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01

WETLANDS AND OTHER EXTENSIVE TREATMENTS

VALORIZATION OF EXHAUSTED ACTIVATED CARBON AND ALUM SLUDGE FROM DRINKING WATER TREATMENT PLANTS AS REACTIVE SUBSTRATE IN TREATMENT WETLANDS

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Abstract

Dehydrated sludge and depleted activated carbon are two inert wastes generated in drinking water treatment stations (ETAPs) that use water collected from surface sources. These residues have usually been disposed of in landfills or applied over agricultural soils. Moreover, dehydrated sludge has successfully used a phosphorus adsorbent substrate in artificial wetlands in recent decades; however, depleted activated carbon advantages as adsorbent material for emerging organic pollutants are a less studied topic. In the present study, the potential of both materials mixture for the tertiary treatment and refining of urban wastewater from small towns is evaluated.

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INTRODUCTION

Dehydrated sludge and depleted activated carbon are two inert wastes generated in the purification process of water captured from surface sources. The dehydrated sludge along with sludge from urban wastewater treatment plants (WWTP), previously treated, is usually managed as an agricultural enhancement whereas the depleted activated carbon is disposed of in inert landfills. In order to take advantage of the adsorbent capacity of this sludge mixture, several studies have been carried out at the IIAMA of the UPV using dehydrated sludge as an active substrate in artificial subsurface flow wetlands since 2016. Once the material is dehydrated, it is crushed into small gravels (about 0.1-0.5 cm) that are used as a substrate; the very fine remnant fraction is disposed of in a landfill. Outcomes from pilot plant scale wetlands are currently being positive in terms of phosphorus and nitrogen removal when these wetlands are used as refining treatments for WWTP effluents after secondary or tertiary treatment (Oliver et al. 2021).

The term depleted granular activated carbon is relative since its disposal to landfill does not take place when it is completely exhausted. However, it is economically more profitable to buy new material instead of regenerating the current one. This is because, after each regeneration cycle, adsorbent properties are lost, so there comes a time when the cost of regeneration no longer compensates for the acquisition of new material. It has been proven that this exhausted carbon is capable of adsorbing organic compounds (Luján-Facundo et al. 2019), but its ability to retain phosphorus is very limited according to previous work carried out by the working group.

The hypothesis of the present study is that the properties of the mixture of both granular activated carbon and dehydrated sludge can simultaneously eliminate nutrients and persistent organic compounds such as pesticides and emerging pollutants more efficiently when comparing both materials separately.

The final objective, to achieve an efficient mixture of both wastes, has a very important secondary objective: the use of the pulverulent residue from the crushed dehydrated sludge, in order to be able to put this residue to use and that it does not have to be managed by landfill disposal.

In this study, the concept of “efficient mixing” is closely related to the permeability of the resulting material. This permeability is key when establishing the surface hydraulic load that will be able to be introduced. Activated carbon has a very high permeability, as it is a classic filter material in the field of water, but dehydrated sludge dust has a very low permeability. Hence the objective of obtaining a mixture that allows treating a significant flow rate and taking advantage of the adsorbent capacities of both materials.

METHODS

Preliminary tests were carried out on a glass column, with a diameter of 4 cm and a length of 12.5 cm, placing a mesh and a layer of gravel 4 mm in size at the bottom of the column, to avoid the loss of material. On top of the gravel layer, a 12.5 cm layer of various material mixtures was introduced: fine dehydrated sludge (5, 10, 15, 20, 30 and 40%), produced in milling and which is currently considered rejection and spent activated carbon (95, 90, 85, 80, 70 and 60% respectively). To evaluate the infiltration rate, 20 ml and 50 ml of distilled water were introduced, and the time was determined in which the water percolates. Once the time and percolated volume were determined, the infiltration rate was determined.

From the last two mixtures, the hydraulic conductivity was determined according to the method of Gabriels et al. (2011). Subsequently, the mixture that provided the best relationship between the proportion of dehydrated sludge and the hydraulic conductivity was selected. With the selected mixture, a larger column (59 mm internal diameter) was assembled. In this, 2 cm of gravel was placed in the bottom and then 35 cm of the selected mixture.

The column was fed for eight weeks with partially treated wastewater from the secondary treatment of the Carrícola WWTP, a small municipality (100 hectares) in the province of Valencia, whose WWTP is based on artificial wetlands. Different hydraulic retention times (HRT) were tested (0.5, 1, 21 and 63 hours). The following variables were analyzed: ammonium, nitrites, nitrates, phosphates and turbidity. Specifically, COD measurements were carried out to compare the performance of the elimination of organic compounds with a column whose packing is granulated dehydrated sludge.

RESULTS AND DISCUSSION

The results show how as the proportion of fine dehydrated sludge increases, the permeability of the filter pack decreases (Figure 1). The hydraulic conductivity was determined for the mixtures with the highest proportion of fine dehydrated sludge, 30 and 40%, resulting in 15.3 and 5.5 m / d respectively. The values obtained for the two different percentages of mixtures are within the range established by Kadlec & Wallace (2008) that oscillates between 6-36 m / d, for particles of 0.5 and 3 mm. Finally, the mixture was selected with 30% fine dehydrated sludge and 70% depleted activated carbon, to ensure a greater durability of the filling material and to enable operation at high hydraulic loads.

One of the purposes is to be able to use the fill in refined artificial wetlands for which large areas are not required.

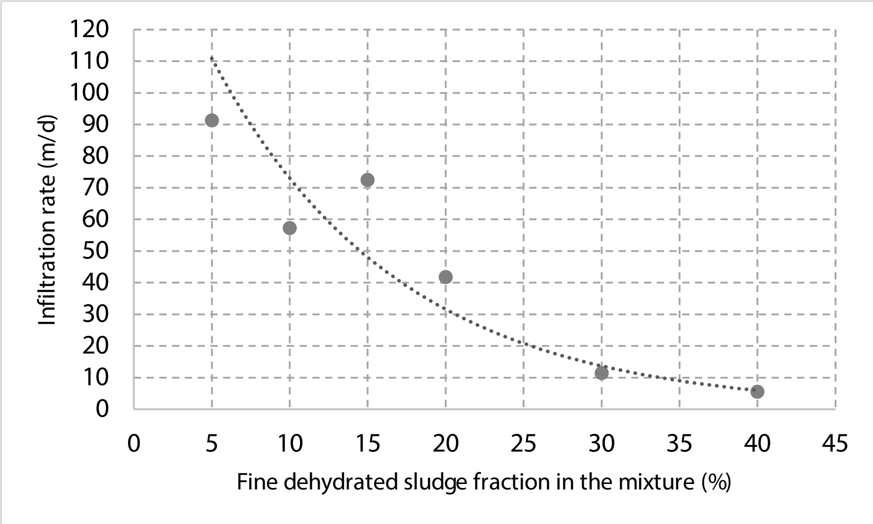

Figure 1. *Infiltrationratesofthedifferent mixtures tested.*

The phosphorus results show a high reduction in the concentration of this nutrient (Figure 2), with an average removal efficiency of 99%. To be able to reduce the phosphorus concentration in rural populations through artificial wetlands with reactive substrate is very advantageous due to its simplicity of operation and especially beneficial in those places where the receiving medium is a watercourse with low dilution capacity and, therefore, sensitive to eutrophication. Regarding the influence of TRH, a clear behavior is not observed. It should be mentioned that the samples were in the vicinity of the detection limit of the analysis method and therefore the observed variations can be considered insignificant.

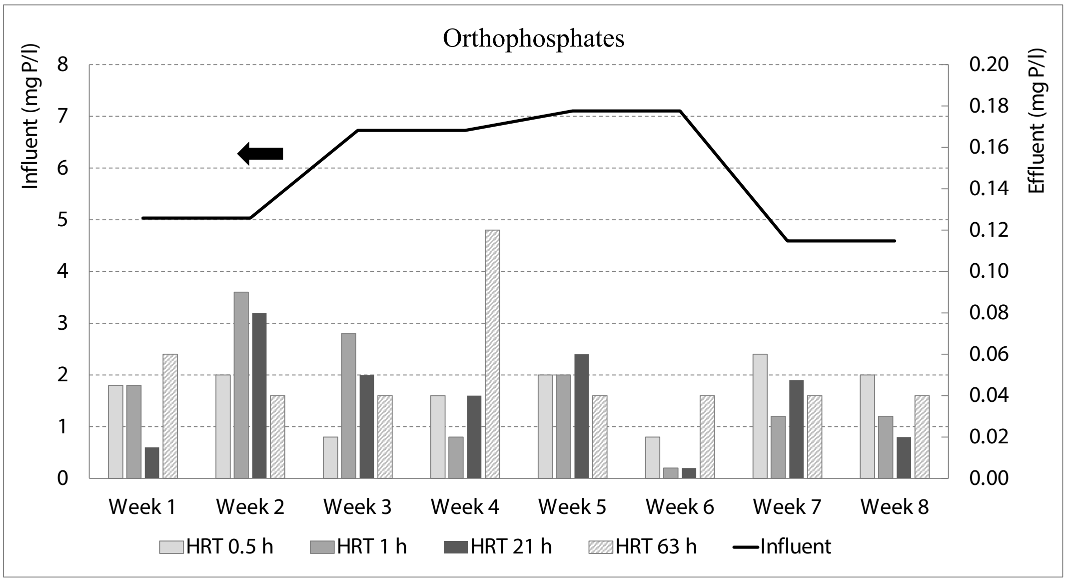

Figure2. *Phosphateconcentration (mg P / l) in theinfluent (main axis) and effluent (secondary axis) ofthecolumn.*

In addition, the ammonium concentration was also significantly reduced, which also translates into a clear benefit for the receiving environment. On average, the efficiency was 76%, the concentration decreased from 21 to 5 mg N / l during the first four weeks, and from 45 to 12 mg N / l in the rest of the period.

Finally, a comparison was made in the elimination of COD with another column whose filling was granulated dehydrated sludge (0.83-2.38 mm). The results show that the column packed with a mixture of fine dehydrated sludge and depleted activated carbon reduced the COD to a greater extent, with efficiencies higher than 70% (average of 74%), while the column with granulated dehydrated sludge provided an average reduction efficiency of 42%. Another aspect to highlight is that in the column with mixture, no significant differences were observed for the different HRTs, possibly because much of the reduction was due to the adsorption process. In contrast, the dehydrated sludge column did present a COD removal efficiency more dependent on TRH because in this the mechanism of elimination of organic matter is biological degradation and, as it is water from a secondary treatment the organic matter present in it was slowly biodegradable.

