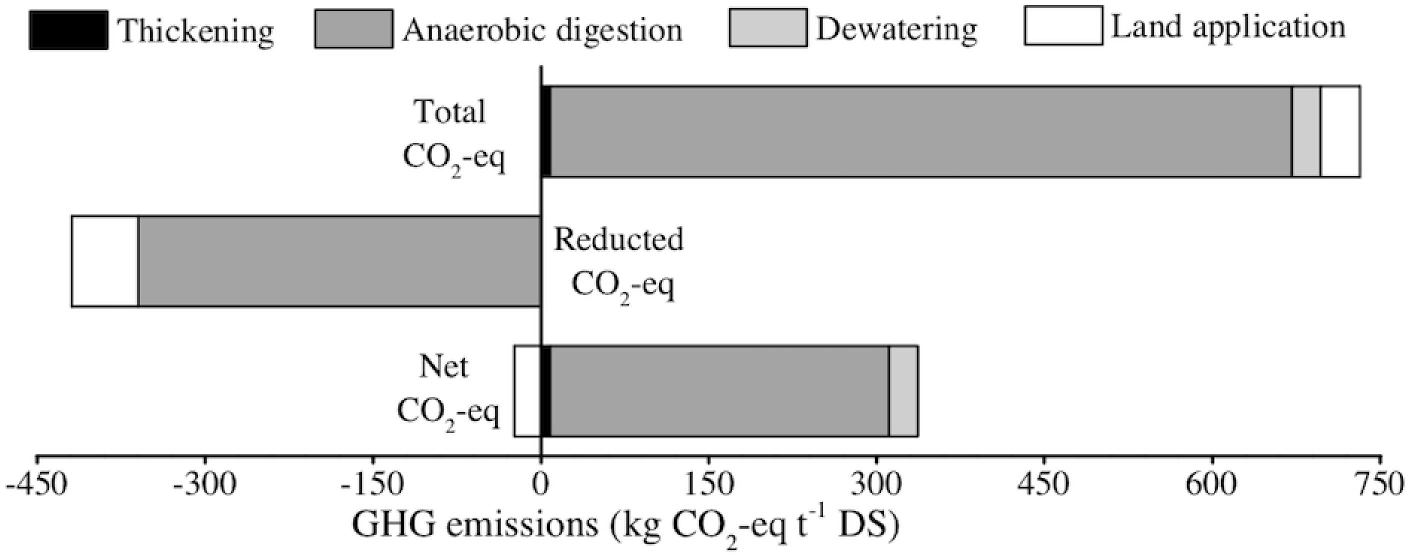

Figure 2.3.1_Greenhouse gas (GHG) emissions rate from the optimal technical route after increasing the methane production rate of sludge treatment and disposal (Fang et al., 2019)

Table 2.3.1_ Direct and indirect GHG emission from wastewater collection, treatment and discharge (Parravicini et al., 2016)

Direct GHG emissions	Indirect GHG emissions
Wastewater collection (sewer system)	Electricity supply
Wastewater treatment (WWTP)	Transportation (e.g. chemicals, sewage sludge)
Wastewater discharge in water bodies	Use of chemicals and additives (including GHG emissions in the upstream stages of production)
	Disposal/reuse of residuals (e.g. biosolids)

It was also discoverd with the carbon footprint analysis that the direct N_2O and CH_4 emissions significantly impact the CO_{2e} balance of municipal WWTPs, and this impact could increase even more if the electricity would be generated exclusively using renewable sources. Anyway the carbon footprint anlyses reveal that GHG emissions from municipal WWTPs have a small impact from a global scale poin of view (Parravicini et al., 2016).
In addition (figure 2.3.2), the digestate can be dewatered in order to separate solid and liquid fractions and obtain an easy handling, transport and disposal.

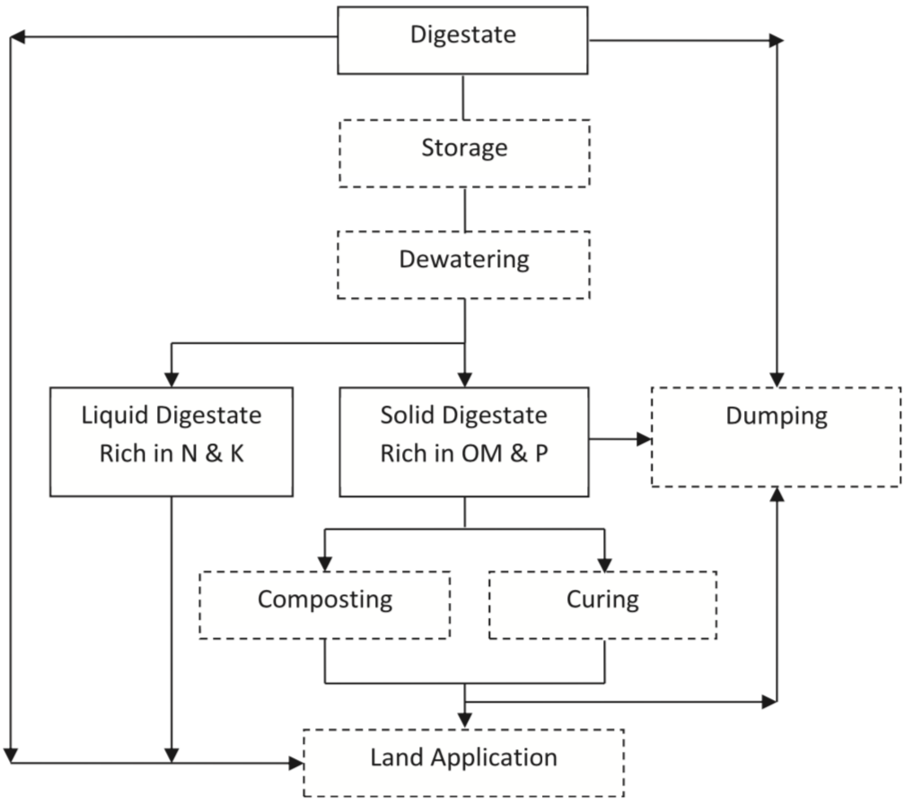

Fig. 1. Possible unit processes of digestate management system.

Figure 2.3.2 Possible unit processes of digestate management system (Zeshan et al., 2014)

It was discovered that once the digestate is dewatered, the liquid fraction could be used as a fertilizer in agriculture, recycled back the anaerobic digestion in order to dilute a new waste steam, treated in a wastewater treatment plant or discharged into sewage. Whereas the solid part can be matured by composting or it can also be used as landfill cover after the stabilization of its organic contents (Zeshan & Visvanathan, 2014).
From table 2.3.2 it can be seen that the most important and common disposal approaches are landfilling, incineration and agricultural use. The landfilling should have a significant and continue decrease from 33 % to 15 % from 2005 to 2016 (Eurostat, 2018) due to the fact that it consumes a large area of land, increases the potential of landfill leachate to groundwater and poses risks to human health (Fang et al., 2019); it also causes the CO_2 emissions directly

to the air and it is not an efficient environmental approach (Kacprzak et al., 2017). Between 2005 and 2016, sludge incineration almost doubled from 11 % to 21 % and also agricultural use and composting increase from 48 % to 54 % (Eurostat, 2018). The thermal valorization should be considered as an energy recovery especially for the large wastewater treatment plants that are present in large metropolitan areas. Moreover, considering the strict limitations of landfilling and agricultural reuse, incineration should play a key role in the long term (Panepinto et al., 2016).

As regards agricultural reuse, it was observed that in 2010 in Belgium, Denmark, Spain, France, Ireland and the United Kingdom more than 50 % of the sludge was used for this purpose; but in other countries such as Finland agricultural reuse was less than 5 %. (Kacprzak et al., 2017). Agricultural reuse is becoming a common and new disposal, but it requires precise analyses on the toxicity of the sludge according to the law (Kacprzak et al., 2017).

Digestate has to be stored before its application to agricultural land; but during storage the solids and nutrients concentration decrease; this phenomenon could be attributed to the ambient conditions such as high pH, high moisture or high temperature, which interfere with the microorganisms that are responsible for this loss of nutrients. It is implied that storage time should be reduced to avoid GHG emission and nutrient loss, furthermore storage should involve dewatered sludge (Zeshan & Visvanathan, 2014). A combination of anaerobic digestion and land application should be the most environmentally sustainable perspective because of the lowest GHG emissions and energy consumption. Moreover, the use of sludge as fertilizer was the best option also from an economic point of view (Fang et al., 2019).

Anyway, incineration is nowadays the most attractive option in Europe; in Denmark 24 % of the total sludge produced is incinerated, whereas in France is the 20 %, in Belgium the 15 % and in Germany the 14 %; in Italy is only the 2 % (Panepinto et al., 2016). Sludge incineration contributes to ozone depletion, photochemical oxidation formation and terrestrial ecotoxicity and it has high costs; another possible way to reuse the sludge is using it as a building material after dewatering and incineration (Fang et al., 2019).

Sewage sludge should be considered as a source of energy and of organic matter, carbon, phosphorus, nitrogen, volatile acids and proteins. It is also important consider that all sewage management strategies should include market analysis of the final products, estimate the size of the markets and determine the time necessary for the possible solution. It is important that these strategies should reject solutions with high potential of risk, taking into account that the main aim is sewage sludge disposal and long-term nature of the applied solution (Kacprzak et al., 2017).

The average costs of different wastewater treatment and disposal is estimated to be 160-210 [euro/tons dry matter, DM] for the non-treated sludge and around 210-300 [euro/tons DM] for the dewatered sludge in agriculture or forestry (Kacprzak et al., 2017).

Table 2.3.2_ Sewage sludge management in selected EU countries (Kacprak et al.,2017)

UE country	Sewage sludge - total disposal (10³ t)		Sewage sludge – landfill (10³ t)		Sewage sludge - incineration (10³ t)		Sewage sludge - agricultural use (10³ t)		Sewage sludge - compost and other application (10³ t)		Sewage sludge - dumping at sea (10³ t)	
	2002	2013	2002	2013	2002	2013	2002	2013	2002	2013	2002	2013
Belgium	113.66	0	12.95	n/a	71.05	n/a	18.97	n/a	0	n/a	0	n/a
Germany	n/a	1794.73	n/a	0	n/a	1034.77	n/a	491.33	n/a	264.4	n/a	0
Estonia	n/a	18.79	n/a	1.81	n/a	n/a	n/a	0.29	n/a	16.27	n/a	0.42
Greece	77.65	n/a	77.65	n/a	0	n/a	0	n/a	0	n/a	0	n/a
France	n/a	869.74	n/a	30.92	n/a	160.63	n/a	368.58	n/a	287.49	n/a	0
Latvia	21.46	20.74	0	0.24	0	0	3.15	7.48	1.42	2.3	0	0
Netherlands	353.9	n/a	39.8	n/a	204.3	n/a	0	n/a	51.1	n/a	0	n/a
Poland	435.74	540.3	192.49	31.4	6.78	72.9	67	105.4	26.54	32.6	n/a	0
Slovakia	51.27	57.43	4.44	6.64	n/a	5.01	41.96	0.52	n/a	35.21	0	0
United Kingdom	1533.82	n/a	123.96	n/a	305.82	n/a	842.53	n/a	n/a	n/a	n/a	n/a
Norway	103.13	131.2	16.09	18.6	n/a	n/a	43.56	82.6	14.71	29.9	n/a	n/a
Switzerland	200	194.5	4	0	153	188.3	38	0	5	0	n/a	0

n/a – data not available.

