Impact of cold and dilute sewage on pre-fermentation – a case study

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Abstract The municipal sewage treatment plant (STP) of the city of Ghent (Belgium) has to be retrofitted to a 43%-increase in the nitrogen treatment capacity and to phosphorus removal. Cold weather, dilute sewage and a critical COD over N ratio make the retrofit a challenge for full biological nutrient removal. The potential for fermentation of primary sludge to alter those critical feed sewage characteristics was experimentally evaluated. The idea was that the pinpoint introduction of fermentate could optimise the available reactors by achieving high-rate denitrification and enhanced biological phosphorus removal.

The fermentation process was evaluated with a bench scale apparatus. At 20°C (heated process), the hydrolysis yield – expressed in terms of soluble COD – varied from 11% to 24% of the total sludge COD. The fermentation yield expressed in VFA COD varied from 8% to 13% of the total sludge COD. The efficiency of heated fermentation of primary sludge was lower during cold and wet weather, due to the different sewage characteristics, as a result of extended dilution periods and low temperature.

The raw sewage, the primary effluent and the fermentate were fractionated according to the requirements for the IAWQ Activated Sludge Model No. 2d. The results clearly show that fermentation in the sewer played an important role and temperature was the driving parameter for the characteristics of the dissolved COD. Instead, the weather flow conditions were the driving parameter for the characteristics of the suspended COD.

The results of the detailed fractionation were used as background for process evaluation. The final scenario choice for the retrofit depends on a cost-efficiency calculation.
Keywords Pre-fermentation; cold weather; dilute sewage; primary sludge; influent characterisation

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Introduction

In Flanders (Belgium), all municipal sewage treatment plants (STPs) >10,000 p.e. are being upgraded to nutrient removal (nitrogen and phosphorus) as part of an on-going programme to implement the EU Urban Wastewater Treatment Directive 91/271 and the OSPAR Convention to protect the North Sea. The implementation of biological nutrient removal (BNR) systems is challenged by several factors. Key points of technical and scientific interest are – among others – the rather diluted wastewater, the often unfavourable influent composition (low BOD/N and BOD/P), the highly fluctuating hydraulic and organic loads due to the nature of the sewer system (combined collection system) and the requirement to treat up to 10 dry weather flows (Q_{DWF}).

All this makes the optimisation of the use of the sewage carbon source an important issue. Pre-fermentation of primary sludge could help in changing the unfavourable fingerprint of the wastewater. Short-chain fatty acids (VFA) may be formed at the plant, which in turn can speed up reaction rates for denitrification and phosphate release (Moser-Engeler *et al.*., 1999). High-rate denitrification can increase the aerobic SRT, while not entailing enlargement of the biological reactors. High-rate phosphate release can reduce the size of the anaerobic compartment for inducing enhanced biological phosphorus removal (EBPR) to 5% of the total biological reactors size (Barnard, 1994).

However, literature shows varying figures on the performance of the fermentation process. Munch and Koch (1999) show that this is not likely to be related to the type of fermenter. Therefore, it must be related to its operation and to the nature of the incoming sewage. In flat, large and mild-temperate catchments, anaerobic transformations of the sewage composition can already occur in the collectors and reduce the VFA content at the STP (e.g.: Hvitved-Jacobsen *et al.*, 1995). The inventory of Munch and Koch (1999) confirms that in regions with cold weather, the fermentation process at the STP is more efficient than in mild-temperate regions.

While in colder climates the recipe (1) unheated fermenter and (2) combined sewage are unfavourable for increasing the VFA production (Wedi,1992), Bundgaard *et al.* (1992) clarified the necessity of the heated process. Doubts remain about the efficiency of processing combined sewage (Danesh and Oleszciewicz, 1997).

The aim of this study is the evaluation of heated fermentation of cold and dilute sewage. Batch tests were performed under varying weather conditions. The results contribute to an evaluation of the fermentation as an option to retrofit a STP to full biological nutrient removal with minimal changes in the plant operation. A practical reference for fermentation of low strength wastewater is developed.

Methods and material

Full-scale plant

The Ghent STP has a nominal load of 175,000 population equivalents (PE) at a sludge loading rate of 0.1 kgBOD.kg^{-1} MLSS. The biological treatment is achieved by a single-stage nitrifying activated sludge system designed in a plug-flow way (Figure 1).