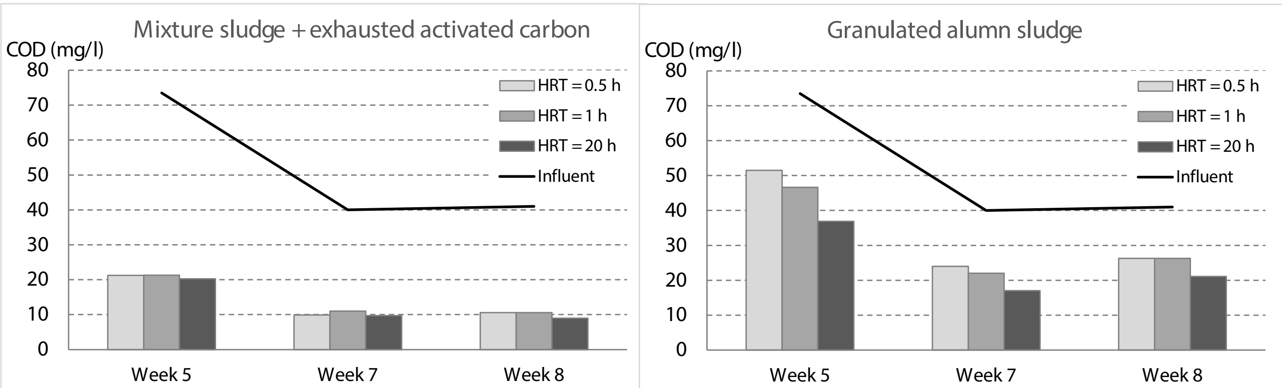

Figura 3. *Comparison in thereductionoforganicmatterbythecolumnwith a mixture of fine dehydratedsludge and exhaustedactivatedcarbon and bythegranulateddehydrated alumslu-dgecolumn.*

CONCLUSIONS

The use of two residues from DWTP, dehydrated sludge and spent activated carbon, as reactive substrates for constructed wetlands has multiple environmental benefits. On the one hand, it contributes to making a more sustainable use of natural resources because by taking advantage of the waste, the useful life of the coagulant and activated carbon used in the ETAP is being extended and it also translates into better management of the waste generated there. With this use, the principles of the circular economy are incorporated into the urban water cycle. On the other hand, the use of this type of substrates in a combined way allows to significantly improve the quality of the treated water, reducing both the concentration of nutrients and organic matter. In this way, a high-quality effluent is generated whose impact on the receiving environment will be significantly less, especially suitable in small towns.

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13 YEARS OF OPERATION OF A TWO-STEPSVERTICAL FLOW CONSTRUCTED WETLAND

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Abstract
This paper analyses the performance during 13 years of a vertical flow constructed wetland.The system is composed of two units that intend to unite the two stages of the French vertical flow constructed wetlands into one single stage, inserting an intermediate passive ventilation system that allows aeration of the filtering layers. After 13 years of operation, the bi-filter shows excellent robustness and stability, achieving BOD and TSS removal efficiencies of around 96% and COD of 92%. Likewise, it shows ammonium removal rates of 90%, demonstrating excellent nitrification.

INTRODUCTION

Various configurations of constructed wetlands (CW) can be found with different forms of feeding and management. Among these systems we find vertical flow CW, where aerobic conditions within the wetland are sought that favour nitrification, feeding them intermittently and through the use of helophytes (emergent hydrophytic plants). On the other hand, these systems generally need a previous primary treatment that removes a good part of the suspended solids to avoid the clogging of the bed.

In France, a particular configuration of constructed wetlands has been developed to treat wastewater in small communities, which has proven to be very successful. Most of them are composed of two stages of constructed wetlands of vertical flow in series, with three units in parallel in the first stage and two in the second, with successive feeding periods (3.5 days) and rest periods of 7 days. in the first stage and 3.5 days in the second, to maintain permeability, oxygen content and control of microbial biomass growth (Molle*et al.,* 2005, Morvannou*et al.,* 2015, Rizzo *et al.,* 2020). The special feature of this system is that it accepts raw sewage directly onto the first stage allowing for easier sludge management in comparison to dealing with primary sludge from an Imhoff settling / digesting tank (Molle et al., 2005). Due to their design and mode of operation, French Vertical Flow Constructed Wetlands (VFCWs) accumulatesuspended solids from the inflow wastewater in the form of a sludge layer at the surface of the first filter (Kania *et al.,* 2019).

In 2007, a vertical flow treatment wetland (HumArt Bi-Filtre) was built by the company OptimiaMedio Ambiente S.L. and put into operation in the experimental wastewater treatment plant (WWTP) of the Centre for New Water Technologies Foundation (CENTA) in Carrión de los Céspedes (Seville).After a period of 13 years, an analysis of the performance has been carried out.

METHODS

The system is an adaptation of the French system and is designed to treat wastewater from a population of 35 P.E. and a maximum flow of 8.7 m^3 d^{-1}. It is installed in 2 equal tanks of 5 m in diameter (20 m^2 of surface) and 1.7 m in height each. Each of the units is made up of two overlapped vertical percolation stages, managing to share the two purification stages in the same surface space.The composition of these stages from top to bottom is: 1) First Stage (primary treatment): 60 cm of gravel substrate 2-6 mm, 15 cm of gravel substrate 15-25 mm and 30 cm of gravel substrate 30-60 mm; 2) Second Stage (secondary treatment): 30 cm of sand substrate 0-2 mm; 10 cm of 2-6 mm gravel substrate and 30 cm of 15-20 mm gravel substrate where the bottom drainage pipe network is inserted. To provide ventilation to first stage, a passive ventilation device has been inserted into the gravel layer (30-60 mm), consisting of a series of connected perforated pipes to a ventilation chamber, in such a way as to optimize the oxygen convection to the second filter stage (Figure 1). To provide ventilation to the second stage, some vertical perforated pipes connected to the bottom drainage system where installed. The system was planted with *Phragmites australis*.

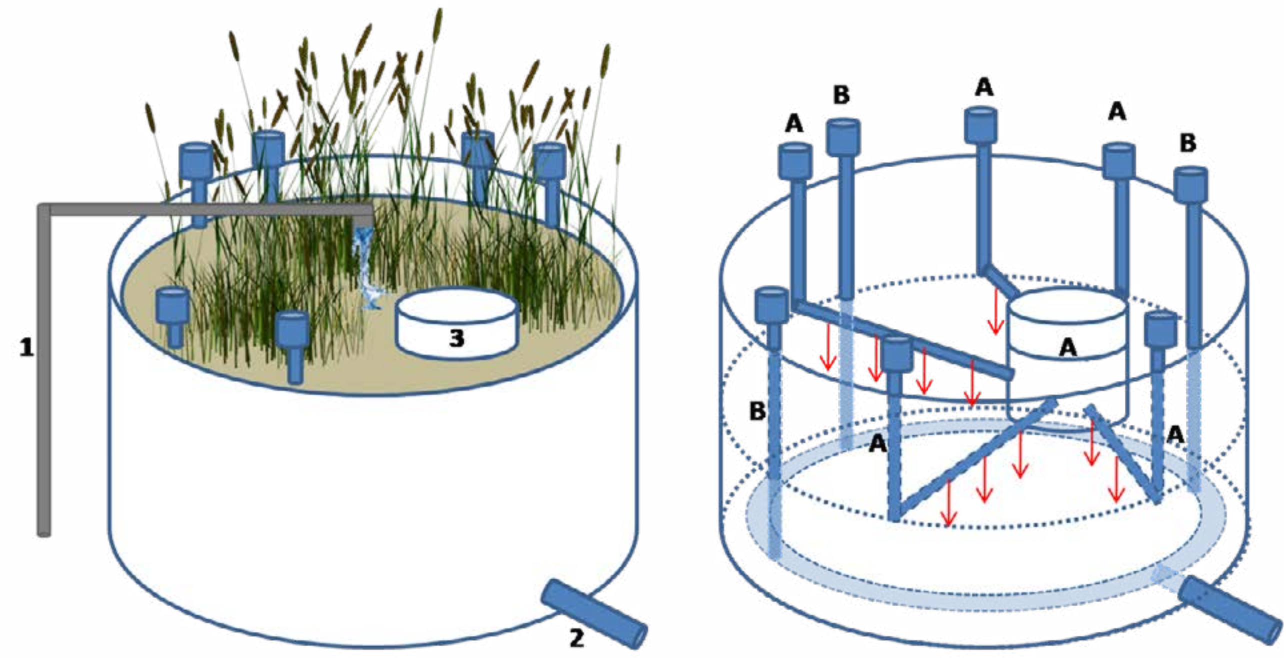

Figure 1 *Schematic of the design of each unit of the system. A. Intermediate passive ventilation system; B. Passive ventilation and bottom drainage system. 1. Feed pipe; 2. Drainage pipe; 3. Ventilation chamber.*

Weekly samples of the influent and effluent have been taken for two consecutive years and, thereafter, periodical samplings have been carried out to verify the operation of the system. The BOD_5, COD, TSS, NTK,NH4-N,NO3-N,TP and PO_4-P have been analysed using standard methods (APHA, 2002).After 13 years of operation, the sludge accumulated on the surface of the wetland has been collected and analysed to study its mineralization.

RESULTS AND DISCUSSION

As can be deduced from the following data, the pollutant removal efficiency achieved by the system has remained stable over time. The system was fed with wastewater from the pretreatment of the experimental plant, with an average flow of 3.6 ± 0.5 m^3 d^{-1}. Despite the high inlet concentrations, it showed effluents with very low concentrations of organic matter (around 10 mg L^{-1} of TSS and 53 mg L^{-1} of COD, and less of 13 mg L^{-1} of BOD_5) (Table 1). The system performs an excellent efficiency in the removal of suspended solids (96 ± 4%) and organic matter (97 ± 2% of BOD and 92 ± 5% of COD) (Figure 2).

Regarding nutrients (Table 1), the ammonium concentration of the effluent is very low (around 5 mg L), but not so much the concentration of total nitrogen and phosphorus. These data show that the system performs a good nitrification, eliminating 90 ± 11% of the ammonium, but a low denitrification, eliminating 43 ± 25% of the total nitrogen. The system also removes 35 ± 14% of the total phosphorus (Figure 2).

Table1. *Influent and effluent concentrations of pollutants (average ± SD) during the first two years and during the period 2010-2019 of performance.*

Period	Sampling point	TSS (mg L^{-1})	BOD (mg L^{-1})	COD (mg L^{-1})	TN (mg N L^{-1})	NH4-N (mg N L^{-1})	NO3-N (mg N L^{-1})	TP (mg P L^{-1})	PO4-P (mg P L^{-1})	OLR (g BOD m^2d^{-1})
2007-2009	Influente	283±205	387±119	684±226	62.7±12.7	44.2±12.1	3.4±47	10.5±2.0	7.6±1.9	64±23
	Efluente	10±10	<13	53±36	39.2±16.8	5 ± 5.6	26.2±18.1	6.2±1.6	6.0±1.5	
2010-2019	Influente	286±212	379±167	652±246	74.6±26.8	53.5±23.2	13.1±6.2	8.7±2.8	6.3±2.4	69±31
	Efluente	15±8	<13	67±39	44.3±21.8	3.0±2.5	32.5±8.4	5.2±0.7	4.7±0.8	

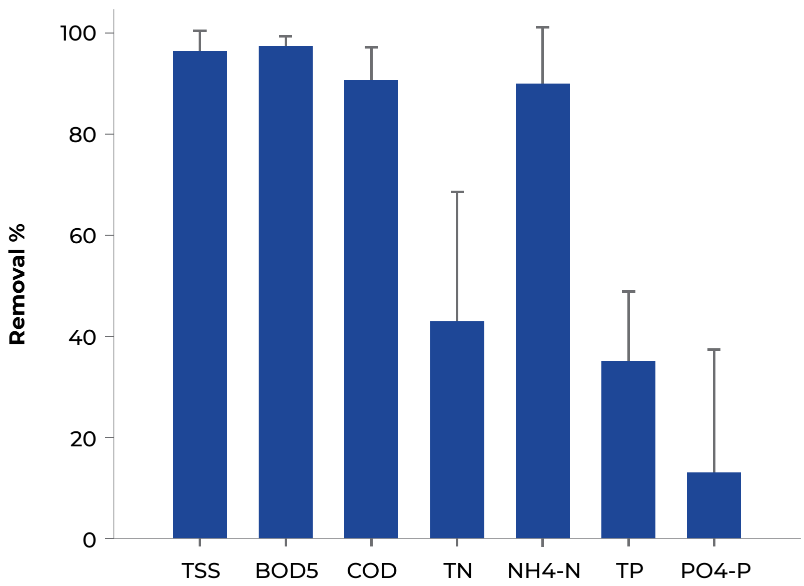

Figure 2. *Removal of pollutants in percentage*

According to Molleet *al.,* (2005), French CW systems, if well designed, can achieve an outlet level of 60 mg L^{-1} in COD, 15 mg L^{-1} in TSS and 8mg L^{-1} in TKN with an area of 2–2.5 m^2 PE^{-1}. Our VFCW shows similar performance with 1.74 m^2 PE^{-1}.