3. Materials and Methods

3.1. Collection of samples

The sludge samples studied and analyzed in this thesis were collected in two different places between March and July 2019. The principal aim of collecting different samples was to analyze, evaluate and compare the dewaterability of different sludges using a flocculant and subsequently a filter-press.

The first set of samples were the influent and effluent from the post-aeration pilot plant in the Central Prague Wastewater Treatment Plant; whereas the second one were mesophilic, thermophilic and TPAD2 sludges from the lab-scale anaerobic digesters in the university.

The collection of both types of sludges was performed every two weeks, in order to work one week on influent and effluent and the other one on the anaerobic digesters' sludges.

In May 2019 the analyzes on the sludge from the post aeration pilot plant were suspended for three weeks in order to analyze every week the other sludges.

On July 2019 just the pilot plant was analyzed more for the last two weeks.

The quantities were different for each sludge, for the influent and effluent 1 liter was sampled every two weeks, whereas for the mesophilic, thermophilic and TPAD2 500 ml were collected every time.

The last week (July 2019) a mesophilic sludge from another wastewater treatment plant from Ceska Lipa was sampled.

3.2. Characterization of the samples

The samples were characterized as it can be seen in table 3.2.1. The values of the dry mass for the thermophilic (T), mesophilic (M) and TPAD2 were changing every week, and it was a critical issue in the optimization of the flocculant dose.

Table 3.2.2_ Dry masses of the different sludges

<table>
<tr><th>Type of sludge</th><th>Dry mass [g/l]</th></tr>
<tr><td>INFLUENT post-aeration</td><td rowspan="2">23,8</td></tr>
<tr><td>EFFLUENT post-aeration</td></tr>
<tr><td>T</td><td rowspan="2">50-60</td></tr>
<tr><td>M</td></tr>
<tr><td>TPAD2</td><td>52-58</td></tr>
</table>

3.3. Flocculant choice

Different types of flocculants were tested during this thesis. For the first period (March – June 2019) it was always used the same type which was in powder. The powdered flocculant was the same used at the Prague WWTP, where dewatering is performed through a centrifuge. During the last month (July 2019) two liquid flocculants (EM440 and EM640) were tested. It was found that the liquid product didn't create good flocks with the considered sludge samples, so it was chosen to use the powdered coagulant.

First of all, the flocculant should be prepared. Using the powdered flocculant, the right amount in milliliters should be calculated considering the dose of active flocculant and the initial dry mass of the sludge; for the calculation there two proportion should be used:

$$x_1 : d_{PE}[gAF] = C\left[\frac{g}{ml}\right] : 1000$$

Where the d_{PE} is the quantity of active flocculant that it should be used in grams, C is the concentration of sludge per 100 ml considering the dry mass of the sludge and considering 1 l.

The powdered flocculant is active for 100% so its volume could be found easily multiplying the value founded for 1000 and dividing it for the number of the concentration of the flocculant:

$$x_2 : 1000 = x_1 : Cf\left[\frac{g}{ml}\right]$$

Where x_2 is the PE amount in ml and C_f the concentration of the flocculant solution.

In order to obtain the flocculant dose (figure 3.3.2), 250 ml of tap water was introduced in a beaker on a magnetic mixer (figure 3.3.1) at 500-600 rpm, later 2.5 g of flocculant was added with 250 ml water, in order to obtain 500 ml of diluted flocculant continuing to mix until the flocks disappear.

Figure 3.3.3_ magnetic mixer

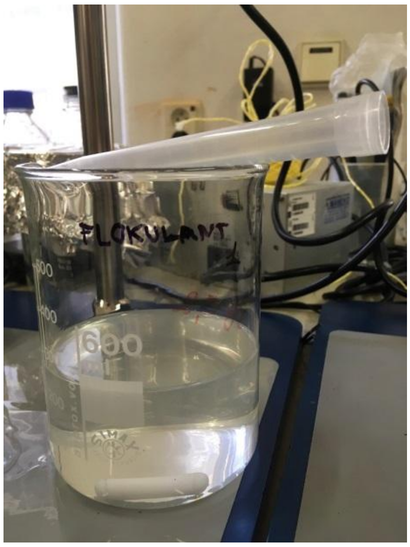

Figure 3.3.2_ Flocculant 5[g/l]

After the preparation of the diluted flocculant and the calculation of its amount, different tests with different doses were performed in order to define the optimal dose for all the considered sludges. At the beginning, all the tests were made using the magnetic mixer, once the flocculant was added the speed of 700-1000 rpm for 20 seconds (rapid mixing phase) and then 250 rpm for 3 minutes (slow mixing phase).
During this first step, the best way to choose the optimal amount of flocculant was using the visual evaluation. In the figure 3.3.3 three different doses of flocculant for the same quantity of sludge studied are shown.

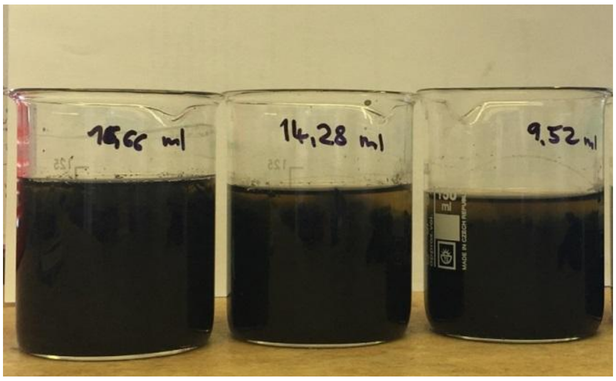

Figure 3.3.3 _ comparison between different quantities of flocculant in the sludge

In the first weeks the right doses of flocculant for each sludge were investigated. It was found that the best dose for the Influent was 35 [gPEeff/kgTS] whereas for the Effluent was 25[gPEeff/kgTS]. For the sludges from the lab-scale anaerobic digesters it was more complicated because the dry mass was different every week, as before mentioned. The last three weeks of experimental activity it was decided to keep the same dose for all the tests although the dry mass was changing. For the mesophilic sludge the dose used was 35 [gPEeff/kgTS], for the thermophilic sludge it was 50 [gPEeff/kgTS] and for the TPAD2 sludge it was 40 [gPEeff/kgTS].
Due to the high density of the sludge samples, the mixing with the magnetic stirrer was also more difficult, so it was adopted mechanical mixing (Figure 3.3.4) since the end of April 2019. The adopted operating conditions were 100-200 rpm, then increased to 500 rpm, for 2 minutes. During the last month just the mechanical mixer for the fast mixing was used, once the flocculant was added it was just mixed with the velocity of 500 rpm for 2 minutes.

Figure 3.3.4_ Mechanical mixer

3.4. Dewaterability

The second step of the procedure was to dewater the samples in a Mareco mini-press MMP-3 lab-scale filter-press (Figure 3.4.1). It can be seen from figure 3.4.2 that the most important elements are the beaker, with dehydrating grooves, and the filter disc, where the sludge sample is placed.

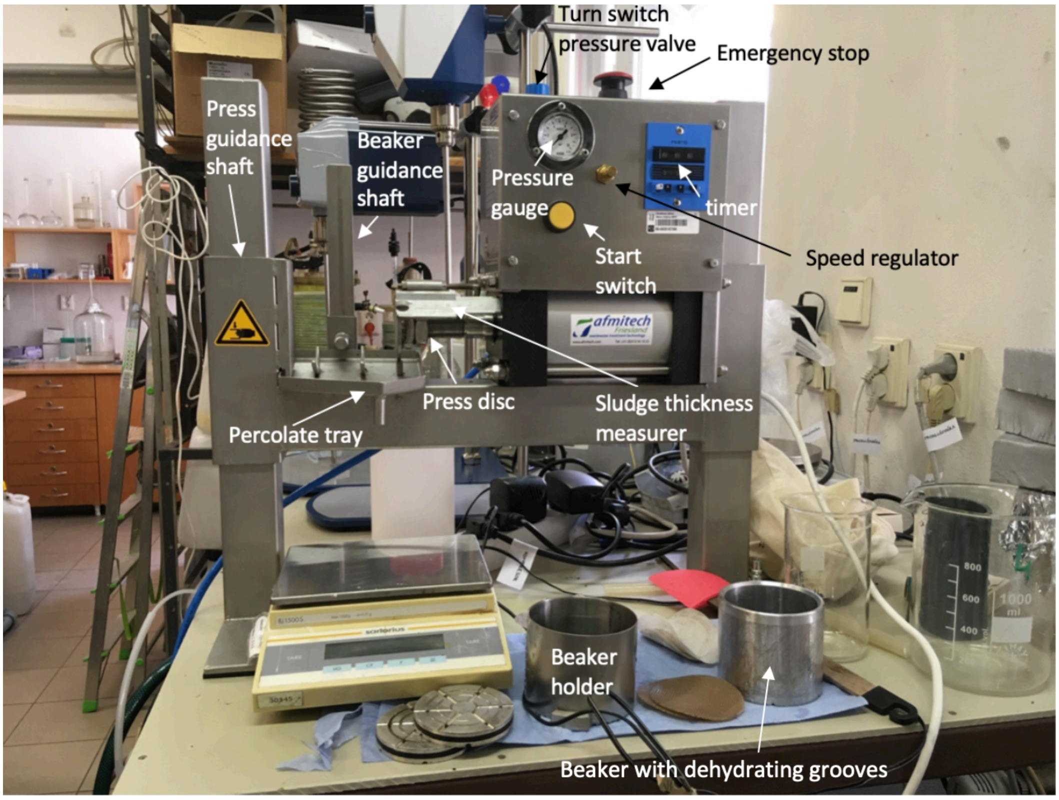

Figure 3.4.1 Elements of the filter press employed during the experimental activity