The plug flow reactors (27,000 m^3) are operated by intermittent aeration. A mixed liquor channel assures post-denitrification (6,500 m^3) followed by a post-aeration step (6,000 m^3). The clarification unit (10,828 m^2) conveys the thickened sludge to an open recirculation channel (7,900 m^3).

The STP has to be retrofitted to full nutrient removal and to a 43%-increase in capacity (250,000 p.e.). On top of this, exceptionally intense first flush phenomena are experienced on a regular basis at the plant. This urged us to consider primary clarifiers as one possible option for the renovation of the STP. An anaerobic digester will be adopted in the future. This unit might optimise the use of the primary sludge for energy recovery and assure all the required steam for heating the pre-fermenters from the exhaust gas from the combined heat power (CHP).

Measuring campaign

Under a prolonged wet weather flow (WWF) period, influent daily samples were collected for little less than two sludge ages (30 measuring points). The samples were analysed for COD, COD_F, BOD_5 and BOD_{21}, SS, VSS, K_jN, NH_4–N, and TP. Alongside the daily

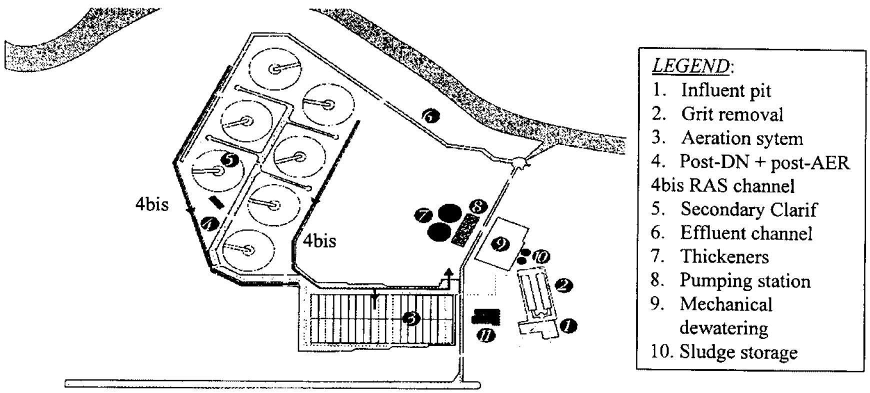

Figure 1 Plant layout of the STP of the city of Ghent

samples collection, a 100 l side stream respirometer (Minworth Ltd.) was placed at the influent of the biological treatment and operated on-line to determine short-term fluctuations of the organic load. After the WWF period, it took more than 3 months for the collection system to reach DWF, mainly due to the groundwater level saturation of the soil and to the infiltration of groundwater into the sewer. During the recovery phase, primary sludge samples were chosen arbitrarily for further fermentation tests (8 measuring points).

Fermentation test

The degritted sewage was pumped to a 1000 l pilot clarifier. The pilot was operated in a fill and draw mode (0.5 h filling, 1.5 h sedimentation). Sludge was withdrawn from the bottom of the containers by means of a valve. Every sampling cycle three samples were taken, namely, one of the crude sewage, one of the primary sludge and one of the supernatant. The samples were collected and immediately transported in refrigerated boxes to the lab for analysis. Crude sewage and primary sludge samples were analysed for COD, COD_F, BOD_5, BOD_{21}, SS, VSS, TN, TP, PO_4–P and VFA potential. In an air-tight vessel, 1200 ml primary sludge was continuously mixed by means of slow-moving magnetic stirrers. The experiments were run at 24±1ºC for a period of 6 days. Samples were extracted after 3 and 4 days and sent to the lab for analysis. The fermentate was used as substrate in nitrate uptake rate (NUR) tests and analysed for VFA and dissolved COD.

Calculation of the maximum denitrification rate and of the yield coefficient.