Tabla2. *Average organic load, population equivalents served by the system and surface area required per population equivalent for every unit.*

Organic load (g DBO_5/d)	Populationequivalents (PE)	Surface (m2/PE)
1386 ± 501	23.1	0.87

After 13 years of operation, the sludge accumulated on the surface of the wetlands represented a volume of 0.55 m^3 in each unit. This is a very small amount of sludge if compared with the sludge produced in a primary treatment. The sludge height was around 5 cm in the centre of the wetland units. Four samples of the sludge in every unit were analysed in order to determine their degree of mineralisation (Table 3). Based on an average daily SS load of 1024 g d^{-1}, and a SS removal rate of 96%, the mass balanceof SS input on these 2 filters over 13 years can be calculated to be 4660 kg SS.Theevacuated massin the 2 filters (around650kg per filter, dry matter (DM) content around 30%) is estimated to be 400 kg SS, which represents almost 10% of the SS introduced by the wastewater. Thus, the mineralisation rate was 90%, much higher that the rates reported in other studies (Boutin *et al*., 1997, Molleet *al*. 2005).Mineralisation of particulate organic matter progressively reduce the organic matter content of the depositswithin the sludge layer (Kania *et al*., 2019).

Table 3. *Dry matter and organic matter contents in the samples of the sludge accumulated over the wetlands.*

Filter unit	Dry matter (g kg^{-1})	Average	Organic matter (% of dry matter)	Average
1	288	317 ± 49	38.2	33.4 ± 6.3
	307		34.0	
	403		24.8	
	309		29.8	
	280		40.3	
2	318	294 ± 23	32.8	38.1 ± 3.5
	266		42.7	
	297		38.3	
	275		38.8	
	313		37.7	

CONCLUSIONS

The HumArt Bi-Filtre maintains a stable performance after 13 years of operation. This type of vertical flow constructed wetland achieves very high removal rates for suspended solids, organic matter and ammonia, complying the discharge requirements of the Directive 91/271/CEE, relative to the quality of urban wastewater with a single treatment stage. The area required for this is less than that required by the French two-stage constructed wetland system.One of the advantages of this system is the management of a minimal amount of sludge that itrequires. CENTA acknowledgetothe company OptimiaMedio Ambiente S.L. for the installation of the CW in the experimental plant.

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INNOVATION IN VEGETATION FILTERS FOR WASTEWATER TREATMENT AND RESOURCE RECOVERY

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Abstract

In the frame of Challenges defined by the Spanish Strategy of Science, Technology and Innovation, in particular to the Challenge entitled "Action for Climate Change and efficiency in use of resources and raw materials", R&D&i related to efficiency in use of resources and raw materials and the integrated management and sustainable use of water resources in rural areas, is a priority. In this context, Vegetation Filters (VFs) can be a sustainable solution to treat wastewater and to recover resources such as water, nutrients and biomass from small municipalities and isolated dwellings. The objective of the present work is to evaluate the treatment capacity of a Nature based Solution (NbS) such as VF to remove contaminants (organic matter, nutrients, and pathogen microorganisms) from a high organic load wastewater and contribute to reduce the surface requirements for the implementation of this solution and to improve further the contaminants attenuation by the evaluation of two sustainable soils amendments, woodchips from pruning remains and biochar.

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INTRODUCTION

In Spain, more than 2.7 million people live in municipalities with less than 2000 inhabitants. The technical and economic limitations of small and scattered populations compose the effective implementation of conventional wastewater treatments. Non-conventional treatments such as nature-based wastewater purification systems have been reported as feasible solutions for these communities, which have limited access to sewage networks (Ortega *et al*., 2011). The main advantages of these treatments are their low cost for implementation, limited energy use and reduced operation andmaintenance requirements.

Vegetation Filters (VFs), as a part of Land Application Systems, are Nature based Solutions (NbS) that take advantage of the natural treatment mechanisms to remove wastewater contaminants. The wastewater is applied to irrigate forestry (mostly poplars and willows) that can be used mainly for biomass production. Physical filtration, sorption onto soil, biodegradation and plant uptake are the main processes responsible of the contaminant attenuation through the subsurface. The efficiency of VFs to remove traditional wastewater-originated contaminants has already been reported (de Miguel *et al*., 2014; Martínez *et al*., 2018). However to our knowledge, their capacity to attenuate microparasites/pathogen (*E. coli*, intestinal helmints eggs) has not yet been evaluated.

The present study, in the frame of the FILVER+ project (Spanish Ministry of Economy and Competitiveness) reports the results of a 4 years of contaminants monitoring (organic matter, nutrients, and pathogen microorganisms) carried out using a poplar VF for the treatment of wastewater originating from an office building, as well as the results from the evaluation of the capacity of two sustainable soils amendments, woodchips from pruning remains and biochar to improve further the contaminants studied attenuation.

METHODS

Study site description

The study site is located in South-East of Spain at the R&D&I Centre of Carrión de los Céspedes (Seville). The system is based on short-rotation coppice of poplars (*Populus alba*) with a high plantation density (10,000 plants/ha). The poplars were planted in March 2011 from unrooted hardwood cuttings and since then they have been harvested every 3 years. During the period of study the VF has treated a wastewater effluent from an office building that produce an average volume of 0.21 m^3/day, primarily treated in an Imhoff tank of 2.5 m^3 capacity.

The Imhoff tank effluent (ITE) was applied according to its production-depending on daily sanitary discharges- in free flow furrows (*Fig.1a*). Irrigation lines were grouped into five blocks. The irrigation schedule was designed to apply effluent to each block once per week (5 working days per week). The quality of the infiltrated water was monitored with a Gee Passive Capillary Lysimeter (Ø = 24.5 cm) (Decagon Devices, Pullman, Washington, USA) that collected water from the upper 90 cm of the vadose zone (*Fig. 1b*). Groundwater was monitored approximately every two months in a down-gradient piezometer (filter screen depth: 10 m) (*Fig. 1c*).

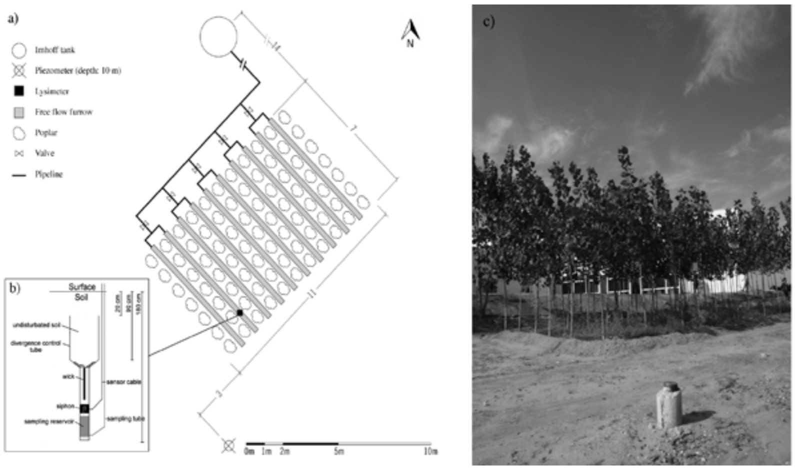

Figure 1. *a) VF scheme; b) lysimeter; and c) real image of the VF with the piezometer in the foreground*

Infiltration experiments

To identify which amendment will improve the attenuation of contaminants during vadose zone infiltration at the VF, three unsaturated infiltration-column experiments at laboratory scale were performed. The systems contain natural soil, natural soil amended with 3% w/w of woodchips from poplar pruning remains and natural soil amended with 3% w/w of biochar. (*Figure 2*).

Figure 2. *Details of the unsatured-infiltration colums, woodchips from pruning remains and biochar*

Target compounds

To evaluate the contaminants removal, both at pilot (VF) and laboratory scale (sustainable soils amendments, woodchips and biochar), the compounds studied are the related to the organic matter (BOD_5, COD), nutrients (N and P) and microparasites/pathogens included in the Spanish regulation for water reuse (RD 1620/2007, i.g. *E. coli* and intestinal helminths eggs, specifically *Ascaris* sp.).

RESULTS

The results obtained from the pilot-scale monitoring of the pollutants under study, mainly organic matter, total nitrogen and E. coli, indicate that there are significant differences between the residual water and the infiltration and underground water; resulting in significantly lower mean concentrations in infiltration and groundwater. The high removal rates show the role of the soil in the unsaturated zone during infiltration into groundwater, as well as its capacity as a natural reclamation technology for the recharge of aquifers by percolation through the ground.

The results obtained from the evaluation of the amendments (poplar wood chips and biochar) by means of infiltration tests in columns under unsaturated conditions, have allowed to determine that biochar is the amendment that presents the best elimination results at the level of suspended solids, COD, *E. coli* and eggs of *Ascari*s sp. On the other hand, chips have turned out to be the amendment that shows the best performance removal at the nutrient level (total nitrogen, ammonia nitrogen, nitrates, total phosphorus and phosphates).

CONCLUSIONS

The study shows that the first 90 cm of soil are the most effective for the attenuation of some pollutants as organic matter, total nitrogen and pathogens as *E. coli*. Also, the results obtained in the project invite us to incorporate woodchips from pruning remains and biochar as sustainable soil amendments and potential treatment improvement.

In this sense, Vegetation Filters can be considered, in addition to highly effective natural systems in the purification and reclamation of wastewater, a clear example of a circular economy; providing highly valuable inputs for the sustainable development of small agglomerations.

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EXTENSIVE WWTP PROJECT WITH FLOATING WETLANDS WITH HELOPHITE SIEVES

Location: **LORA DE ESTEPA**
Promoted by: **SUPRAMUNICIPAL PUBLIC SERVICES AREA of the SEVILLE COUNCIL.**
LORA DE ESTEPA CITY COUNCIL.
Projected: **MARTINEZ CASTILLA** Ramón Ignacio (Civil Engineer).
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1. OBJECTIVE:

Define, design, and calculate the works necessary for the construction of a new urban WWTP with the premise of maximum energy savings during the operation

After studying various possible extensive treatment technologies (vertical flow, horizontal flow and floating subsurface with helophyte sieves), the design of an artificial wetland using floating helophyte sieves was chosen due to its lower investment cost as it does not use a granular bed support (which would imply a higher initial investment and the possibility of clogging the system in the short term).

2. STARTING DATA:

Sizing is based on the following data.