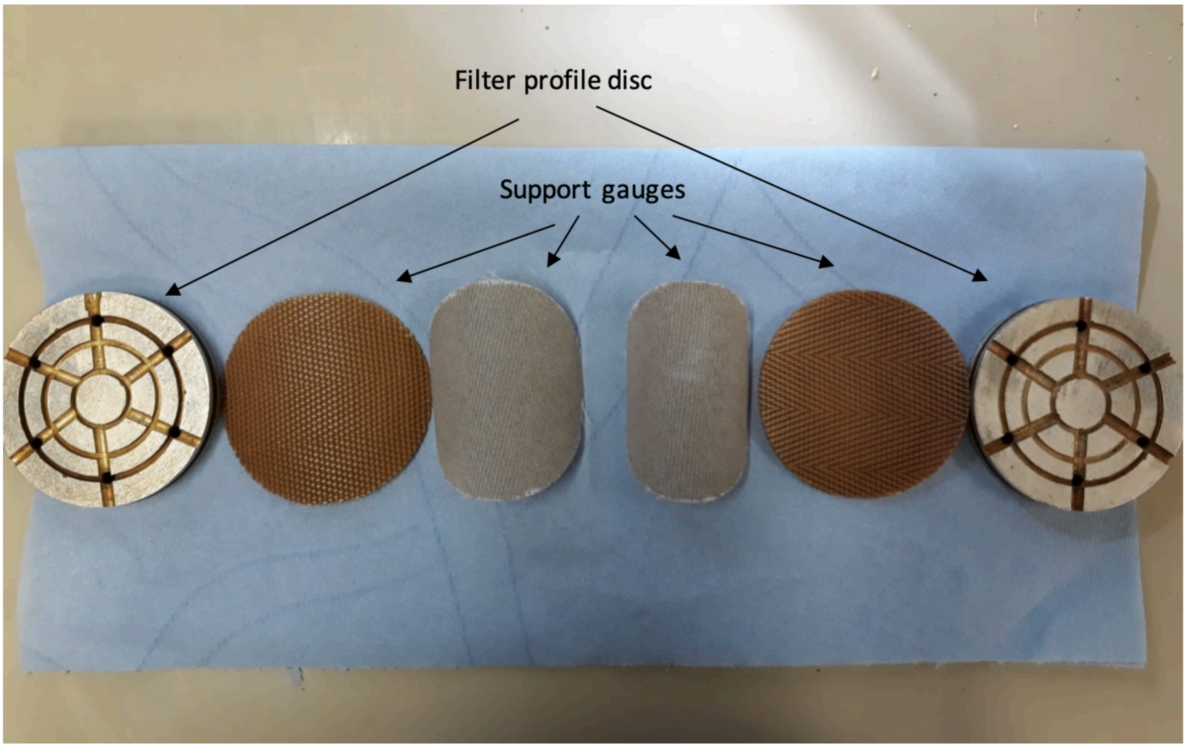

Figure 3.4.2 Elements of the filter press used for dewatering

The control parameters of the filter press are: the dewatering time, which consisted of 999 s for each round; and the pressure, which can be changed but it was always kept at 4 bar. The maximum amount of sludge that can be inserted in the filter-press is 250 ml.

First the sludge sample was placed in the beaker with the dehydrating grooves inside the beaker holder, then the filter profile disc, and all it should be put vertically into the beaker guidance shaft. Afterwards the two different support gauges were be inserted (see figure 3.4.2, firstly the darkest one and then the lightest), then the other two support gauges (white one first and then the dark one) and last the filter profile disc. Finally, the beaker holder can be turned horizontally into the layer and the pressing phase can start.

The aim of these tests was to optimize the dewatering procedure. Thus, the best amount of sludge should be that one that can be dewatered in just one round, which means 999 s. During the tests different parameters have been changed to see the effect on the total solids (TS) values residual in the sludge after the press filtering (see paragraph 3.5):

- the amount of sludge
- the pressure of the machine.

With the change of the amount of sludge it was seen that the amount of TS wasn't changing drastically, the only aspect that was changing was the thickness of the sludge cake. With 150 ml of sludge the thickness of the sludge cake was around 3 mm whereas with the amount of 50 ml it was less then 1 mm.

Also, with the change of the pressure, the number of rounds wasn't changing so it was discovered that increasing the pressure, the minutes necessary to dewater were exactly the same and this parameter wasn't useful to reduce the time of the process.
During the performed tests it was observed that the amount of sludge which was suitable was 50 ml, equivalent to 1.19 g of dry mass.

Another parameter that was evaluated was the dewatering over time (figure 3.4.3), measured by the weight of water that was released from the flocculated sludge sample by the machine; at the beginning of the tests most water was released and then after a while the curve tended to be almost horizontal because almost all the water has already been released. The weight of the dewatered liquor was measured every 120 s. In figure 3.4.3, two different dewatering over time were compared due to the use of two different flocculants: the powdered one and the liquid one named EM440.

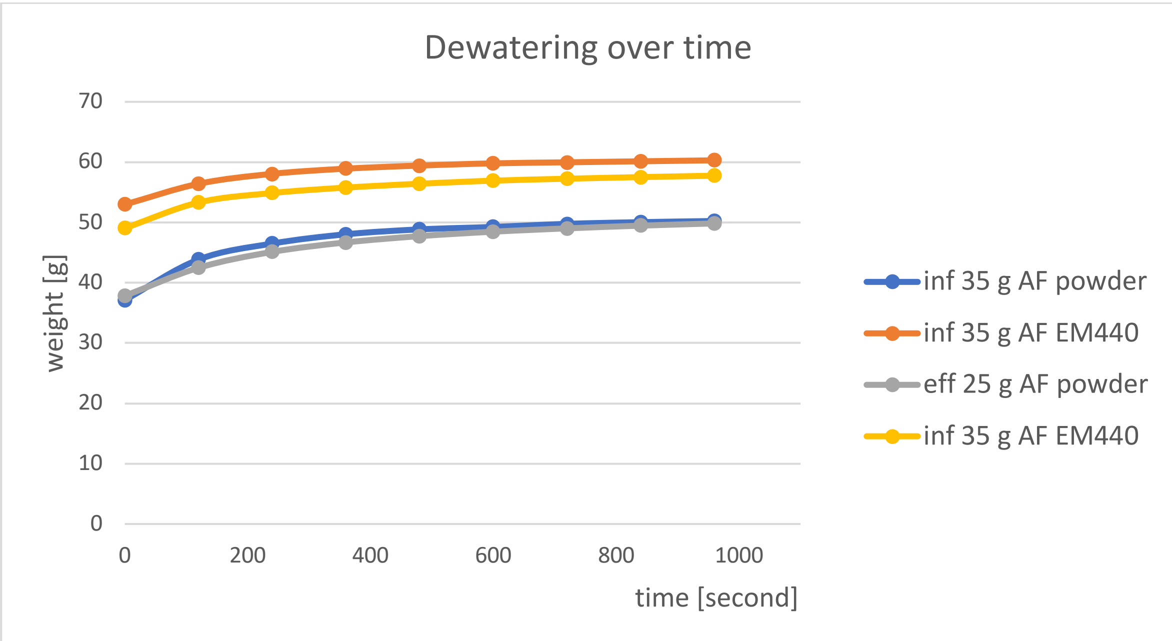

Figure 3.4.3_ Dewatering over time for the sludge using two different types of flocculant

3.5 Analysis of dewatering performances

In order to evaluate the % of TS residual in the sludge after the press filtering, the press filtered sample was weighed when it was still wet (wet cake), then it was dried at 105 °C for 12 h, put into the desiccator for 3 - 4 h and weighed again (dry cake).
Using the following formula, the percentage of the total solids was evaluated:

$$TS = 100 * \frac{w_d}{w_w}$$

Where TS = total solids [%], w_d= dry cake [g] and w_w= wet cake [g].
These values were used to measure the efficiency of the dewatering process; during the last month of experimental activity, the TS residual in the liquor were also calculated.

3.6. Statistical Analysis

The statistical analysis considered only the tests involving post-aeration (influent and effluent) sludge samples and mechanical mixing were performed on five replicates. With the use of the mechanical mixer the flocs were better formed and also the dewatering resulted being better. The other sludge samples exhibited a high variability, as before mentioned, so they were excluded from statistical evaluations.
As we can see in Figure 3.6.1a, five samples were analyzed within the same boundary conditions: 87.5 ml of influent sludge; pressure of the machine 4 bar; 15 [gPEeff/kgTS] of flocculant; 2 rounds of 999 seconds in the filter press. The figure on the left shows the standard deviation of the 5 samples, whereas the figure on the right shows the same values on a larger scale (less detailed) because the range on the y-axis was changed from 0 to 50.
In figure 3.6.1b the same statistical evaluation is shown for the effluent sludge. This time just one round of 999 s was enough to dewater it whereas the other parameters were the same as for the influent.

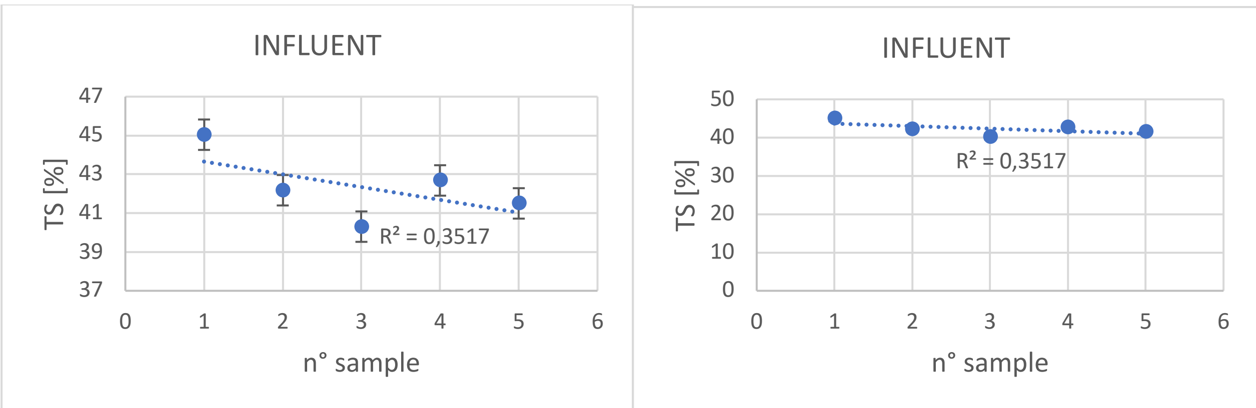

Figure 3.6.1a_ Statistical evaluation of influent sludge using the magnetic mixer

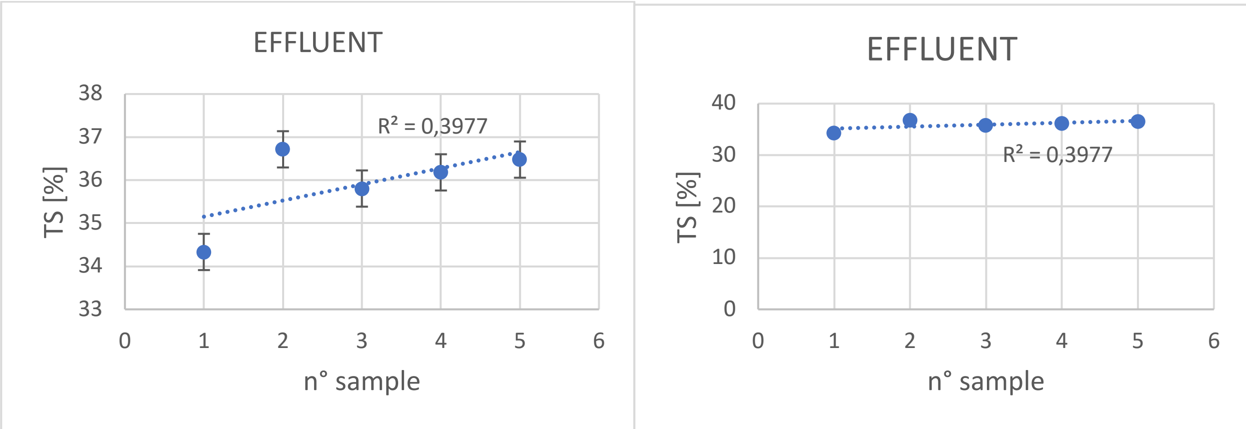

Figure 3.6.1b_ Statistical evaluation of effluent sludge using the magnetic mixer

As it can be seen in figure 3.6.2 a and b the same procedure has been performed also with the mechanical mixer. This time the boundary conditions were slightly different: 50 ml of sludge; pressure of the machine 4 bar; 35 [gPEeff/kgTS] flocculant for the influent and 25 [gPEeff/kgTS] for the effluent; one 999 s round in the filter press to dewater all the samples.