Denitrification kinetics was assessed in batch reactors, which were continuously agitated at ambient temperature. The effective working volume was 0.5l. At the start-up of the fermentation experiment, the NUR response of 250 ml activated sludge was determined for 250 ml primary sludge. With the fermentate as substrate, the NUR response of 300 ml activated sludge was determined for 200 ml fermentate. At the beginning of the tests, a nitrate nitrogen concentration of 14 mg NO_3–N.l^{-1} was inoculated to allow for anoxic conditions. To avoid limitations, 14 mg.l^{-1} NO_3–N was dosed every time nitrate decreased below 5 mg.NO_3–N.l^{-1}. The pH was controlled at 7.5±0.1 by addition of 0.5 N HCl or NaOH. When substrate and nitrate are not limiting, denitrification follows a zero order kinetic. Therefore, the maximum specific denitrification rate may be calculated from the linear parts of the kinetics. The specific nitrate nitrogen uptake rate (NO_3–N. g^{-1} MLSS. h^{-1}) is used for calculations based on the assumption that no NO_2, NO or N_2O intermediates are accumulated. Nitrate nitrogen was measured every 15 min during the first hour and every 30 min afterwards. The yield coefficient Y_H, expressed in gCOD.g^{-1}COD, was determined by the following conversion formula:

$$\text{bCOD (mg } O_2/l) = 2.86/(1-Y_H) \times \Delta NO_3\text{–N} \times (V_T/V_S).$$

The bCOD value was obtained by COD fractionation and the (NO_3–N value by the difference between the initial and the residual NO_3–N concentrations (corrected for the inoculations of NO_3–N). 2.86 (=5/4 × 32/14) is the difference in the electron accepting capacity for oxygen (4 electrons per mole) and nitrate (5 electrons per mole). VT and VS are respectively the total working and the substrate volumes (dilution factor).

COD fractionation method

The COD fractionation was based on a physical-chemical method to characterise the dissolved (S) and suspended (X) fractions, combined with long-term (21 days) BOD assessment for characterising the biodegradable fraction of the influent COD. A VFA potential test (Lie *et al.*, 1997) allowed for direct assessment of the readily fermentable COD fraction and fermentation products. The dissolved fractions were measured after filtration through a 0.45 μm filter.

Table 1 Proposed Conversion equations for COD fractions (Y_{BOD}=0.15, Roeleveld and Kruit, 1997)

Measurement	State variable obtained	Proposed conversion equations
COD	Suspended inert COD (X_I)	$X_I = COD - S_A - S_F - S_I - X_S$
BOD_{21}	Slowly biodegradable COD (X_S)	$X_s = BOD_{21}/(1 - Y_{BOD}) - S_A - S_F$
VFA pot	Fermentable COD (S_F)	$S_F = VFA_{POT} - S_A$
VFA	Fermentation products (S_A)	S_A
COD_f	Inset soluble COD (S_I)	$S_I = COD_F - S_A - S_F$

Table 2 Average fractionation of the crude sewage

	T	COD/N	bCOD		sbCOD	Inert COD	
	(°C)	$gO_2.g^{-1}N$	Sa	Sf	Xs	Si	Xi
1. DWF	18–20	7.8	16%	6%	45%	9%	24%
2. WWF	11–14	5.9	2%	10%	54%	9%	25%

Chemical methods

All the samples were analysed according to the *Standard Methods* (APHA, 1995). Speciation of VFA was performed by gas chromatography. All VFA concentrations in this document are quoted as COD. VFA concentrations are expressed as COD by multiplying the concentration of each acid with its respective COD equivalent (in $gCOD.g^{-1}$ acid). The used COD-equivalents for acetic, propionic, butyric and valeric acid are respectively 1.067, 1.514, 1.818 and 2.039 $gCOD.g^{-1}$ acid.

Results

Influent characterisation

The results of the COD fractionation of the raw sewage are summarised in Table 2.

The large variation in the VFA content shows that significant anaerobic transformations might already occur in the 23-km pressure mains and at the 16 pumping stations. During summer, the fermentable matter might be even significantly reduced in the sewer (Hvitved-Jacobsen *et al.*; 1995). Because of the large catchment area (51 km^2), the HRT is always rather long. This indicates that the driving parameter for in-sewer anaerobic transformations is mainly the temperature.

When highly dilute sewage reached the STP, oxygen concentrations as high as 4 $mgO_2.l^{-1}$ were measured at the influent. Özer and Kasirga (1995) show that in the presence of sufficient oxygen, suspended and attached growth may play a relevant role in long and ramified sewer collectors as those of the drainage system of Ghent. Important sewage aerobic transformations might occur in the sewer system under the above circumstances. This is supported by the following field measurements. During WWF periods, the COD/BOD increases and the BOD/N decreases (Figure 2). More, when during prolonged WWF periods occur, while the nitrogen load remains relatively constant (no in-sewer nitrification), the BOD load steadily decreases (data not shown).