Datos de Cálculo			
POBLACIONES Y CAUDALES			
		ACTUAL	HORIZONTE (2038)
Población estimada	(hab)	875	849
Población estimada	(hab-eq))	1,025	999
Dotación	(l/hab./dia)	190	190
Caudales			
Caudal medio diario	(m3/dia)	195	190
Caudal medio horario	(m3/h)	8,11	7,91
	(l/s)	2,25	2,20
Coeficiente punta adoptado		1,50	1,50
Caudal punta diario	(m3/d)	292,13	284,72
Caudal punta horario	(m3/h)	12,17	11,86
	(l/s)	3,38	3,30

Datos de Cálculo			
CARGAS CONTAMINANTES ENTRADA			
		ACTUAL	HORIZONTE (2038)
DBO5			
Concentración media	(mg/l)	315,00	315,00
Carga diaria	(kg/dia)	61,35	59,79
DQO			
Concentración media	(mg/l)	819,00	819,00
Carga diaria	(kg/dia)	159,50	155,45
SS			
Concentración media	(mg/l)	146,00	146,00
Carga diaria	(kg/dia)	28,43	27,71
pH		7,70	7,70
Tª media del agua invierno	(°C)	18,00	18,00

Datos de Cálculo			
RESULTADOS AGUA TRATADA			
		ACTUAL (A2)	HORIZONTE (A1)
DBO5	(mg/l)	<25	<25
DQO	(mg/l)	<125	<125
Sólidos en Suspensión	(mg/l)	<25	<25
PARÁMETROS MICROBIOLÓGICOS DE VERTIDO			
E. Coli		< 5.000 ml.	< 5.000 ml.

3. TREATMENT LINE:

The treatment line studied is the following:

– **Inflow** from the urban center.

– A **pre-treatment**, which includes:

  – **Roughing.**

  – **De-sanding-degreasing.**

– A **biological treatment**, which includes:

– **Anaerobic pond. DTH (**1 unit).

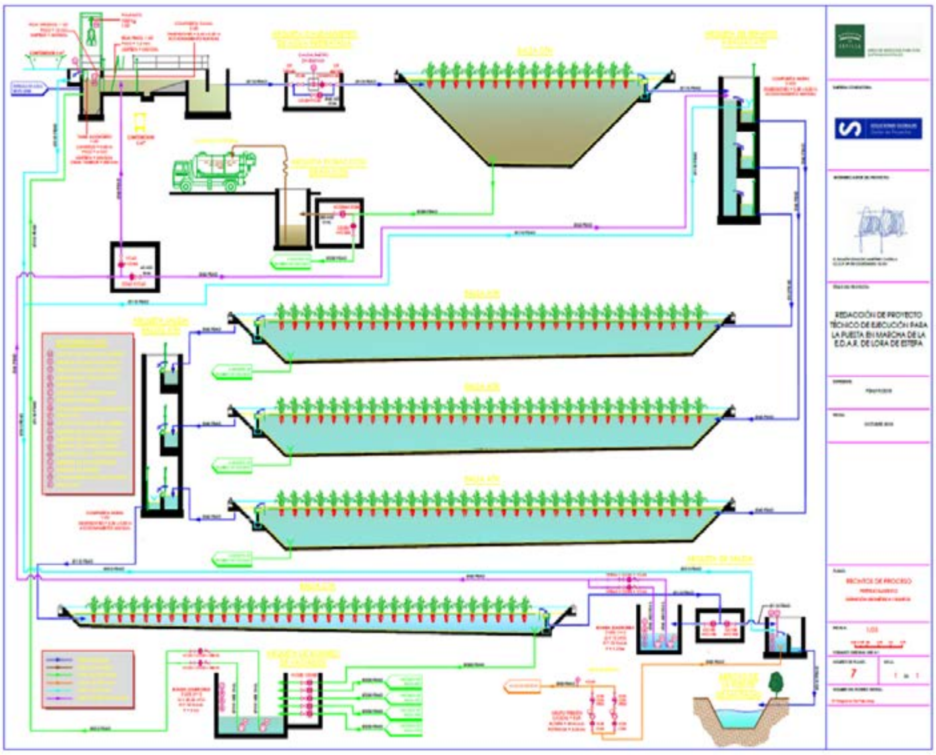
  – **Aerobic pond with flotation helophyte sieve. ATH** (3 units).

  – **Refining pond with floating helophyte sieve, CTH** (1 unit).

– **Outlet water chlorination.**

– **Discharge pipeline.**

The purification process through the floating wetland using helophyte sieves is as follows:



Correct pre-treatment, maintenance, and operation of the WWTP are essential for the correct running of the entire process.

The adopted solution involves less electromechanical equipment, which facilitates the operation of the plant and minimizes the operating costs, making it the most economical and rational solution for this type of facilities.

1st PRE-TREATMENT

The pre-treatment consists of a prefilter well equipped with a clamshell grab. A general by-pass chamber for the plant will be built adjacent to this enclosure. A weir screen will be installed in the wall adjacent to the two enclosures. An ultrasonic meter will be installed on the spillway in order to measure the possible flow of water discharged by the by-pass.

In the wall adjacent to both enclosures, a manual grating with a 15 mm clearance will be installed. The width of the roughing channel adopted is 0.4 meters, and a manual inclined grating with a 1.5 mm pitch will be installed in it.

The channel is delimited by two manual channel penstocks sluice gates valves measuring 0.4 x 0.5 meters each. In this way, by closing both penstocks, the channel can be isolated if necessary.

The discharge will then flow into the grit-degreasing area, which consists of an initial area 0.64 meters wide and 3 meters long, which has the function of gritting.

Sand and / or decantable solids will settle to the bottom of the channel. From that pit, with the periodicity deemed convenient and / or necessary, the solids will be manually extracted.

After the de-sanding process, the water will continue its course in an adjoining pit, which contains two chambers separated by an intermediate wall. This intermediate wall does not reach the bottom of the chamber so that the grease and floats are retained in the first chamber of the pit, and the water will follow a downward flow, circulating under the intermediate wall, and passing into the second chamber which is now free of grease.

Through buried pipe we will direct the discharge to the first stage of the secondary process.

2nd BIOLOGICAL TREATMENT

The first stage of the biological treatment consists of a DTH-type pond measuring 17.90 x 17.90m in plan, and 4 meters of useful depth, where the anaerobic stage of the treatment takes place.

Thanks to the depth of this pond, the chemical / biological processes of influent nitrification and anaerobic fermentation and sludge digestion can take place in it.

The surface of the pond is equipped with a vegetable sieve, with the function of mitigating the emission of odors and avoiding the proliferation of algae (which negatively influence the anaerobic processes that must take place in the pond)

The density of these plants is less than the water, so they float, but a structure is necessary to allow the plants to spread evenly over the surface of the pond. It is also essential to keep the aerial part upright and out of contact with water, which is essential for the survival of the plant.

This double function is fulfilled by a structure that will be installed on the surface of the pond's water, consisting of a polypropylene piece that allows assembly with others, forming a flat macro-structure that has alveoli where the root balls of the plants can be inserted.

The complexity lies in achieving good buoyancy, so that the macro-structure formed is resistant and at the same time flexible enough to withstand the movements of water, wind and the stresses that are generated when the plants grow.

To fulfill all the intended functions of the macro-structure, it will be equipped with expanders to increase buoyancy, and hinge-type joints that will give it flexibility.

The slopes of the pond are 3H: 2V and the buffer zone will be 0.3 meters. Once the excavation is done, a 15 cm layer of selected soil will be spread and compacted at the bottom of the pond. A geotextile sheet and a HDPE sheet will then be installed.

The bottom will have a 1% slope so that, at the lowest point, we will place the sludge outlet / drain pipe.

For the inlet water conduction, a pipe is foreseen, which will ramify into several, for a better distribution and homogeneity of the inlet discharge into the pond.

As regards the outlet of the discharge from the pond, a collection channel will be installed with a Thomson weir.

The treated discharge of this pond will be piped to the next element of the installation, which is the water distribution chamber to the 3 aerobic ATH ponds.

Finally, the floating polypropylene structure will be assembled, inserting the plants in the holes provided for that purpose, at a ratio of 14 plants / m^2.

The anaerobic sludge generated will be deposited at the bottom of the pond, and it will be necessary to remove it from time to time (it is expected that every 3.5 years will have to be removed from this pond about 53.7 m3 of sludge, the first removal being after 7 years). A chamber with a set of two gate valves has been foreseen to either direct the sludge through a pipe to a second chamber from which the tank truck with a pump will suck the sludge when appropriate, or, with the second valve, to be able to empty the pond in case of need for maintenance.

The second stage of the process is the **aerobic stage**. Once the anaerobic stage has been completed, the discharge will be piped underground to a distribution chamber

The function of this chamber is to be able to distribute the water effectively and equally to the three ponds that comprise this aerobic phase of the process.

Entering a first chamber that is connected through weirs to three contiguous chambers, each of which is connected with one of the three aerobic ponds of this phase.

Given that the distances between the three ponds and the distribution chamber are not equal, a 0.3 x 0.3 meter deep manual adjustable penstock sluice gate valve has been provided in each distribution chamber so that, during the start-up phase, it is possible to regulate the flow in such a way that the three ponds treat the same flow. These valves can also be used to isolate any of the ponds if required for maintenance work.

Each of the three measures approximately 14.25 x 42.75 meter. The useful depth will be 1.5 meters.

In these ponds the phenomenon of denitrification and the biological removal of part of phosphorus from the influent water occurs.

A plant sieve and a macro-structure, both with identical characteristics to those described in the previous phase, complete the equipment of the ponds.

This second phase is the aerobic stage, where the plant sieve transfers the oxygen captured by the aerial part of the plants to their roots, and from these the oxygen is transferred to the aquatic environment creating the necessary support for development of aerobic bacteria.

These bacteria degrade organic matter, by having sufficient oxygen, serving as nutrients to the plants themselves.

Therefore, the process is natural, sustainable and without energy expenditure.

At the end of each of the ponds, the clarified water is collected in a channel arranged at the end of the pond, which is led to a common pit from which it passes to the next phase of the process.

The third stage of the process is **the refining** raft.

Once the aerobic stage is completed, the discharge will be conducted to an inlet chamber to the CTH-type pond.

The function of this chamber is to unify the discharges from the three previous ponds, in such a way that the water from each aerobic pond enters to a pit with weirs, which connects with a common chamber with a penstock from which the single conduit leaves towards the refining pond.

In this pond, the water level is only 0.5 m, so the water is forced to pass through the sieve that forms the radicle of the plants, avoiding any possibility of a short circuit or preferential flow.

In the next chamber of the process, two submersible pumps will be installed (one standby pump) that will allow the recirculation of sludge or the emptying of one of the ponds at the head of the WWTP.

All necessary networks for the operation of the WWTP have also been arranged in the facility: water, sludge, rainwater, irrigation, control and observation, power and lighting.

4. SIZING OF THE FLOATING WETLAND WITH A HELOPHYTE SIEVE:

The sizing model has been developed by Quarq Enterprise SA with the GEMMA group, within the framework of the Biotechnology Research Project for the treatment of urban wastewater through phytodepuration.

The piston flow model with dispersion including a first-order reactive term under open--open conditions has been experimentally determined as a theoretical model closest to the real operating conditions.

Therefore, the analytical solution used is the equation proposed by Wehner and Wilhelm.

$$\frac{C'}{C'_0} = \frac{4a\exp(\frac{1}{2\mathcal{D}})}{(1+a)^2\exp(\frac{a}{2\mathcal{D}}) - (1-a)^2\exp(-\frac{a}{2\mathcal{D}})}$$

Dónde:

– C_0 es la concentración de DBO a la entrada

– C es la concentración de DBO a la salida

– $\mathcal{D}$ es el número de dispersión dependiente de la relación de forma del sistema y obtenido experimentalmente

$$a = (1 + 4kt\mathcal{D})^{0,5}$$

Where:

– C_0 is the concentration of BOD at the inlet

– C is the BOD concentration at the outlet

– $\mathcal{D}$ is the experimentally derived dispersion number depending on the shape ratio of the system

$$a = (1 + 4kt\mathcal{D})^{0,5}$$

Dónde:

– t es el tiempo de retención hidráulico del sistema dependiente del volumen de la balsa y del caudal del efluente

– K es la constante cinética de primer orden para remoción de DBO en sistemas de helofitas en $k_T = k_{20}(\theta)^{(T-20^\circ)}$ nte de la temperatura de forma que:

Where:

– t is the hydraulic retention time of the system depending on the volume of the basin and the effluent flow

– K is the first order kinetic constant for BOD removal in floating helophyte systems, temperature dependent as follows:

$$k_T = k_{20}(\theta)^{(T-20^\circ)}$$

Where:

– T is the average water temperature of the coldest month in °C

– K20 is the first order kinetic constant for BOD removal in floating helophyte systems at 20°C, obtained experimentally for floating helophyte systems and corrected by application of the sizing model to real operating floating helophyte WWTP data. This constant has been empirically calculated by Quarq Enterprise for each depth of floating helophytes pond

– ϴ= (empirically validated constant)

The project complies with the requirements of the State General Contracting Regulations and covers a complete work that can be handed over for general use, ensuring compliance with the current regulations at the time of drafting the project.