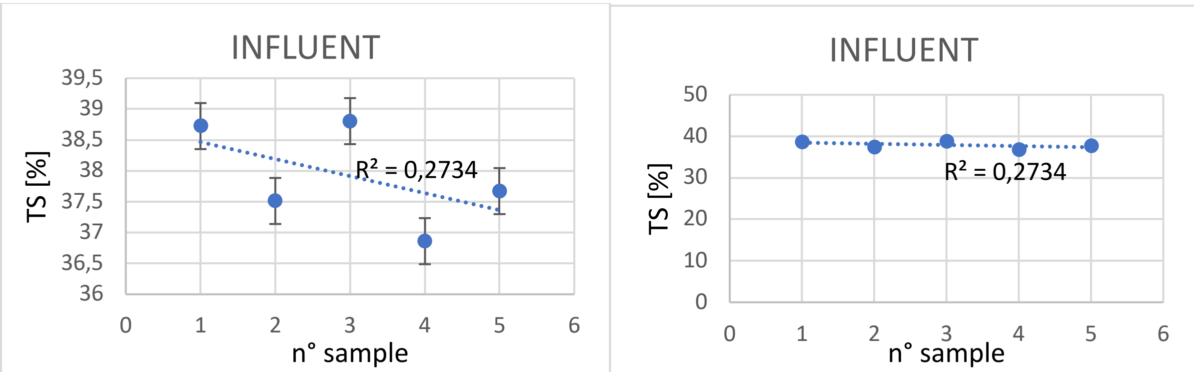

Figure 3.6.2a_ Statistical evaluation of influent sludge using the mechanical mixer

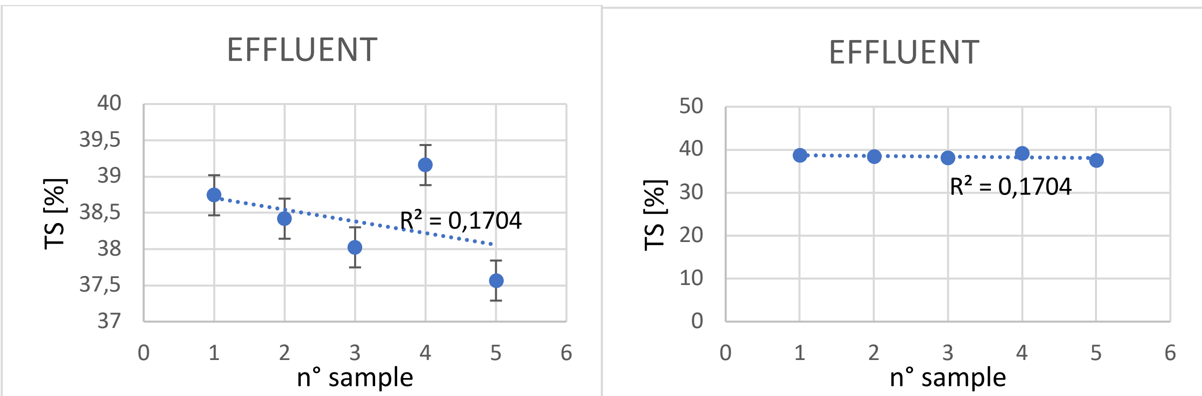

Figure 3.6.2b_ Statistical evaluation of effluent sludge using the mechanical mixer

4. Result and discussion

4.1 Flocculant choice

Our tests investigating the flocculant dose showed that there were four different kind of flocs that could be created within the considered sludge samples (Figure 4.4.1).

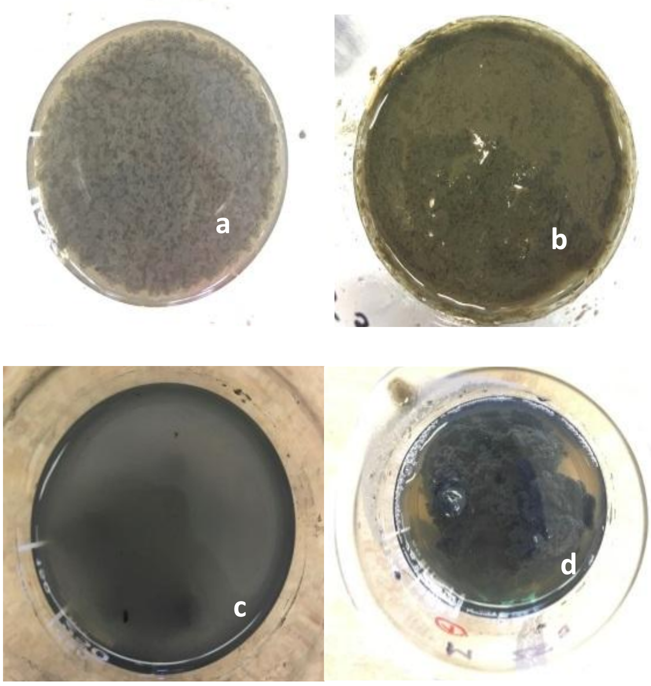

Figure 4.1.1 a) right dose of flocculant b) "spinach" c) overdose d) one big flock

Figure 4.1.1 shows that in the case a, the dose of flocculant was the right one that create small flocs that can be dewatered in the right way. The case b is the "spinach" effect when the mixing is too long, and the flocs are destroyed. The case c is the overdose because of the color of the water and also the water is sticky. The fourth case, d, is the creation of only one big floc using the magnetic mixer that is due to the wrong/insufficient mixing of the flocculant with the sludge.

4.2. Dewatering tests

4.2.1. Sludge from the Post Aeration Pilot Plant

At the beginning we analyzed if different amounts of sludge caused differences in the evaluation of residual TS in the cake. As it can be seen in Figure 4.2.1 there are no big differences between the total solids deriving from 115 ml and 87.5 ml of sludge.

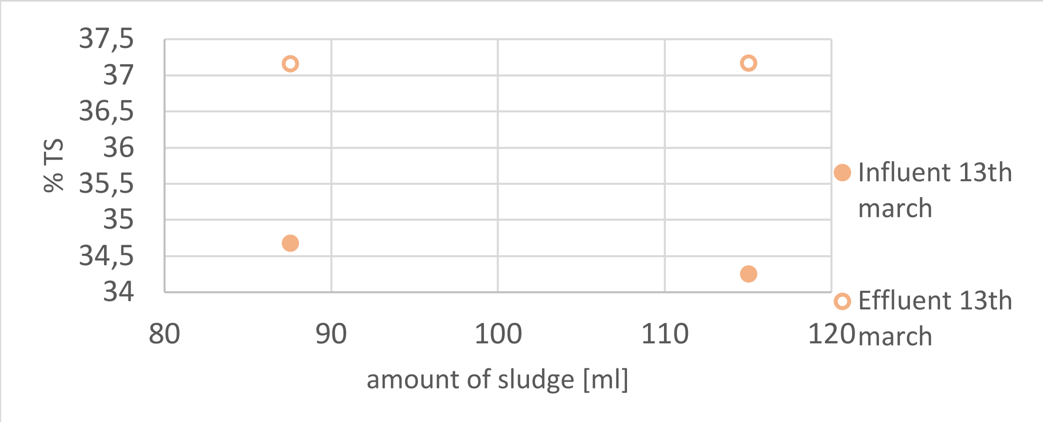

Figure 4.2.1_ Comparison between different amounts of sludge samples from post aeration plant

After this analysis TS residual concentration in the cake was calculated from 87.5 ml of sludge because it had the best quantity (Figure 4.2.1) and also needed less rounds for dewatering and the cake thickness of the sample was around 1-2 mm.
At the beginning of the analysis it was discovered that the value of total solids of the effluent was lower than the influent one as can be seen in the graph 4.2.2, this was probably due to some problems with the aeration in the pilot plant.