The O_2 input and the rather dilute wastewater would have relevant implications on the stability of EBPR (e.g.: Baetens *et al.*). This is because of the low anaerobic substrate which would be available, in combination with less anaerobic HRT (due to the O_2 input and the high flow).

Fermentation of primary sludge

The results of the fermentation tests are summarised in Table 3.

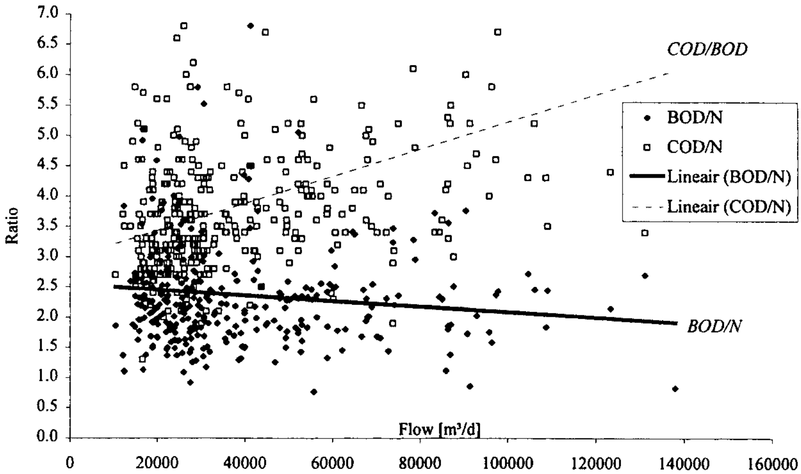

Figure 2 COD/BOD and BOD/TN versus flow (376 measuring points)

Table 3 Main determinants of pre-fermentation batch tests under optimum conditions

Parameter	Units	WWF → transient period → DWF		
1. Fermentation yield	g COD.g^{-1}COD	0.13	0.08	0.098–0.106
2. VFA speciation	%Hac. %$^{-1}$ Hpr	53/47	79/5	91/0
3. Hydrolysis yield	g SCOD.g^{-1}COD	0.12	0.11	0.19–0.24

Hac=Acetate, Hpr= propionate, S_{COD}=soluble COD

Under the tested optimum conditions, varying figures were obtained. In particular, Table 2 shows the following.

1. *Fermentation yield*: while there are case studies where the specific VFA production is as low as 0.04 gCOD.g^{-1}COD (dilute sewage!, Danesh and Oleszkiewicz, 1995) or as high as 0.17g COD.g^{-1}COD (Lilley *et al.*, 1990), most of the data available in the literature varies from 0.09 to 0.12 g COD.g^{-1}COD (e.g.: Rabinowitz, 1985; Wentzel *et al.*, 1989; Elefsiniotis and Oldham, 1994). The fermentation yield obtained during DWF is therefore in good agreement with the literature values. Instead, the fermentation yield obtained during WWF is higher, which is in contradiction with the findings of Danesh and Oleszkiewicz (1995). Moreover, the fact that the fermentation yield is as high as 13% does not mean that in absolute terms more readily biodegradable matter is produced. On the contrary, the production of rbCOD under WWF is about 50% lower than under DWF conditions (see Table 5).
2. *VFA speciation*: the large variation of relative production of individual VFAs may be related to the effect of varying influent alkalinity and temperature. While pH does not appear to affect significantly the net total VFA production (Gupta *et al.*, 1985; Moser-Engeler *et al.*, 1999), it affects their proportions (Hobson and Summers, 1967; Willimon and Andrews, 1967). This cannot be supported by experimental data because alkalinity was not measured and pH was not controlled during the experiments, as fermentation is capable of maintaining its own pH (Das *et al.*, 1995). However, the speciation of VFA has only minor implications for design.
3. *The hydrolysis yield*: the major design parameter for this type of wastewater is the hydrolysis yield. The hydrolysis yield obtained by fermentation under DWF is close to the maximally attainable hydrolysis plateau for domestic sewage, which after Henze (1992) is 30% of the sludge COD. Although, the soluble COD produced per unit sludge COD varied considerably from WWF (11%) to DWF (19–24%). We consider that the