URBAN WASTEWATER TREATMENT ASSESMENT USING A CONSTRUCTED WETLAND WITH *SCHOENOPLECTUSCALIFORNICUS*

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Resumen
Constructed wetlands are suitable systems for the treatment of urban wastewater, and are capable of generating good water treatment results, especially for small communities. In this sense, this study focused on the removal efficiency evaluation of contaminants from a synthetic urban wastewater using a subsurface flow artificial wetland system with the macrophyte *Schoenoplectuscalifornicus* (HA), comparing it with a control system without macrophytes (HC) under different hydraulic retention times (HRT = 7, 5 and 2.4 d). Various physical and chemical parameters of wastewater were evaluated such as COD, ammonia nitrogen, orthophosphates and solids. Both systems presented results that demonstrate their ability to remove COD. Thus, the HA system was the one that presented the best results with average removal efficiencies of 79 ± 14% with an HRT of 7 d. On the other hand, the orthophosphates removal was significantly higher in HA where the highest average values (83 ± 11) were also registered with an HRT of 7 d. Similarly, nitrogen removal remained at an average of 59 ± 22% in HA, representing a significantly better performance than that observed in HC, which suggests that the presence of

macrophyte greatly promotes the elimination of this nutrient. Finally, the removal of total suspended solids (TSS) in HA remained on average above 80% while HC remained below 30%, demonstrating that the presence of macrophyte roots contributes to wastewater solids filtration.

INTRODUCTION

Wastewater treatment in Paraguay is a pending task since only 11% of the population has sewage collection system, and only 2% of the collected wastewater is effectively treated through a primary or secondary system (Ministerio de Obras Públicas y Comunicaciones - MOPC, 2018). Most of the population has improved sanitation systems, which consist mainly of septic tank coupled to an infiltration well. These systems currently constitutea great threat to underground water sources, which supplies more than 50% of the country's population. This have been evidenced in studies carried out by Arrabal &Álvarez (2017), who showed that high levels of underground water contamination could be detected by nitrates. In this sense, it is necessary to look for low-cost and easy-to-operate alternative systems, for their implementation in communities where conventional wastewater treatment plants cannot be accessed due to their high cost. Therefore, the use of artificial wetlands emerges as alternative systems that can be considered as a good option over conventional systems (Serrano & Corzo Hernández, 2008). Several studies on constructed wetlands have demonstrated their efficiency and robustness for the purification of urban wastewater and in some cases industrial wastewater (Vymazal, 2014). Therefore, this work focused on the evaluation of the performance of an artificial wetland with the macrophyte *Schoenoplectuscalifornicus* for the treatment of urban wastewater under different hydraulic retention times, comparing it with a control system without vegetation.

METHODOLOGY

Pilot system operation

The ponds were built in fiberglass with the following dimensions: 1.2 m long, 0.8 m wide and 0.45 m deep. Gravel of ¼ inch was used as substrate, with a depth of approximately 30 cm. The control system (HC) was filled with substrate only. On the other hand, in the vegetated system (HA), in addition to the substrate, individuals of *Schoenoplectuscalifornicus* were planted sito an approximate distance of 30 cm, the plants were collected from a wetland in a city near the installation site of the pilot systems (Figure 1). After acclimatization of 45 days, the systems were operated at constant flow rate with horizontal subsurface flow for a period of 125 days, in which they worked under 3 different theoretical hydraulic retention times (HRT) (7 d, 5 d and 2.4 d) to evaluate the effect of the load on the performance of the system with vegetation and without vegetation.

Figure 1. Set and started up of the pilot scale wetlands HA y HC

Synthetic wastewater and physico-chemical parameters

In view of the current sanitary conditions, we worked with a synthetic wastewater with characteristics similar to urban wastewater, whose components were previously reported (Chuang et al., 1998). The characteristics of the influent wastewater can be seen in Table 1. The systems' performance was evaluated based on the weekly measurement of chemical oxygen demand (COD), total suspended solids (TSS), orthophosphates, ammonia nitrogen, dissolved oxygen, and pH of the influent and effluent. All parameters were analyzed following standardized methods (APHA 2005). Based on the results, the removal efficiency (%) and the removal capacity (g/m^2.d) of the systems were calculated. In order to identify significant differences between the systems, under the three HRTs, the results of pollutant removal efficiency observed in the systems were analyzed through an ANOVA, and pair comparison by Tukey's post-hoc test. To do this, Shapiro-Wilk tests were carried out to establish the normality of the data. In cases where normality was not evidenced, the non-parametric Kruskal-Wallis ANOVA was performed. In all cases, a significance level of 0.05 was considered.

Table 1. *Synthetic wastewater characteristics*

Parameters	Concentration (mg/L)
COD	209,9 ± 113,5
Orthophosphate	3,9 ± 1,9
Ammonia N	20,6 ± 7,3
TSS	70 ± 39,6
pH	7 ± 0,9
Temperature	22,7 ± 3

RESULTS AND DISCUSSION

The evolution of the COD removal efficiency of both systems is presented in Figure 2, where it can be seen that the highest efficiency values were obtained with an HRT of 7 d with performances of 79 ± 14% for HA and 55 ± 25 % for HC. These performances resulted in elimination capacity of 6 ± 2 and 4 ± 2 $g/m^2.d$, for HA and HC, respectively (Table 2). With the HRT reduction to 5 d, no substantial changes in removal performance were observed due to the high variability of the data, however, the observed removal efficiency in HA decreased to values of 60 ± 23%. Finally, the results observed in the TRH of 2.4 d showed elimination efficiency levels of 48 ± 23 and 41 ± 29% for HA and HC, respectively. These differences were not significant ($p> 0.05$) due to the high variability of the data (Table 2). However, the removal capacity of the systems increased with the decrease in HRT due to the higher load supplied to the system. In a study conducted by Ilyas and Masih (2017), COD removal efficiency values (69-88%) similar to those observed in this study were reported. On the other hand, another study has reported removal efficiencies of between 83 and 92% with removal capacity of 14 to 16 $g/m^2.d$ for systems of similar dimensions to the one operated in this investigation (approximately 242 L) and low a theoretical HRT of 6 d, the macrophytes used were *Cyperus ligularis* and *Echinochloacolonum* (Charris & Caselles-Osorio, 2016).

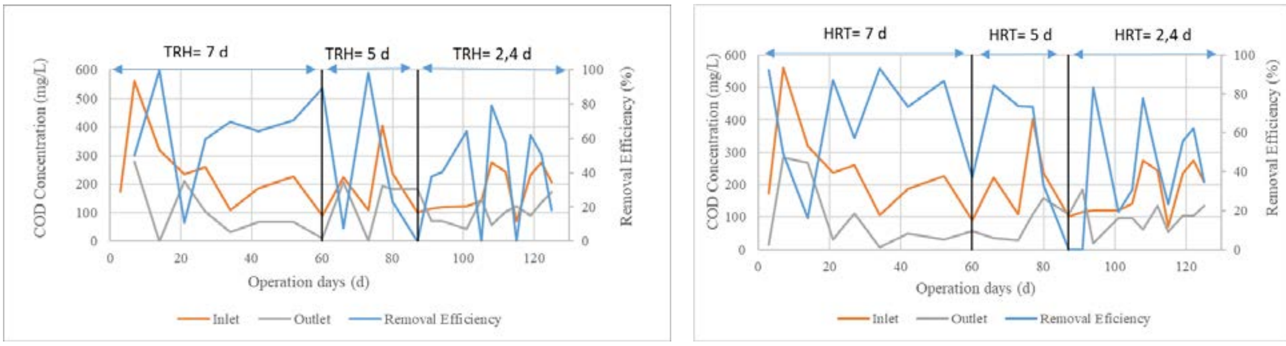

Figure 2. *Temporal variation of inlet and outlet COD concentrations and the removal efficiency of both HA (a) and HC (b) systems.*

Orthophosphate removal was significantly higher ($p <0.05$) in HA with a HRT of 7 d where an average of 83 ± 11% was reported, however, this was decreasing and remained close to 50% in both HRT subsequently evaluated (Table 2). On the other hand, the removal of orthophosphates in the HC remained on average below 50% in most of the analysis period regardless of the HRT, which demonstrates the effect of vegetation on a better performance of the system ammonia nitrogen removal.

Similarly, ammonia nitrogen removal in HA (59 ± 22%) remained well above that observed in HC (12 ± 17%) for the HRT of 5 d. While, with aHRT of 2.4 d, both systems demonstrated a low removal efficiency of 10 ± 22 and 9 ± 15% for HA and HC respectively. The greater performance observed in the vegetated system may be due to the very presence of macrophytes that promote the growth of aerobic microorganisms which carried out the oxidation of ammonium by nitrification, converting it to nitrite and nitrate, or by assimilation of the plants. The study conducted by Charris and Caselles-Osorio (2016) under similar conditions, showed ammonia nitrogen removal efficiencies of 49 to 61%. On the other hand, more favorable results of ammonia nitrogen removal were reported (73 - 89%) in the study developed by Romero-Aguilar and coworkers in 2009, where a pilot subsuperficial constructed wetland with *Phragmites australis* y *Typha dominguensis*was operated at a 5 d HRT.

The removal of suspended solids in the vegetated system was also significantly better than that observed in the control system (p<0,05). In the HA, the average removal remained above 80%, while in the HC the average removal efficiency remained between 2 ± 5 and 27 ± 43% (Table 2). This indicates the high influence of macrophyte roots on the retention of solid materials that enter the system. In fact, on several occasions it was possible to detect TSS levels of the effluent above the concentration of the influent in the HC system, which indicates that part of the biofilm was detached more easily in this system.

Table 2. *Performance assessment results of HA y HC**

Removal efficiency and capacity of pollutants HA		HRT: 7 días		HRT: 5 días		HRT: 2,4 días	
		HC	HA	HC	HA	HC	
COD	%	79 ± 14 (a)	55 ± 25 (a)	60 ± 23 (a)	60 ± 40 (a)	48 ± 23 (a)	41 ± 29 (a)
	g/m².d	6 ± 2	4 ± 2	7 ± 6	5 ± 4	11 ± 7	10 ± 8
Orthophosphates	%	83 ± 11 (c)	46 ± 16 (b)	49 ± 28 (b)	8 ± 18 (a)	53 ± 17 (b)	48 ± 13 (ab)
	g/m².d	0,103 ± 0,04	0,55 ± 0,02	0,061 ± 0,04	0,014 ± 0,03	0,076 ± 0,03	0,058 ± 0,03
TSS	%	77 ± 29 (b)	23 ± 39 (a)	87 ± 13 (b)	38 ± 48 (ab)	83 ± 19 (b)	2 ± 4 (a)
	g/m².d						
Ammonia-N	%	-	-	59 ± 22 (b)	12 ± 17 (a)	10 ± 22 (ab)	9 ± 15 (a)
	g/m².d	-	-	0,484 ± 0,19	0,109 ± 0,14	0,131 ± 0,31	0,085 ± 0,17

**HA: Vegetated wetland, HC: system control. Results presented by means and standard deviations; the values with a letter in common are not significantly different (p > 0,05).*

CONCLUSIONS

Constructed wetland systems are an effective and robust technology for urban wastewater treatment. The results of elimination of organic contaminants and nutrients observed in this study reinforce this assertion. In the same way, the performance in the removal of nutrients and solids can be affected by the presence or not of the macrophytes, since the results have shown that the direct or indirect action of the vegetation collaborates for the elimination of ammonia nitrogen, orthophosphates and suspended solids from wastewater.