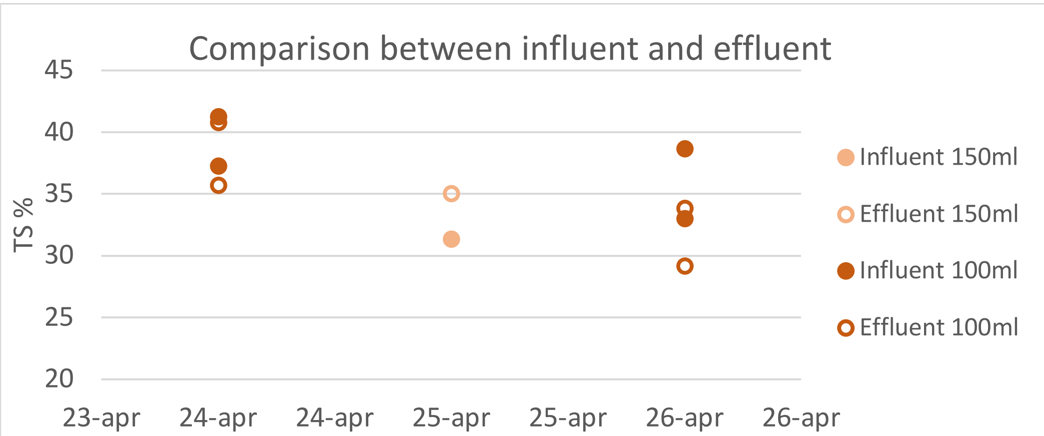

Figure 4.2.2_ comparison between influent and effluent TS obtained from 100-150 ml samples

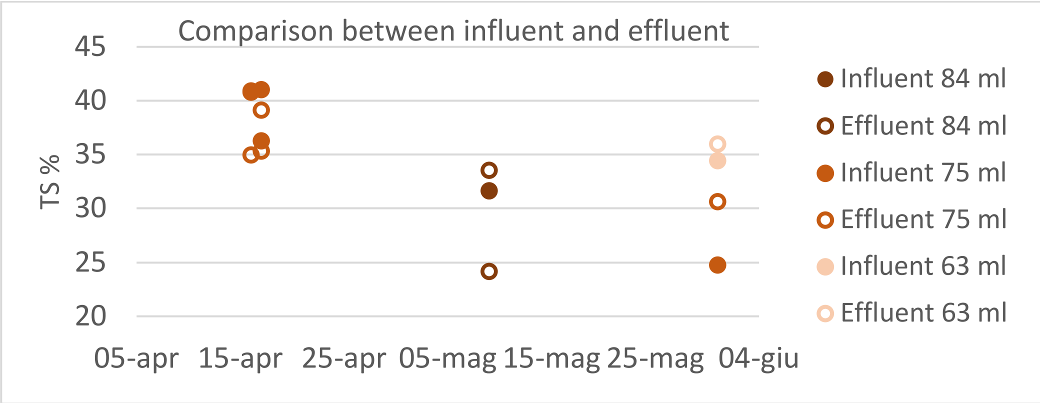

Figure 4.2.3_ Comparison between influent and effluent TS obtained from 84, 75 and 63 ml samples

The amounts of 84, 75 and 63 ml were also tested (figure 4.2.3) and it was discovered that from the end of April the effluent had higher values of TS then influent as expected (Table 4.2.1).

Table 4.2.1_ Values of total solids from the 31st of May

	sample	Dose [gPEeff/kgTS)	%TS
31-may	inf 1	35	24,8
	eff 1	35	**34,5**
	inf 2	25	30,7
	eff 2	25	**36,0**

4.2.2 Sludge from Lab-Scale Anaerobic Digesters

The same procedure described in section 4.2.1. was adopted for the sludge samples deriving from lab-scale digesters (M, I and TPAD2).

At the beginning it was really difficult to mix the flocculant with the sludge. Since the mechanical mixer was used, the mixing was easier and better, and it was possible to dewater the samples and calculate the total solids.

As it can be seen in Figure 4.2.4, at the beginning of the tests the Thermophilic sludge seemed to achieve the highest value of TS residual in the cake, but later it was found that this kind of sludge was the most difficult to treat and to study. It was difficult to find the right dose of flocculant, because every time a different dose was needed and the value of residual TS was also fluctuating.

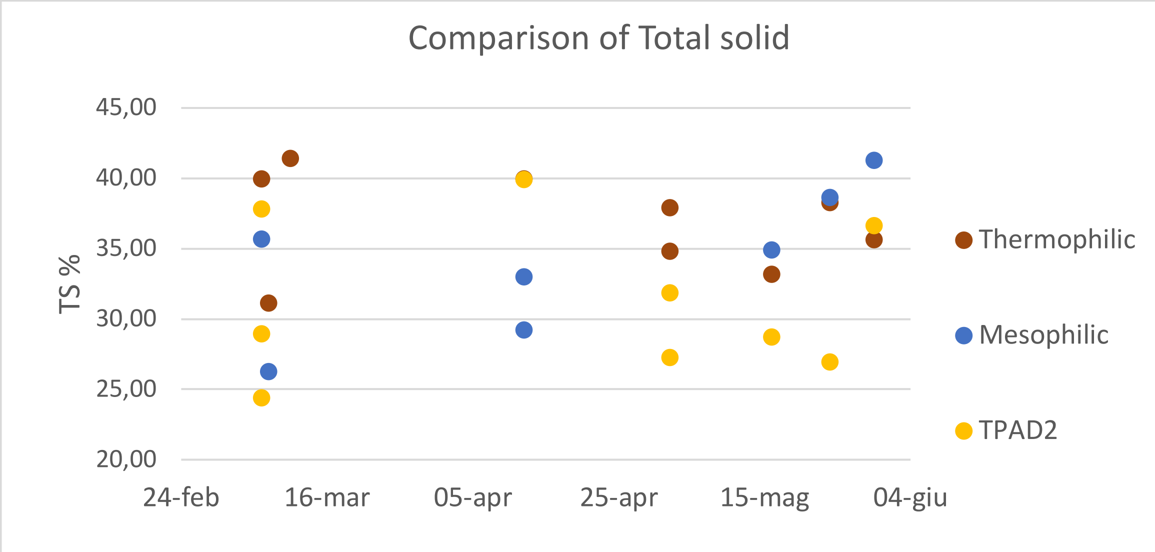

Figure 4.2.4_ Comparison of Total Solids residual in the press filtered cake obtained from lab-scale digesters' sludge samples along the whole sampling period

Using the same dose of flocculant for three weeks in a row and adopting the same mixing, the best performances were exhibited by the Mesophilic sludge (figure 4.1.5).

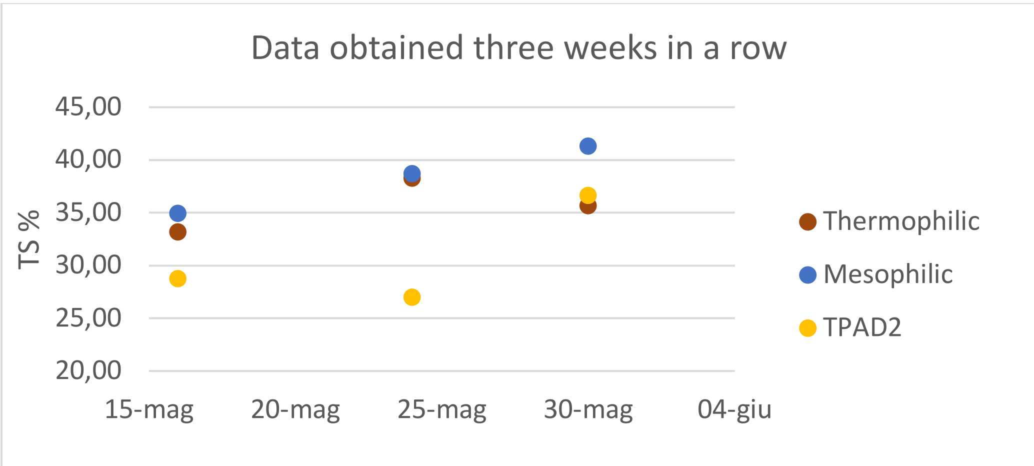

Figure 4.2.5_ Comparison of Total Solids residual in the press filtered cake obtained from lab-scale digesters' sludge samples along the last 3 weeks of sampling period

As can be seen in table 4.2.2, the sludge with the highest value of total solids residual in the press filtered cake was the Mesophilic one, with values between 35 and 41 [%].

Table 4.2.2_ Value of total solids from the last three weeks of analysis in a row

	sample	**Dose [gPEeff/kgTS)**	**%TS**
16-may	TPAD2	40	28,7
	T	50	33,2
	M	35	**34,9**
	TPAD2	40	27,0

24-may	T	50	38,3
	M	35	**38,7**
30-may	TPAD2	40	36,6
	T	50	35,7
	M	35	**41,3**

4.3 Statistical evaluation

Table 4.3.1 shows the average and standard deviation for all the studied samples. The first two sludges, mixed with the magnetic mixer, show a higher standard deviation. It means that the mixing was not efficient. Whereas when the mechanical mixer was used the two standard deviations were lower,and the best results were obtained for the effluent sludge.

Table 4.3.1_ Average and standard deviation for influent and effluent sludges.

		AVERAGE	*STANDARD DEVIATION*
MAGNETIC MIXING	*INFLUENT*	*42,34*	*1,75*
	EFFLUENT	*35,90*	*0,94*
MECHANICAL MIXING	*INFLUENT*	*37,91*	*0,83*
	EFFLUENT	*38,38*	*0,62*

4.4 Technical and economical assessment of the scale up

In this thesis an economical and energetic assessment of the scale up of the considered process chain (Figure 4.4.1) was performed. The aim of the analysis was to investigate the minimum size of the anaerobic digestor (AD) that could sustain through the bioenergy deriving from the produced methane the energy needs of the whole process chain.

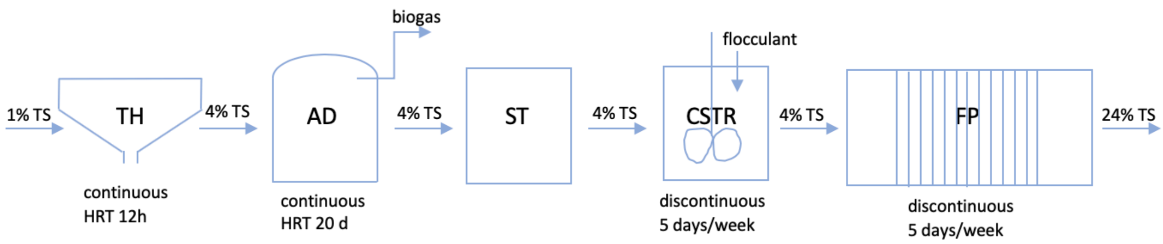

Figure 4.4.1_ scheme of a scaleup system evaluated technically and economically

This process scheme (Fig. 4.4.1) is composed by: a thickener (TH) which allows to improve the TS content in the sludge from 1 % to 4 %; and an anaerobic digester (AD) working in continuous (hydraulic retention time, HRT, equal to sludge retention time, SRT, equal to 20 days) in mesophilic conditions. The dewatering module is made of: a continuously stirred reactor (CSTR), in which the flocculant is added, and a filter press, producing a cake with 24 % TS. Since one cycle of the filter press lasts around 8 hours, it was assumed that the filter press works for 2 cycles per day and 5 days per week. As the thickener and the anaerobic digester are fed in continuous, there is afterwards a storage tank (ST), dimensioned to contain the sludge which is produced during the weekend and the nights by the anaerobic digester, when the dewatering module doesn't work.
The technical and economic assessment of the scale up also considered a belt filter as an alternative to the filter press.

The aim of the assessment was to find the amount of sludge that has to be fed in the digester that can produce enough biogas in order to be energetically self-sufficient and to sustain the energy needs of the dewatering module. Later the amount of sludge was related to the necessary size of the WWTP. For this reason, the first phase of the analysis was to evaluate the energy needs of all the elements of the whole process chain; secondly the energy that the anaerobic digester should produce and finally the size of the corresponding WWTP was calculated. All the elements of the process chain were dimensioned and the costs and energy needs were calculated considering data provided by suppliers of full-scale devices.