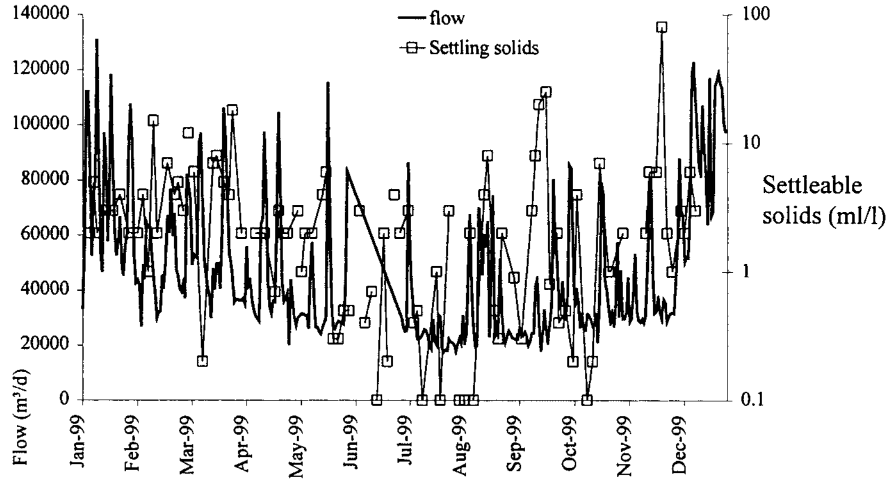

Figure 3 Settleable solids measurements versus flow (90 measuring points)

major role is likely played by the sedimentation/resuspension phenomenon in the collectors. The rationale behind it is that, during WWF, suspended compounds with better settling properties, and prone to settle in the sewer during DWF, would be transported to the STP. Indeed, Figure 3 clearly shows that at higher flows higher settleable solids are measured for the incoming wastewater. It has been shown that such extra settleable compounds have worse biodegradability characteristics (Table 2).

Moreover, the hydrolysis yield could also have been affected by the difference in temperature. Paraskevas *et al.* (1993) have shown that at higher temperature an increase in the clarifier performance as high as 100% can be obtained, and that especially the bigger particles will settle better.

The results of the COD fractionation of the fermentation products are summarised in Table 4. Column 2 represents the part of the raw sewage COD conveyed to the fermenter. The fermentate fractions are related to that COD stream.

When considering the final nominal load, the figures reported in Table 4 indicate that the readily biodegradable COD stream may be increased between +25% and +137%, when applying fermentation (Table 5).

Table 5 shows that:

1. the readily biodegradable matter produced is proportionally higher under cold weather conditions;
2. the wet weather will decrease the rbCOD production considerably (i.e.: more than 50% decrease).

An important issue for the renovation is the optimal use of the available sewage carbon source. It is therefore important to estimate the biodegradable COD losses of the system “clarifier + prefermenter”. Table 4 shows that under optimally operated conditions the biodegradable COD wasted from the pre-fermenter to the sludge handling would be about 25% of the sludge COD. Since 25–30% of the influent COD would be removed by primary clarification, this would reduce the bCOD to the bioreactors of 8%. On top of this, while wasting sludge from the fermentation unit a loss of entrapped rbCOD will also occur (Bundgaard *et al.*, 1992). At the design operating conditions of the STP of Ghent, a 15–20% fermentate loss has to be accounted for.

Denitrification tests

The results obtained by the denitrification tests are summarised in Table 6.

Table 4 COD fractionation of the fermentate

	COD primary sludge	Sa	Sf	Xs	Si	Xi
DWF	30% raw sewage COD	22%	17%	25%	9%	27%
WWF	25% raw sewage COD	13%	3%	56%	8%	23%

Table 5 Expected increase on readily biodegradable COD due to the pre-fermentation process (no rbCOD losses)

	Temperature °C	rbCOD produced $kg.d^{-1}$	rbCOD increase %
DWF	20	2186	43%
	10	2416	137%
WWF	20	966	25%
	10	628	30%

Table 6 Main determinants of the NUR batch tests fed with the fermentate under optimum conditions

Parameter	Units	WWF →	transient period →	DWF
NUR_{max}	$mg.g^{-1}MLSS.h^{-1}$	2.2–2.5	3.2	2.5–2.8
Y_H	$gCOD.g^{-1}COD$	–	–	0.55

The denitrification rate attainable with fermentate was on average 2.5 mg $N.g^{-1}MLSS.h^{-1}$ (~3.7 mg $N.g^{-1}$ $MLVSS.h^{-1}$), which is close to the denitrification rate based on acetate (Moser-Engeler *et al.*, 1999).