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EFFECT OF CHANGING THE FEEDING/ RESTING CYCLE IN MODIFIED VERTICAL WETLANDS FOR DOMESTIC WASTEWATER TREATMENT

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Abstract

The aim of this work was to evaluate the behavior of a modified (height, media and saturation)vertical subsurface flow treatment wetland(VSSF-TW) system, when changes are made to the feeding / resting cycle.For this, three experimental units of VSSF-TW of 0.6 m in height were built.The systems were operated in two phases of three months each: a) phase II, each VSSF-TW operated for 5 d and resting for 10 d, using three experimental units, and b) phase III, each VSSF-TW operated for 3.5 d and resting for 3.5 d, using two experimental units. In both experimental phases, a bottom layer of natural zeolite was saturated. Water quality and plant development of*Schoenoplectuscalifornicus*were evaluated. The results showed that there is similarity in the quality of the effluent water, highlighting the results of solids, organic matter, ammonium and phosphate. However, the feeding/resting cycle changes negatively affected total ni-

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trogen removal. For its part, plant growth was more positively influenced by the operating season than by the change in the feeding / resting cycle. Thus, the results of this work suggest that the modification of feeding / resting cycles in modified vertical wetlands is feasible, but the elimination of total nitrogen is not improved.

INTRODUCTION

Vertical Subsurface Flow Treatment Wetlands (VSSF-TWs) with intermittent application (most common mode of operation), have shown to be effective in removing solids (measured as total suspended solids (TSS)) and organic matter (measured as chemical demand oxygen (COD)), with removals between 35 and 95%, when they are used for the treatment of domestic wastewater (Stefanakis *et al.*, 2014). However, the removal of total nitrogen (TN) and phosphorus is still limited, and regularly less than 60% (Lana *et al.*, 2013). Total Nitrogen removal in VSSF-TWs is limited by denitrification process due to lack of anoxic conditions and organic matter sources (Sánchez *et al.*, 2020). To overcome this limitation, the addition of an external carbon sources or the modifications in the operation, such as variations in the feeding strategy, or the partial saturation of the bottom have been proposed (Sánchez *et al.*, 2020; Silveira *et al.*, 2015). For the feeding strategy, characterized by the variation of daily distribution of the applied hydraulic load (regularly between 4 and 12 pulses / day(Brix y Arias, 2005),for a single bedduring the resting periodsacross the day, the bed is recharged with O_2, (bed without resting periods of days). Another option is the use of several beds and alternate feeding of the beds (which allow resting periods of days).Modifications in feeding / resting, is sought to improve oxygen transfer rates and to improve the aerobic degradation of organic matter, nitrogen, and prevent the phenomenon of clogging (Stefanakis *et al.*, 2014).On the other hand, the partial saturation of the bed depth of the VSSF-TW has shown to improve the elimination of TN (Pelissari*et al.*, 2017), and assumes that the use of this operating strategy, in combination with the use of special media with adsorption capacity of ammonium and especially phosphorus (particularly phosphate), such as natural zeolites (Andrés *et al.*, 2018),could increase the capacity of elimination of these parameters in a VSSF-TW.On the other hand, traditionally the beds of VSSF-TWs have depths that are close to one meter in height (1.0 m) (Brix y Arias, 2005). However, it has been indicated that the main bacterial activity for the removal of contaminants occurs in the first 0.25 m depth of the bed (measured from top to bottom) (Arias *et al.*, 2013). Thus, heights between 0.08 and 0.8 m have been used and studied (Taniguchi *et al.*, 2009), which would indicate their feasibility of modification.

Taking into account the above, the aim of this work is to evaluate the behavior of a modified (height, media and bottom saturation) VSSF-TWs system, when changes to the feeding / resting cycle are made. All with a view to optimizing its use and adaptation to local conditions.

MATERIALS ANDMETHODS

1. *Experimental units*. Three VSSF-TWs (understood as a treatment system) were implemented in the NBS Laboratory in the city of Talca (Central Valley, Maule Region, Chile). All the VSSF-TWs experimental units were built with PVC pipes (Ø: 0.2 m) with dimensions and materials shown in Figure 1.Sand (Ø: 0.08-5.0 mm) and Zeolite (Ø: 3-5 mm) were used as filling material, and gravel (Ø: 5-19 mm) was used in the upper and lower part. *Schoenoplectuscalifornicus* was used as a plant species.

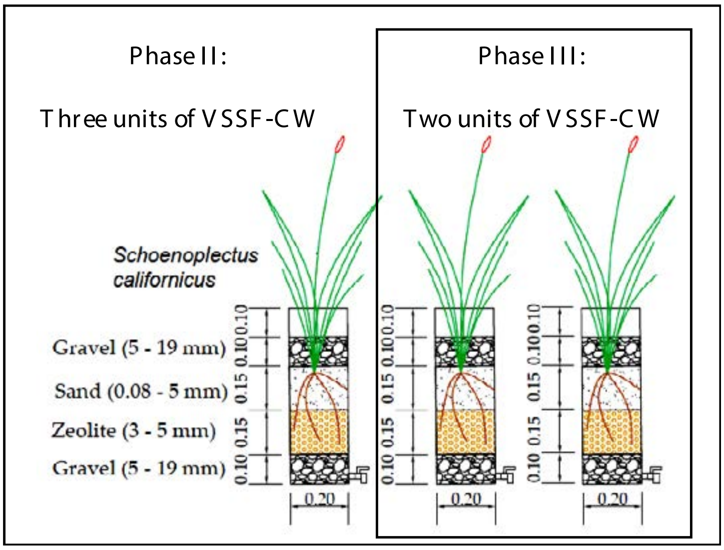

Figure 1. *Schematic of the experimental setup. Measurements in meters, except where otherwise indicated.*

2. *Operation and monitoring strategy*. The VSSF-TWs were fed with effluent domestic wastewater from a septic tank serving a single household of six inhabitants. The VSSF-TWs were operated for three months prior to evaluate other design parameters (Phase I) (Vera-Puerto *et al.*, 2021). After this, the operation was carried out for a period of six months. The Hydraulic Loading Rate (HLR) applied was 0.12 m / d. This HLR represents the mean value between 80 and 160 mm/d defined by Vera-Puerto *et al*. (2021) as representative for Chile. The application of this HLR was carried out by performing 12 pulses / day (Brix and Arias, 2005; Stefanakis *et al.,* 2014). An application strategy was defined with feeding / resting periods of three months each, being: a) Phase II, each VSSF-TW fed for 5 d and resting for 10 d, using three experimental units and b) Phase III, each VSSF-TW fedfor 3.5 d and resting for 3.5 d, using two experimental units. In both experimental phases the bottom zeolite layer remained completely saturated. The samples to analyze the water quality were taken every two weeks. The sampling points were: a) influent and b) effluent. Parameters evaluated include pH, Temperature (T), Electrical Conductivity (EC), Oxidation Reduction Potential (ORP), COD, TSS, Ammonium (NH_4^+), Nitrate (NO_3^-), Total Nitrogen (TN) andPhosphate (PO_4^{-3}).

The samples were analyzed in the Water Quality Laboratory of the Universidad Católica del Maule. To evaluate the adaptation of the plants, growth measurements (height and number of leaves), as well as chlorophyll in the leaves, were measured weekly.

3. *Analytical methodologies.*pH, T, ECand ORP were measured with multiparameter field equipmentHI 98194. For the remaining parameters, the samples were filtered (glass fiber filter, 0.7 mm pore size). COD, NH_4^+, NO_3^-, TNand PO_4^{-3}were measured photometrically with Multiparametric Photometer Hanna HI 83399 using reagent kits for reading. These determinations are simplified procedures based on procedures described in APHA-AWWA-WEF (2017). The height measurements of the plants were made using a tape measure, and the number of leaves was carried out by manual counting. Chlorophyll was measured with Konica Minolta SPAD-502 Plus.

RESULTS AND DISCUSSIONS

1. *Performance of water quality.* Table 1 shows a summary of the influent and effluent water quality results. Regarding the influent, the values in Table 1 are consistent with that reported for domestic wastewater in Chile (Vera-Puerto *et al.*, 2021).

Table 1. *Characteristics by phase of operation of influent domestic wastewater and the effluent of the VSSF-TWs system. n=6*

Parameter	Unit	Influent		Effluent to VSSF-TWs System	
		Phase II	Phase III	Phase II	Phase III
pH	Unit	7.9±0.1	7.8±0.3	7.0±0.1	7.1±0.3
T	°C	10.7±1.7	16.4±2.7	10.6±1.5	17.1±2.6
EC	μS/cm	1002.0±103.4	746.0±180.2	805.3±82.2	780.8±61.7
ORP	mV	-133.2±135.5	-272.6±66.5	+195.2±71.4	+160.8±107.1
COD	mg/L	209.1±70.7	90.3 ± 44.2	74.5 ± 28.4	30.2±16.5
TSS	mg/L	194.0±225.2	78.9±79.2	6.5 ± 3.7	3.1±1.2
NH_4^+	mg/L	56.4±6.8	34.7±18.2	3.0±3.5	0.6±0.2
NO_3^-	mg/L	4.3±1,9	3.4±0.0	86.2±79.1	50.8±17.9
TN	mg/L	46.1±9.0	28.8±17.0	15.5 ± 13.6	36.7±12.9
PO_4^{-3}	mg/L	20.8±4.9	15.3±6.0	14.4±3.8	9.1±7.1

In the effluents, the pH remained neutral at around 7.0; the water temperature increases more than 5 °C in Phase III, which is expected since thisphase wascarried out during the austral spring, where average temperatures progressively increase from 8 to 15 °C, but maximum temperatures were above 20 °C. The EC remains stable during the two phases of operation with values below 1 000 μS / cm.These results indicate that there is no modification of these macro-parameters when modifying the feeding / resting cycle. Regarding the COD effluent concentration, Phase III presents lower mean values, which are less than 50 mg/L. Despite the above, the two phases present average removal efficiencies above 60%. Regarding the TSS, the two phases show good performance, with average values below 10 mg/L. Again, these results indicate that there is no modification of COD and TSS effluent concentrations when modifying the feeding/resting cycle.Additionally, Table 1 shows that there was a similar transformation from ammonium to nitrate for the two phases, since ammonium is reduced to nitrate by more than 90%, but in Phase III, less nitrate is generated. In turn, the TN for Phase III presented mean effluent concentrations with values similar to the initial concentration. This could indicate that in Phase III, as the feeding/resting frequency increases, and despite the saturation of the bed depth, the necessary conditions for denitrification are modified, affecting the effluent quality, since in Phase II at least 50% removal of TN was achieved. Finally, the performance of phosphate presentremovals above 20% in the two phases, showing that the saturation of the zeolite keeps the removal stable despite the modification in the feeding/resting cycle.