First of all it was considered a waste activated sludge entering the system having the characteristics presented in Table 4.4.1 (Elalami et al., 2019).

Table 4.4.1_ Characteristics of WAS considered in the scale up assessment

PARAMETERS	VALUE
TS %	1
VS [% TS]	74

The thickener, able to increase TS from 1 % to 4 % by weight, was dimensioned according to (Metcalf & Eddy, 2014).
The energy consumed by the anaerobic digester working in mesophilic conditions (Carrère et al., 2010) is detailed in Table 4.4.2.

Table 4.4.2_ Energy consumed by anaerobic digester

METHOD	TREATMENT CONDITIONS	ELECTRICAL ENERGY CONSUMED [kWh/ kgVSfed]	THERMAL ENERGY CONSUMED [kWh/ kgVSfed]
Mesophilic	37°C	0,04	0,5

The production of biogas was assumed to be 0,25 [Nm^3/kgVS] and methane content 65 % by volume (Carrère et al., 2010). It was also assumed that 1[Nm^3] of methane can produce 10kWh of primary energy, with a conversion yield of 35 % into electric energy and of 40 % into thermal energy.
For the calculation of the energy consumed by the other elements it was assumed that the power for the thickener was around 1kW and that it worked in continuous for 8760 [h/y]; regarding the CSTR it was considered the power of a mixer from (Grec, 2019) equal to 0,0035 [kW/m^3], which resulted to be 0,00332 [kW] considering the volume of the CSTR equal to 0,95[m^3]. The mixer was assumed to work discontinuously for 2.5 minutes for 2 cycles for 5 days per week. The power for the filter press was calculated from the technical parameters declared in a technical catalogue (SDG Depurazione, 2019) and hypothesizing to use a device having a power equal to 3 kW.
With all these data was subsequently calculated on excel all the energy consumed by the thickener, the anaerobic digester, the CSTR and the filter press and once were obtained all these values, it was made an energy balance: for the thermal energy it was found that the anaerobic digester was already self-sustainable with only 1 kg of VS incoming in the system. Whereas for the electrical energy the following formula (4.4.1) was employed to calculate the amount of sludge feeding the anaerobic digester that is necessary to cover all the energy consumed.

$$En_{el\ p,AD} = En_{el\ c,TH} + En_{el\ c,AD} + En_{el\ c,CSTR} + En_{el\ c,FP} \tag{4.4.1}$$

Where $En_{el,p,AD}$ and $En_{el,c,AD}$ are respectively the electrical energy produced and consumed by the anaerobic digester, while the other three elements are the energy consumed by the other components of the system.

It has to be taken into account that both the electrical energies from the AD are expressed in [kg sludge/kgVSfed] so using the formula (4.4.2) it was calculated the amount of sludge (4 % TS) necessary to feed the anaerobic digester:

$$sludge\ (VS)_{fed} = \frac{En_{el\ c,,TH}+En_{el\ c,CSTR}+En_{el\ c,FP}}{(En_{el\ p,AD}-En_{el\ c,AD})} = 24456{,}7\ \left[\frac{kgVS}{y}\right] \tag{4.4.2}$$

Afterwards it was also calculated the amount of fresh sludge (1 % TS) entering the thickener with equation (4.4.3), and subsequently the size of the wastewater plant, assuming the quantity of fresh sludge produced per year equal to 24,99 kg_{sludge}/yrPE (Eurostat, 2018), was measured with equation (4.4.4).

$$sludge_{in} = \frac{sludge(VS)\ _{fed}}{1\%TS * 74\%VS} = 3304964{,}7\ \left[\frac{kgsludge}{y}\right] \tag{4.4.3}$$

$$plant\ size = \frac{sludge_{in}}{sludge_{produced}} \cong 135000\ \left[\frac{kgsludge}{y}\right] \tag{4.4.4}$$

It was also calculated the size and the capacity necessary for all the elements of the plant in order to evaluate it also from an economical point of view.

The first device studied was the thickener: in order to evaluate its volume, it was considered the sludge incoming with 1 % TS and 74% VS/TS and also a HRT equal to SRT equal to 12 h according to (Metcalf et Eddy, 2014).

With formula (4.4.5) was calculated the amount of sludge feeding the thickener and afterwards the volume of the reactor.

$$amount_{sludge1\%} = \frac{sludge(VS)}{1\ \%(TS) * 74\ \%(VS)} = 377{,}28\ \left[\frac{kg_{sludge}}{h}\right] \tag{4.4.5}$$

Considering the sludge (VS) as the kilograms of volatile solids which feeds the thickener with 1% of TS. Then it was assumed that the density of the sludge was equal to 1000 [kg/m^3] (Metcalf et Eddy, 2014), so the flow rate of the sludge ($Q_{sludge1\%}$) was calculated as 0.377 [m^3/h]. Finally, with formula (4.4.6) we dimension the volume of the thickener:

$$V_{TH} = HRT * Q_{sludge1\%} = 12 * 0.377 \cong 5[m^3] \qquad (4.4.6)$$

Secondly it was dimensioned the anaerobic digester, considering the sludge incoming with 4 % TS and 74 % VS/TS as in equation (4.4.7).

$$amount_{sludge4\%} = \frac{sludge(VS)}{4\ \%(TS) * 74\ \%(VS)} = 2263.67\ \left[\frac{kg_{sludge}}{d}\right] \qquad (4.4.7)$$

The flow rate of the sludge ($Q_{sludge4\%}$) will be equal to about 2.26 [m^3/d], assuming and considering the HRT of anaerobic digester equal to 20 [d] according to (Metcalf & Eddy, 2014). With equation (4.4.8) the working volume (wV_{AD}) necessary for the anaerobic digester and the volume (V_{AD}) with a safety factor equal to 1.2 (4.4.9):

$$wV_{AD} = HRT * Q_{sludge\ 4\%} = 20 * 2.26 = 45.27[m^3] \qquad (4.4.8)$$

$$V_{AD} = wV_{AD} * SF = 54\ [m^3] \qquad (4.4.9)$$

The organic loading rate (OLR) of the digester was verified through equation (4.4.10), knowing from (Metcalf & Eddy) that the OLR for this kind of sludge and for an HRT of 20 [d] is around 1.4.

$$OLR = \frac{\frac{sludge\ VS_{in}}{d}}{V_{AD}} = 1.48\ \left[\frac{kgVS}{d\ m^3}\right] \qquad (4.4.10)$$

Once obtained the volume of the anaerobic digester, we calculated the necessary dimensions of the mixer in order to evaluate the relative costs. It was assumed that the diameter of the digester was around 3.7 m and the height around 5 meters; it was hypothesized to use a top entry mixer, low speed with gearbox with two propellers and a diameter of 800 mm (one used for the surface of the sludge and one at 800 mm from the bottom of the digester). The material used it was supposed to be AISI 304 with mechanical seal. The cost of this machine it is around 5.500 € (table 4.4.3) (Grec, 2019).

In order to calculate the size for the storage tank (ST) and for the filter press (FP) it was taken into account that after the anerobic digestion 50 % of volatile solids were removed (Metcalf & Eddy, 2014), therefore the amount of sludge entering the storage tank and the filter press is lower due to the production of biogas. In the anaerobic digester entered 67.0 [kgVS/d] and only half of it will go out: 33.5 [kgVS/d]. Once it was calculated the amount of dry sludge, it was calculated again the flow of sludge at 4 % TS with formula (4.4.7) obtaining $Q_{sludge\ 4\%}$ = 0,047 [m^3/h]. It was considered for the storage tank that the moment in which the largest amount of sludge can be accumulated was during the weekend while the filter press was not working; depending on this assumption the amount of sludge should be accumulated every weekend for approximately 56h ($t_{storage}$), in order to be able to contain all the incoming flow rate, with formula (4.4.11) can be calculated the volume of the tank that is necessary to contain all the sludge during the weekend.

$$V_{ST} = Q\ sludge_{\ 4\%} * t_{storage} = \ 2.9[m^3] \tag{4.4.11}$$

Considering again a safety factor equal to 1.2 the volume results to be equal to 3,5 [m^3]
For the size of the CSTR it was considered and assumed that the one cycle of the filter press lasts for 8 hours. Considering the accumulation of the sludge after one week and considering also that every week the filter press work for 10 cycles the amount of sludge for each cycle was equal to:

$$Q_{sludge} = \frac{\frac{kg_{sludge}}{h} * 24h * 7days}{10\ cycles} = 792.3\ [kg_{sludge}/cycle] \tag{4.4.12}$$

This way the amount of sludge that enter in the mixer each 8 hours was equal to 792.3 [kg_{sludge}] and so, with 1.2 safety factor, the volume for the CSTR was around 0.95 [m^3]. In order to dimension the filter press the equations (4.4.13, 4.4.14) were used.

$$amount_{sludgeVS} = \frac{amount_{sludge\ 4\%}}{cycle} * 4\ \% = 31.69\ \left[{}^{kg_{sludge\ VS}}\!\big/_{cycle}\right] \tag{4.4.13}$$

Assuming that the sludge coming out from the filter press has 24 % TS it was calculated with the formula (4.4.14) the amount of sludge produced by the filter press:

$$out\ put\ cake\ at\ 24\ \% = \frac{{}^{amount_{sludgeVS}}\!\big/_{cycle}}{0.24} = 132.05\ \left[\frac{kg_{sludge}}{cycle}\right] = \left[\frac{l}{cycle}\right] \tag{4.4.14}$$

Considering the plates, having dimensions of 60x60x2.5 (SDG Depurazione, 2019), the volume was 9000 [cm^3], and each plate can contain 9 [liters/cycle]; for one cycle it will be necessary to have the following number of plates (4.4.15):

$$n°\ plates = \frac{V_{FP}}{V_{plates}} = \frac{132.05\ \left[\frac{l}{cycle}\right]}{9\ \left[\frac{l}{cycle}\right]} \cong 15\ plates \tag{4.4.15}$$

Therefore a filter press with 15 or 20 plates can be considered, having a cost around 25.000-35.000 € (SDG Depurazione, 2019).

Once the energy assessment and the dimensioning of the plant were completed, the following phase was to analyze its costs (table 4.4.3). Considering the whole process chain, the hypothesis is that biogas produced from the anaerobic digester could be able to supply the total energy required by the single devices. In table 4.4.3 are shown the capital and operational costs and the revenues.

For this kind of plant it was supposed that for each 8 hour shift one worker will operate anerobic digester and the filter press; therefore in total, considering two 8-hours shifts, there will 2 people working.

As it can be seen in the table 4.4.3 the revenues obtain from the thermal energy are higher than the costs, so it will be possible to use the surplus for other needs of the plant. Whereas the electrical revenues are just enough to cover the energetic expenditures from all the machines used.