Discussion

The above figures were used as a background for process evaluation in order to find the optimal concept for the retrofit of the STP. A comprehensive modelling exercise, which is fully described in Bixio *et al.* (2000), was used for supporting scenario analysis.

Simulations show that the effect of different weather flow conditions on the effluent nitrogen concentrations is relevant. Indeed, the effect of pre-fermentation on the nitrogen emissions is limited for WWF (i.e.: – 0.6 mg N_{EFF}/L) and important for DWF (i.e.: – 2 mg N_{EFF} /L).

The slowly biodegradable COD constitutes the bulk of the bCOD of the crude sewage (Table 2). Since the bCOD/N is critical (Figure 2), enlarging the STP reactors would be necessary if no side-stream hydrolysis took place. In that case, the dimensioning of a new bioreactor must be based on the anoxic hydrolysis of slowly degradable matter, which is a slow-rate process. An alternative to the enlargement is the dosage of external carbon source, but this implies extra operating costs for chemical dosage and for extra sludge production handling. The three options were compared in terms of cost-effectiveness. Table 7 shows that pre-fermentation is the most cost-effective, if not considering the benefits enjoyed by the anaerobic digestion of the sludge.

In comparing the benefits of pre-fermentation, the type of sludge handling should be accounted for in the analysis. Simulations show that the pre-fermenter would decrease the energy recovery from the anaerobic digester (Figure 4).

Figure 4 shows that the energy losses would be very different for WWF and DWF. When comparing the prefermentation option to the one with primary clarifier, a decrease of the energy recovery of respectively 4720 $MJ.d^{-1}$ and 7279 $MJ.d^{-1}$ is expected. Moreover, an

Table 7 Estimation of the investment and operating costs for the dosage of internal and external carbon source

	Operating cost (Euros/y)	Investment costs (Euros)	Δ [ext. C-source cost] (Euros/year)	Euros/kgN
Methanol	165,000	0	+75,000	1.04
Industrial by-products	113,000	0	+15,000	0.72
Endogenous resp.	97,000	1,454,000	0	1.17
Fermentate	97,000	0	0	0.59

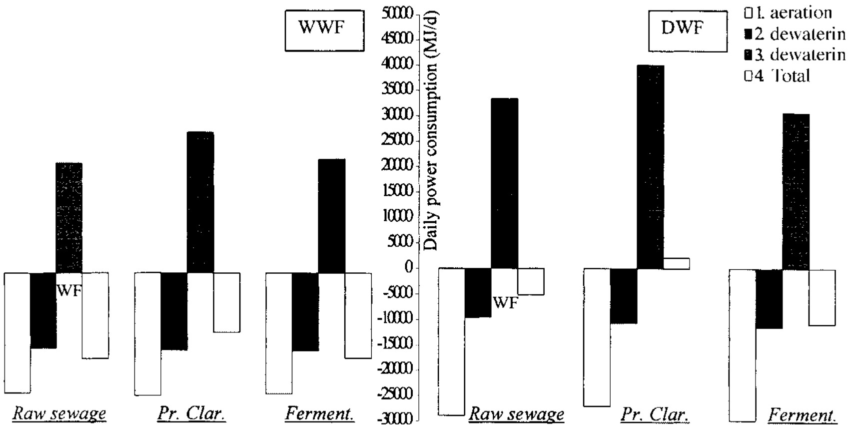

Figure 4 Simulated energy balance, expressed in MJ.d^{-1}: (a) WWF, (b) DWF

extra solid waste would be produced (for WWF and DWF, respectively 0.2 ton.d^{-1} and 1.7 ton. d^{-1} extra sludge production). The type of sludge treatment is therefore an important factor when assessing the feasibility of pre-fermentation technologies.

Conclusions

Wastewater in sewer systems is subject to important physical, chemical and biological changes. In combined sewers, the transport of sediments and the anaerobic and aerobic transformation of organic matter may play an important role in the overall efficiency of pre-fermentation of primary sludge.

For the case under study, the dilute sewage affects considerably the performance of the fermentation process, which will decrease by more than 50%.

When assessing the performance of pre-fermentation, the sewer should be considered as an integral part of the urban wastewater system. This step is essential for scenario analysis.

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