2. *Plant growth.*According to the growth data, the *SchoenoplectusCalifornicus* individuals increased in size from 0.9 m, stable during Phase II, to 2.0 m (average) at the end of Phase III. Furthermore, the number of leaves increased from 15 leaves (average) in Phase II, to over 20 leaves in Phase III. This growth is consistent with the experimentation season of Phase III: spring, and it was not affected by the modification of the feeding / resting cycle. Furthermore, the chlorophyll results show stable values in Phase II and III: between 70 and 80 SPAD units. Despite the above, the results of this work suggest that there is no contribution of plant growth to the nutrients removals: nitrogen and phosphorus.

CONCLUSIONS

The effluent water quality results pH, T, EC and ORP, as well asTSS, COD, $NH_4^+$and PO_4^{-3}, showed that modifying the feeding / resting cycle does not affect their concentration. However, the situation changes with the TN, a parameter that was negatively affected by this modification. In addition, the parameters taken to evaluate the adaptation of the plants show that *Schoenoplectuscalifornicus* has a positive growth as expected responding to the seasons but apparently not affected by the treatment of the wastewater and not related to being a response to the modification of the feeding / resting cycle. However, being plants, it is recommended to follow the evolution of this growth for a longer time. Thus, the results of this work would show that the feeding / resting cycle can be reduced, reducing the number of vertical wetland beds, but the water quality with respect to the TN would be affected.

ACKNOWLEDGEMENTS.

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FEASIBILITY ANALYSIS OF THE UTILIZATION OF CONSTRUCTED WETLANDS IN SMALL HIGH- ANDEAN URBAN AGGLOMERATIONS

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Abstract

This article shows the results of bibliographic and documentary research and an in-situ visit to the treatment plants of small populations around the circunlacustrine ring of LakeTiticaca;with the aim of carrying out a feasibility analysis of the utilization and exploitation of artificial wetlands (wetlands) as an alternative for the domestic wastewater treatment of small urban agglomerations in high-Andean areas. From the visit to the thirteen localities and fourteen treatment plants, it was observed that in six of them, they just have stabilization ponds, and without maintenance, and eight have biological filter treatments and from these, just three have some maintenance, that is why the great majority is practically abandoned. In addition, they do not have an energy supply and they have sloping orographic conditions, as far as there is a great potential of eleven treatment plants, for the extensive technologies implementation (artificial wetlands), thereby reducing the discharge of wastewater without any treatment to the Lake Titicaca.

INTRODUCTION

Constructed Wetlands (CW) are man-made wetlands that treat organic, inorganic pollutants and excess nutrient in surface water, municipal wastewater, domestic wastewater, refinery effluent, uranium acid mine drainage, or landfill leachate (De Filippis, 2015), using biological processes that involve wetland vegetation, soils and their associated microbiota (Milledge, Thompson, Sauvêtre, Schroeder, & Harvey, 2019). The CW for wastewater treatment can be classified according to the dominant vascular plant life forms and the water flow regime (Figure 1) (Blau, 2002). CW can still be classified according to the dominant macrophyte life form into floating macrophytes, floating leaves, emergent, and submerged. CWwith emerging macrophytes is the most widely used system (Vymazal, 2008)

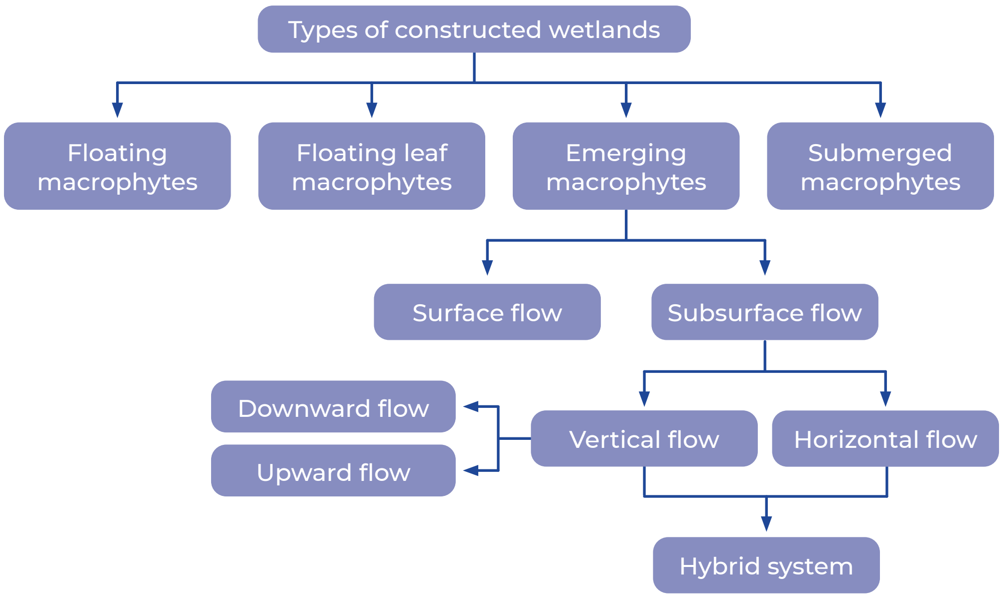

Figure 1. *Constructed wetlands classification for wastewater treatment based on dominant vascular plant life forms and water flow regime.Adapted from (Jan Vymazal, 2001) Types of constructed wetlands for wastewater treatment: their potential for nutrient removal. Transformations of Nutrients in Constructed Natural and Constructed Wetlands.*

Surface flow systems are those where water preferentially circulates through the plants' stems and are directly exposed to the atmosphere (Figure 2). This type of wetlands is a modification to the conventional lagoon system. Unlike these, they are shallower (no more than 0.6 m) and have plants (Delgadillo, Camacho, & Andrade, 2010).

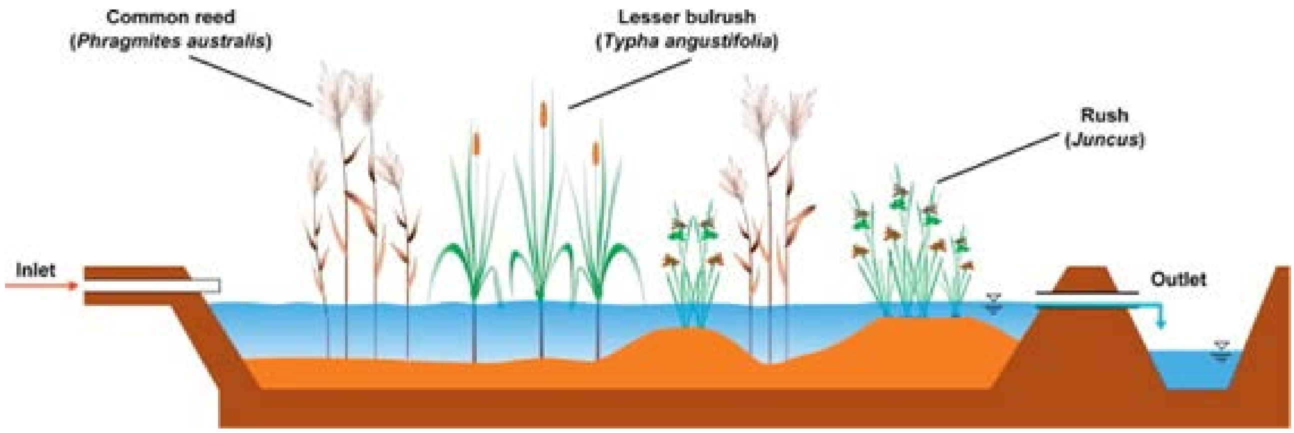

Figure 2. *Surface flow artificial wetland of free-water using rooted emergent macrophytes. Source: Raboni, Torretta, &Urbini (2013)*

In constructed subsurface flow wetlands, wastewater flows horizontally (vertically in some applications) through a porous medium consisting of a bed of washed gravel (characteristic gravel dimension: 2 to 10 mm; bed height: 0,60 to 0,90 m) with a topsoil cover (Figure 3). The bottom of the gravel bed slopes slightly (about 1-3%) in the direction of flow to facilitate water flow, while the water level is kept 0.05-0.10 m below the top of the bed. gravel. The incoming wastewater is distributed in the porous medium as evenly as possible. The treatment is based on both physical (filtration and sedimentation) and biochemical principles (Copelli, Raboni, &Urbini Molecular Sciences and Chemical Engineering, 2015).

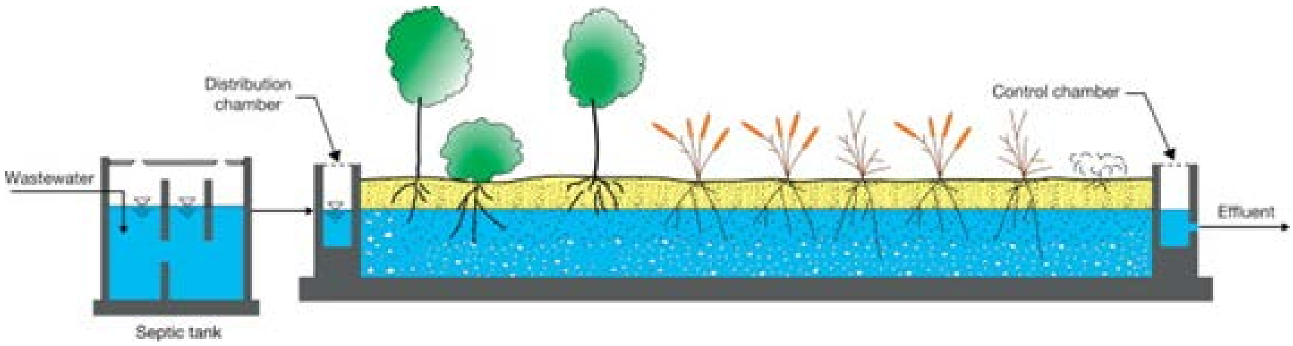

Figure 3. *Artificial subsurface flow wetland; Source: Raboni, Torretta, &Urbini (2013)*

Constructed wetland application is more appropriate and beneficial in decentralized wastewater treatment in cities and small rural areas, as well as for slightly polluted waters of rivers and lakes due to the low costs of construction, operation, and maintenance (Li et al., 2018).

Lake Titicaca is the largest freshwater lake in South America and the highest commercially navigable body of water in the world (Global Nature Fund, 2012). This large water mirror is within an endorheic basin and receives effluents from 25 rivers, the principal it is being the Ramis river of 14,700 km^2, on the other hand, Desaguadero river of 29,843 km^2 as an outward flow from the Lake Titicaca (Medrano, Mamani, Muñoz, Díaz, & Medrano, 2020). Unfortunately, the human activities development has generated a little-studied pollution process for some time, but that, without doubts, is causing some type of decrease in the water quality in the basin (rivers and the lake itself), this due to the discharge of its wastewater (Ocola&Laqui, 2017).Because of the water contamination of the Titicaca basin, the Peruvian state with the help of private investment will launch the execution of the design, construction, expansion, rehabilitation, operation, and maintenance of six new treatment plants and the operation and maintenance of four plants already installed in the Titicaca basin (ProInversión, 2018). These plants do not consider small urban agglomerations, whereof it is necessary the intervention through technologies that do not require more investment from the state. That is why, the review and updating of the technology called “artificial wetlands” were seen pertinent to be applied in small urban agglomerations near Lake Titicaca, in addition to identifying the potentialities of its implementation in conditions such as those found in the highlands of Puno.