Table 4.4.3_ Economical evaluation of the scale up analyzed

	Machinery's cost	**Effective cost**	
CAPITAL COSTS			
Thickener			
Reactor	2.514,7 [€/m^3]	12.537,5 €	(Chiappero et al. 2019)
Anaerobic digester			
Reactor	2.514,7 [€/m^3]	136.619,1 €	(Chiappero et al. 2019)
Stirrer		5.500 €	(grec,2019)
Storage tank			
Reactor	2.514,7[€/m^3]	8.801,45 €	(Chiappero et al. 2019)
CSTR			
Reactor	2.514,7 [€/m^3]	2.390,83 €	(Chiappero et al. 2019)
Agitator		700€	(grec,2019)
Flocculant	3[€/kg]	3.470,21 €	(assumed from Czech agency)
Filter press			
Filter Press	25.000-35.000 [€]	30.000 €	(sgddepurazioni, 2019)
TOTAL COST		**0,19 M€**	
OPERATIONAL COST			

Thermal energy	0,2 [€/kWh]	2.445,67 €	(Chiappero et al. 2019)
Electric energy	0,2 [€/kWh]	3.060,15 €	(Chiappero et al. 2019)
Digestate disposal	0,55[€/kg]	9.016,52 €	(Arpa, 2019)
Labour	45.000 [€/person]	90.000 €	(Chiappero et al. 2019)
TOTAL COST		**104.522,34€**	
REVENUES			
Thermal energy	0,2 [€/kWh]	3.179,37 €	
Electrical energy	0,2 [€/kWh]	3.060,15 €	(Demichelis et al., 2018)
TOTAL		**6.239,52 €**	

It was also made a comparison between the use of a filter-press and a belt-filter; in table 4.4.4 are summarized the different costs and the different uses of power in order to evaluate which one could be the best option.

Table 4.4.4_ Comparison of cost and power of the filter and belt press

MACHINE	**POWER**	**USE**	**COSTS**	**DRYNESS OUT**
FILTER PRESS	3 kW	Discontinuous	25.000-35.000 €	30 %
BELT PRESS	3,5-4,5kW	continuous	35.000-45.000€	18-20 %

The filter press has lower cost, lower use of power and a higher dryness of the sludge obtained, but the only problem is that it can work only discontinuously with a cycle of 8 hours; whereas the belt filter can work continuously, but it costs more than the filter press and it has also a higher power and the sludge going out will have a lower dryness.

5. Conclusions

Current literature show that an optimized operation of activated sludge treatment in terms of Total Nitrogen removal and use of chemicals are prerequisites for low carbon footprints of WWTPs. The carbon footprint analyses also revealed that GHG emissions from municipal WWTPs have a small impact at global scale, corresponding in average to 0.45 of the yearly average pro capita CO_2e emission in Europe (Parravicini et al., 2016). Moreover, increasing total solids content in the digestate, it was demonstrated that its volatile solids content and GHG emission potential increase and hence the removal of solids decreased in dry anaerobic digestion. Another discover was that the raw digestate contributes to the highest GHG emissions and should not be landfilled. The storage time of the digestate should be reduced to the minimum necessary in order to reduce and avoid nutrients and carbon losses, because after only two months it emits 10 % of the total GHG potential. Literature also reveals that land application of digestate allows to reduce GHG emissions, but C/N ratio and time of application to land are necessary to be studied and analyzed (Zeshan & Visvanathan, 2014).

Recent studies indicate that the dewatering performance for WAS still needs to be improved because it results in high-costs of sludge treatment. The dewatering process still presents critical issues that need to be studied and improved such as the large amount of EPS, the high energy required for the drainage of bound water, the strong compressibility of the sludge cake and the highly complex structure of the sludge. A good way to improve the treatment of the sludge and its dewatering capacity, is to optimize the conditioning phase to achieve high dewatering performance, low cost and environmental impacts. The dewatering process should be studied in details based on both the structural characteristics of sludge and on those of coagulants/flocculants. An effective method could also be the combination of conditioning with other pretreatments (Wei et al., 2018). In addition, pre-treatment methods could have a high influence on dewatering performances because they can improve and reduce the moisture content of sludge (Rao et al., 2019).

The optimization of flocculation is still difficult, but the use of the drainage index, easy and quick to measure, could be useful to set the conditions necessary for the flocculation prior to a gravity drainage (Olivier et al., 2018). Moreover, among all the different conditioning methods that have been studied the use of biomass ash alone or in conjunction with the chemical conditioners could be considered an ideal and optimal option in waste management (Wójcik & Stachowicz, 2019). Temperature and solids concentration can affect conditioning and flocculation processes of the digested sludge; moreover, using the optimum temperature condition for the dewatering and varying the polymer dose as a function of total solid concentration, polymer consumption can be decreases (Yeneneh et al., 2016).

It has been demonstrated that the decomposition of the flocs can be affected by rheological behavior of the sludge, and also the level of organic matters released can be improved or deteriorated (Miryahyaei et al., 2019).

As it can also be shown from the experimental activity performed at VSCHT University in Prague, the most difficult part of the study was the choice of the conditioner dose and also the way to mix it with the sludge. In fact, if mixing is not efficient also dewatering will not be successful. Another characteristic that during the tests turned out to be very fluctuating is the dry substance amount of sludge samples, especially considering those deriving from the lab-scale anaerobic digesters in the university, which were changing almost every week. With this fluctuation it resulted very difficult to optimize the dose of flocculant.
Furthermore, dewatering is a critical step in preparing sludge for further drying, because the basic particle structure of a sludge is a system of biological polymers linked together which form a floc matrix in which is present water in their surface or inside their matrix (Novak, 2006). It was found that dewaterability and rheological behavior of digestate from a WWTP depend on different conditions such as polymer dose, total solid concentration and temperature. Especially the organic fraction of the total solid content, containing polymeric substances such as protein and polysaccharides, affects the variation of the rheological property of digested sludge (Yeneneh et al., 2016). A good way to improve and enhance the dewaterability should be the thermal hydrolysis as a pretreatment during anaerobic digestion. But the structural strength of sludge after the pretreatment resulted to be weaker and was broken during anaerobic digestion; this could explain relationships between the improved dewaterability and the loose of sludge structure (Zhang et al., 2018).
Another important index that can help to assess sludge dewaterability is the “dryness limit”, which is obtained after a lab scale filtration-compression test and it refers to the dryness obtained after an infinite time; which means that in order to extrapolate data from the experiment is necessary the use of a mathematical model. This index depends on final cake dryness, expression pressure and compression time. It was found that in order to achieve tests in a reasonable time (4 hours) it should be used a pressure of 4 [bar] and a cake thickness of 5 [mm] (Ginisty et al., 2014).
From the economic and energetic evaluation of the scale up of the overall sludge management system it was found that the size of the wastewater treatment plant necessary to feed the sludge line and achieve energy self-sufficiency was around 125,000 PE. The total cost of the overall sludge management line was calculated as around 0,19 M€ for the capital costs and 98.282,8 €/y for the operational costs. A filter press or a belt filter could be foreseen depending on the final destination of the sludge. From an economical point and technical perspective, in order to have a higher dryness for the sludge coming out, the best option is the filter press.

6. Acknowledgment

Firstly, I would like to thank the professor Jenicek who accepted me in his team and gave me the opportunity to carry out my master thesis with them in their department. Also, a big thank you to my PhD student supervisor Marie, for her support, she helped me every time that I needed. Also for all the nice moments that we spent together in the laboratory waiting for the samples. Thanks also to the rest of the department group which help me to feel part of the group organizing activities and events all together.

Thanks to all my Erasmus friends, with whom I shared incredible moments in these 9 months knowing new cultures and traditions. Especially I would like to thank my roommates, Jessi, Maite, Sheila, Marica, and also Daniela that were and are still always present when I really need them, in good and bad times, always supporting me and helping me a lot and living together all the good experience that we had in Prague.

Un grande ringraziamento anche alla professoressa Silvia Fiore che mi ha seguito con competenza e pazienza, supportandomi nei momenti di difficoltà con i suoi preziosi consigli, osservazioni e correzioni.
Ringrazio l'Ing. Marco Mariano e Martini che mi hanno fornito le indicazioni necessarie per ultimare l'analisi energetica ed economica dell'impianto.

Vorrei ringraziare i miei genitori che hanno reso tutto questo possibile, nonostante alcuni imprevisti avuti nel corso dell'anno, stando sempre al mio fianco anche a chilometri di distanza. Li ringrazio anche per aver sopportato, al mio rientro, gli alti e bassi di umore dovuti allo stress per il ritorno e per la stesura della tesi.

Vorrei inoltre ringraziare mia sorella Gloria che, fin da quando ero piccola, è stata per me un punto di riferimento da seguire, riuscendo ad intuire quando avevo bisogno dei suoi consigli e del suo aiuto. Un grazie anche a mio fratello Gabriele che mi è stato vicino e d'aiuto con il suo interessamento e la sua capacità di sdrammatizzare quando ero giù di morale o affaticata.

7. References

1. Arpa (2019). Agenzia Regionale Per la Protezione Dell'Ambiente, Italian http://www. arpa.piemonte.gov.it/approfondimenti/temi-ambientali/rifiuti.
2. Åkerberg, C., & Zacchi, G. (2000). An economic evaluation of the fermentative production of lactic acid from wheat flour. *Bioresource Technology*, *75*(2), 119–126. https://doi.org/10.1016/S0960-8524(00)00057-2
3. Bonilla, S., & Allen, D. G. (2018). Cationic proteins for enhancing biosludge dewaterability: A comparative assessment of surface and conditioning characteristics of synthetic polymers, surfactants and proteins. *Separation and Purification Technology*, *191*(January 2017), 200–207. https://doi.org/10.1016/j.seppur.2017.08.048
4. Braguglia, C. M., Carozza, N., Gagliano, M. C., Gallipoli, A., Gianico, A., Rossetti, S., … Mininni, G. (2014). Advanced anaerobic processes to enhance waste activated sludge stabilization. *Water Science and Technology*, *69*(8), 1728–1734. https://doi.org/10.2166/wst.2014.067
5. Braguglia, C. M., Coors, A., Gallipoli, A., Gianico, A., Guillon, E., Kunkel, U., … Mininni, G. (2015). Quality assessment of digested sludges produced by advanced stabilization processes. *Environmental Science and Pollution Research*, *22*(10), 7216–7235. https://doi.org/10.1007/s11356-014-3090-6
6. Carrère, H., Dumas, C., Battimelli, A., Batstone, D. J., Delgenès, J. P., Steyer, J. P., & Ferrer, I. (2010). Pretreatment methods to improve sludge anaerobic degradability: A review. *Journal of Hazardous Materials*, *183*(1–3), 1–15. https://doi.org/10.1016/j.jhazmat.2010.06.129
7. Chiappero, M., Demichelis, F., Lin, X., Liu, C., Frigon, D., Fiore, S.(2019). Investigation of pre-treatments improving low-temperature anaerobic digestion of waste activated sludge. *Process Safety and Environmental Protection, 131, 28-37. https://doi.org/10.1016/j.psep.2019.08.034.*
8. *Demichelis, F., Fiore, S., Onofrio, M. (2018).* Pre-treatments aimed at increasing the biodegradability of cosmetic industrial waste. *Process Safety and Environmental Protection, 118, 245-253. https://doi.org/10.1016/j.psep.2018.07.001.*
9. Diak, J., Örmeci, B., & Kennedy, K. J. (2013). Effect of micro-aeration on anaerobic digestion of primary sludge under septic tank conditions. *Bioprocess and Biosystems Engineering*, *36*(4), 417–424. https://doi.org/10.1007/s00449-012-0798-x
10. Elalami, D., Carrere, H., Monlau, F., Abdelouahdi, K., Oukarroum, A., & Barakat, A. (2019). Pretreatment and co-digestion of wastewater sludge for biogas production: Recent research advances and trends. *Renewable and Sustainable Energy Reviews*, *114*(August), 109287. https://doi.org/10.1016/j.rser.2019.109287
11. Eurostat (2019) URL:http://ec.europa.eu/eurostat/tgm/table.do?tab=table&plugin= 1&language =en&pcode=ten00030 (accessed 5.21.18) 2018. Sewage Sludge Production and Disposal from Urban Wastewater (in Dry Substance (d.s)) [WWW Document].
12. Fang, Y. R., Li, S., Zhang, Y., & Xie, G. H. (2019). Spatio-temporal distribution of sewage sludge, its methane production potential, and a greenhouse gas emissions analysis. *Journal of Cleaner Production*, 117895. https://doi.org/10.1016/j.jclepro.2019.117895
13. Ginisty, P., Girault, R., Tosoni, J., Dieudé-Fauvel, E., Olivier, J., & Vaxelaire, J. (2014). Sludge Dewatering: Influence of operating parameters on dryness limit determination. *European Conference on Sludge Management*, (May 2014), 1–8. Retrieved from https://hal.archives-ouvertes.fr/hal-01294456
14. Grec (2019) URL: https://www.grec.it/it/prodotti/agitatori-verticali
15. Guilayn, F., Jimenez, J., Rouez, M., Crest, M., Patureau, D., (2019). Digestate mechanical separation: Efficiency profiles based on anaerobic digestion feedstock and equipment choice.*Bioresource Technology*, *274,180-189.*
16. Kacprzak, M., Neczaj, E., Fijałkowski, K., Grobelak, A., Grosser, A., Worwag, M., … Singh, B. R.

(2017). Sewage sludge disposal strategies for sustainable development. *Environmental Research*, *156*(January), 39–46. https://doi.org/10.1016/j.envres.2017.03.010
17. Kevbrina, M. V., Nikolaev, Y. A., Danilovich, D. A., & Vanyushina, A. Y. (2011). Aerobic biological treatment of thermophilically digested sludge. *Water Science and Technology*, *63*(10), 2340–2345. https://doi.org/10.2166/wst.2011.492
18. Liu, R., Yu, X., Yu, P., Guo, X., Zhang, B., & Xiao, B. (2019). New insights into the effect of thermal treatment on sludge dewaterability. *Science of the Total Environment*, *656*, 1082–1090. https://doi.org/10.1016/j.scitotenv.2018.11.436
19. Lü, F., Zhou, Q., Wu, D., Wang, T., Shao, L., & He, P. (2015). Dewaterability of anaerobic digestate from food waste: Relationship with extracellular polymeric substances. *Chemical Engineering Journal*, *262*, 932–938. https://doi.org/10.1016/j.cej.2014.10.051
20. Masuda, S., Sano, I., Hojo, T., Li, Y. Y., & Nishimura, O. (2018). The comparison of greenhouse gas emissions in sewage treatment plants with different treatment processes. *Chemosphere*, *193*, 581–590. https://doi.org/10.1016/j.chemosphere.2017.11.018
21. Metcalf & Eddy, Wastewater Engineering: Treatment and Resource Recovery, fifth edition (2014)
22. Miryahyaei, S., Olinga, K., Abdul Muthalib, F. A., Das, T., Ab Aziz, M. S., Othman, M., … Eshtiaghi, N. (2019). Impact of rheological properties of substrate on anaerobic digestion and digestate dewaterability: New insights through rheological and physico-chemical interaction. *Water Research*, *150*, 56–67. https://doi.org/10.1016/j.watres.2018.11.049
23. Morras, M., Dosta, J., & García-Heras, J. L. (2015). Aerobic/anoxic post-treatment of anaerobically digested sewage sludge as an alternative to biological nitrogen removal from reject water. *Bioprocess and Biosystems Engineering*, *38*(5), 823–831. https://doi.org/10.1007/s00449-014-1325-z
24. Ning, X. an, Luo, H., Liang, X., Lin, M., & Liang, X. (2013). Effects of tannery sludge incineration slag pretreatment on sludge dewaterability. *Chemical Engineering Journal*, *221*, 1–7. https://doi.org/10.1016/j.cej.2013.01.106
25. Novak, J. T. (2006). Dewatering of sewage sludge. *Drying Technology*, *24*(10), 1257–1262. https://doi.org/10.1080/07373930600840419
26. Novak, J. T. (2010). The role of organic colloids in dewatering. *Drying Technology*, *28*(7), 871–876. https://doi.org/10.1080/07373937.2010.490499
27. Olivier, J., Ginisty, P., & Vaxelaire, J. (2018). Sludge conditioning prior to dewatering: Introducing drainage index as a new parameter. *Environmental Technology and Innovation*, *11*, 286–298. https://doi.org/10.1016/j.eti.2018.06.010
28. Panepinto, D., Fiore, S., Genon, G., & Acri, M. (2016). Thermal valorization of sewer sludge: Perspectives for large wastewater treatment plants. *Journal of Cleaner Production*, *137*(2016), 1323–1329. https://doi.org/10.1016/j.jclepro.2016.08.014
29. Parravicini, V., Svardal, K., & Kroiss, H. (2008). Post-aeration of anaerobically digested sewage sludge for advanced COD and nitrogen removal: Results and cost-benefit analysis at large-scale. *Water Science and Technology*, *57*(7), 1087–1094. https://doi.org/10.2166/wst.2008.211
30. Parravicini, Vanessa, Svardal, K., & Krampe, J. (2016). Greenhouse Gas Emissions from Wastewater Treatment Plants. *Energy Procedia*, *97*, 246–253. https://doi.org/10.1016/j.egypro.2016.10.067
31. Rao, B., Zhu, Y., Yu, M., Lu, X., Wan, Y., Huang, G., … Liu, X. (2019). High-dry dewatering of sludge based on different pretreatment conditions. *Process Safety and Environmental Protection*, *122*, 288–297. https://doi.org/10.1016/j.psep.2018.12.018
32. Sawalha, O., & Scholz, M. (2010). Modeling the Relationship between Capillary Suction Time

and Specific Resistance to Filtration. *Journal of Environmental Engineering*, *136*(9), 983–991. https://doi.org/10.1061/(asce)ee.1943-7870.0000223
33. Sawalha, O., & Scholz, M. (2012). Impact of temperature on sludge dewatering properties assessed by the capillary suction time. *Industrial and Engineering Chemistry Research*, *51*(6), 2782–2788. https://doi.org/10.1021/ie202381r
34. Sdgdepurazioni (2019) URL: https://www.sdgdepurazione.com/it/trattamento-fanghi
35. SHAO, L., HE, P., YU, G., & HE, P. (2009). Effect of proteins, polysaccharides, and particle sizes on sludge dewaterability. *Journal of Environmental Sciences*, *21*(1), 83–88. https://doi.org/10.1016/S1001-0742(09)60015-2
36. Skinner, S. J., Studer, L. J., Dixon, D. R., Hillis, P., Rees, C. A., Wall, R. C., … Scales, P. J. (2015). Quantification of wastewater sludge dewatering. *Water Research*, *82*, 2–13. https://doi.org/10.1016/j.watres.2015.04.045
37. Tomei, M. C., Carozza, N. A., & Mosca Angelucci, D. (2016). Post-aerobic digestion of waste sludge: performance analysis and modelling of nitrogen fate under alternating aeration. *International Journal of Environmental Science and Technology*, *13*(1), 21–30. https://doi.org/10.1007/s13762-015-0839-5
38. Tomei, M. Concetta, & Carozza, N. A. (2015). Sequential anaerobic/anaerobic digestion for enhanced sludge stabilization: comparison of the process performance for mixed and waste sludge. *Environmental Science and Pollution Research*, *22*(10), 7271–7279. https://doi.org/10.1007/s11356-014-3130-2
39. Tosoni, J., Baudez, J. C., & Girault, R. (2015). Effect of Operating Parameters on the Dewatering Performance of Press Filters: A Sensitivity Analysis. *Drying Technology*, *33*(11), 1327–1338. https://doi.org/10.1080/07373937.2015.1043013
40. Vaxelaire, J., & Cézac, P. (2004). Moisture distribution in activated sludges: A review. *Water Research*, *38*(9), 2215–2230. https://doi.org/10.1016/j.watres.2004.02.021
41. Vojtiskova, M., Marco, M. Di, Satkova, B., & Jenicek, P. (n.d.). *Postaeration of the digested sludge and its potential benefits*.
42. Wei, H., Gao, B., Ren, J., Li, A., & Yang, H. (2018). Coagulation/flocculation in dewatering of sludge: A review. *Water Research*, *143*(2015), 608–631. https://doi.org/10.1016/j.watres.2018.07.029
43. Wójcik, M., & Stachowicz, F. (2019). Influence of physical, chemical and dual sewage sludge conditioning methods on the dewatering efficiency. *Powder Technology*, *344*, 96–102. https://doi.org/10.1016/j.powtec.2018.12.001
44. Yeneneh, A. M., Hong, E., Sen, T. K., Kayaalp, A., & Ang, H. M. (2016). Effects of Temperature, Polymer Dose, and Solid Concentration on the Rheological Characteristics and Dewaterability of Digested Sludge of Wastewater Treatment Plant (WWTP). *Water, Air, and Soil Pollution*, *227*(4). https://doi.org/10.1007/s11270-016-2820-4
45. Zeshan, & Visvanathan, C. (2014). Evaluation of anaerobic digestate for greenhouse gas emissions at various stages of its management. *International Biodeterioration and Biodegradation*, *95*(PA), 167–175. https://doi.org/10.1016/j.ibiod.2014.06.020
46. Zhang, J., Li, N., Dai, X., Tao, W., Jenkinson, I. R., & Li, Z. (2018). Enhanced dewaterability of sludge during anaerobic digestion with thermal hydrolysis pretreatment: New insights through structure evolution. *Water Research*, *131*, 177–185. https://doi.org/10.1016/j.watres.2017.12.042