METHODOLOGY

The research has a quantitative approach, inductive method, non-experimental design. It is transectional. Basic investigation. Descriptive. The population is the same as the sample. It was done next Protocol where the following activities are carried out: First phase: Documentary compilation, (1) Investigations carried out that demonstrate the pollution of Lake Titicaca (2) Investigations on Wastewater Treatment Plants. (3) Coordination with authorities for technical field visits and interviews with Operators / Technicians in charge of the Domestic Wastewater Treatment Plants (DWWTP). (4) Preparation of the instrument, formulary. (5) List of Plants. (6) Control sheets of the number of plants visited. (7) Daily tour schedule. (8) Record of photos and filming of the state of the PTARD. The population is the 14 Domestic Wastewater Treatment Plants, and the sample is equal to the population, as it requires the investigation of the total universe, in order to, individually specify the problems of each one of the DWWTP.

RESULTS AND DISCUSSION

The obtained results are the summary of the in-situ observation of the fourteen treatment plants belonging to thirteen localities (Huata, Capachica, Samán, Taraco, Chupa, Zepita, Juli, Acora, Laraqueri, Chucuito, Pomata, Tilali, and Conima), located in the circunlacustrine ring of Lake Titicaca and they are outside the project of “Wastewater Treatment System PTAR Titicaca”.

In the access to the treatment plants, it is observed that 9 of them are near the urban area, so they can be accessed on foot and 5 of them require the use of a vehicle as they are far from the urban area.

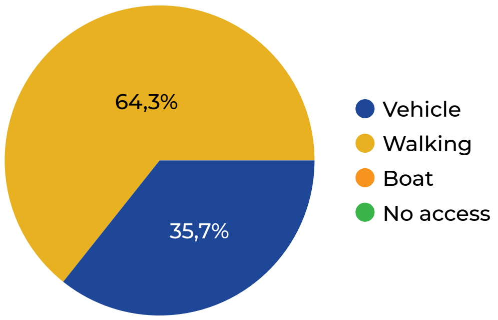

Figure 4. ***Access to treatment plants***

In the treatment system that they have, it was observed that 8 treatment plants have small biological filter plants, and 6 only have stabilization ponds.

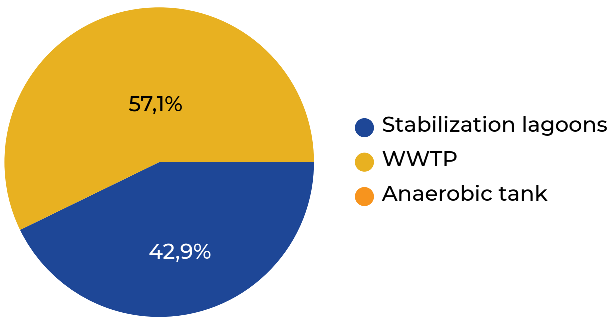

Figure 5. ***Current treatment system***

For the current conditions of the treatment plants, 3 were valued as “good”, 3 as “regular”, 6 as “deficient” and 2 as “inoperative”. It evidenced to more than 79.6% require immediate intervention.

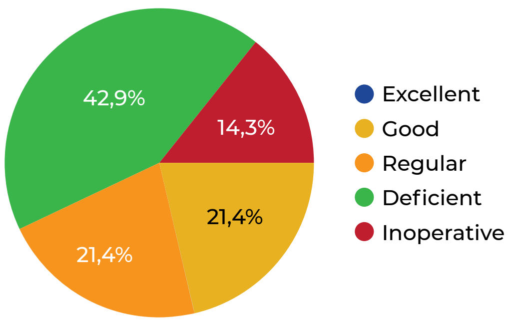

Figure 6. ***Conditions of the treatment system***

The final discharge after the treatment of these waters, in 12 they are discharged to bodies of water such as rivers and lakes, and only in 2, the discharge is on the ground.

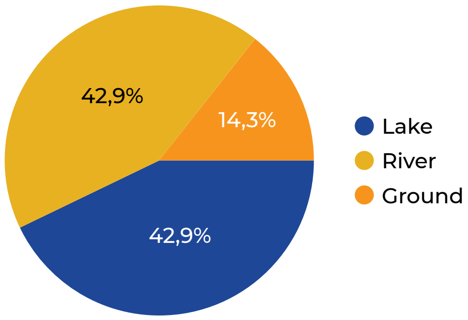

Figure 7. ***Final discharge of wastewater.***

Regarding the orography where the treatment plants are located, in 10 of them, they were valued as having a good and regular slope, and only in 4 as unfavorable.

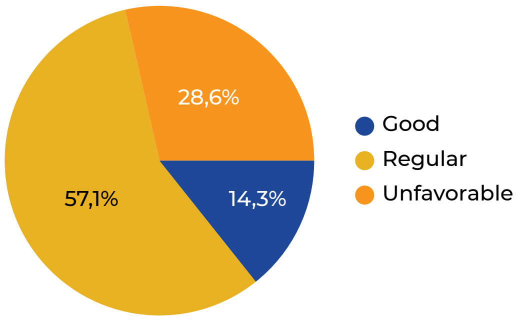

Figure 8. ***The orography of the location of the treatment plants***

In 10 treatment plants they do not have an energy supply, and only in 4 treatment plants have an energy supply. All shows that implement conventional treatment plants, would have greater difficulty.

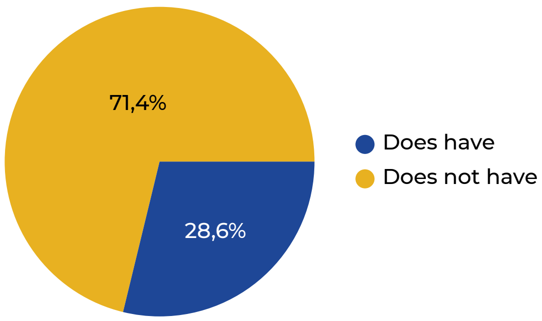

Figure 9. ***The orography of the location of the treatment plants.***

According to these results observed in these visits, they show us that it very likely the implementation of technologies such as "artificial wetlands", since artificial wetlands are an eco-technological alternative for the treatment of all types of water. They are designed to overcome the disadvantages of the natural wetlands and enhance their qualities since they can treat several types of pollutants simultaneously to satisfactory levels compared to other conventional treatment systems (Idris et al, 2010; García and Corzo, 2008)

Even though there are many discrepancies regarding thewetlands functioning of different aspects: dimensioning, operation, etc., including the purification capacity of the implanted species. Several authors affirm that the obtained results with the use of photosystems are better than those obtained with the same system, but without plants.Stearman et al. (2003) report a reduction of pollutants in general, furthermore 20% in implanted wetlands compared to non-implanted ones. Additionally, the purifying potential of wetlands varies seasonally. This can be explained both by simple meteorological variations and by changes in the physiology of plants. Soto et al. (1999) report that the activity of the plant increases in summer, therefore its capacity to treat various factors (removal of phosphorus, phosphates, and nitrogen). Zúñiga et al. (2004) also reported a strong variation in the removalof ammonia and phosphorus between spring and winter, with better results in planted wetlands than in wetlands without plants.

CONCLUSIONS

This investigation identified 13 localities with 14 wastewater treatment systems, located in the circunlacustrine ring of Lake Titicaca, who are not considered in the Project of "Wastewater Treatment System PTAR Titicaca". localities whose water treatment systems wastewater shows deficiency because only 21.4% have adequate operating conditions for wastewater treatment, and more than 78% of the identified wastewater treatment systems need of immediate intervention; likewise, it was identified that the 86% of the final discharge from wastewater treatment reaches to the Lake Titicaca, directly or indirectly. Therefore, it is concluded that 11 of the 14 Wastewater Treatment Systems can implement Artificial Wetland Systems for Wastewater Treatment.

Wetlands are an alternative for the treatment of domestic wastewater for small urban agglomerations, where economic conditions are decisive when launching the type of wastewater treatment, therefore the design models must be adapted to local conditions and analyze the behavior of the different design and operating factors involved, among the most outstanding, are temperature, slope, terrain, vegetation.

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THE VERSATILITY OF CONSTRUCTED WETLANDS FOR WASTEWATER TREATMENT

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Abstract

Constructed wetlands are well known sustainable wastewater treatment solutions that can have a variety of applications. The development of the technology has enabled its application in fields other than wastewater treatment, which can help address current societal challenges. This paper presents two novel applications of constructed wetlands: a merge between vertical treatment wetlands and green walls to treat greywater and the potencial of traditional constructed wetlands to degrade debris from biodegradable packaging.

INTRODUCTION

Constructed wetlands are well known Nature-Based Solutions, acknowledged by their sustainable approach: they require lower raw material consumption when compared to conventional treatment systems (e.g. activated sludge), low operation and maintenance costs and low energy requirements (Matos et al., 2009), while providing additional benefits such as pleasant landscape (Campbell & Ogden, 1999) and wildlife support (Hsu et al., 2011).

CW can be classified in two main types regarding flow: i) surface flow, where wastewater flows along an impermeable basin with macrophyte plants at the surface, emergent or submerged; ii) subsurface systems, where wastewater flows through a porous media, which supports biofilm growth.

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Subsurface flow (horizontal and vertical) is the most widely used type since they provide good removal capacity and prevent contact with wastewater. This last characteristic is especially relevant in warm climates to prevent proliferation of mosquitoes that can act as vectors for several water borne diseases. Although this technology was initially developed to be applied for wastewater treatment in small agglomerations, its applications nowadays extends far beyond it. In the last decades CW types have been widely engineered to allow their adaptation to treat different types of effluents, including upflow systems (Aguirre-Sierra et al., 2020), aerated systems (Maltais-Landry et al., 2009) or floating systems (Headley & Tanner, 2012).

The range of effluents where CW are nowadays applied are also very vast, including industrial effluents (dyes, mining industry (Heiderscheidt et al., 2020; Yadav et al., 2012)), combined sewage overflows (Pisoeiro et al., 2016)organized in two groups (A and B, temporary events (Lakho et al., 2020) and agriculture runoff (Wang et al., 2018), to mention but a few. Several examples can also be identified regarding the coupling between CW and advanced technologies such as a fluidized-bed Fenton coupled with surface and subsurface CW (Xing et al., 2020)the feasibility and safety of papermaking wastewater for the use as ecological water supplement after the treatment by fluidized-bed Fenton (FBF, CW coupled with ozonation (Gomes et al., 2020) or photocatalysis combined with a subsurface flow CW (Chen et al., 2011).

Moreover, the combination with other technologies can also provide benefits for current and future challenges, such as management of water scarcity and degradation of new biodegradable packaging products. The present paper provides an overview of different application of constructed wetlands, focusing on two new applications: green walls for greywater treatment and the degradation of plastic substitutes in traditional CW.

METHODS

Overview

Two case studies were developed to present new applications of constructed wetlands:

- an experimental system of green walls for greywater treatment, which represents the merge between conventional green walls and vertical flow constructed wetlands. This system was irrigated with greywater (i.e. wastewater from shower and sinks), reducing the pressure over potable water resources, while also allowing for greywater treatment and reuse.
- the degradation potential of conventional constructed wetlands in the degradation of biodegradable packaging. The awareness of the effects of plastics and microplastics in the environment is leading to an increased use of plastic substitutes with biodegradable materials. Since their debris can also reach aquatic systems, and CW are very good filters, their retention in CW provides an opportunity for degradation, avoiding a release in the aquatic system.

The green wall for greywater treatment presented in the present paper was based on the master thesis developed by Pissarra (2019) and the degradation of biodegradable packaging was based on the master thesis developed by Fistrale (2020).

Experimental systems

A green wall for greywater treatment was set up at Instituto Superior Técnico, in Lisbon. It was adapted from an existing modular green wall system, to provide three treatment lines with vertical flow. Two of the treatment lines were planted (L1 and L3) and the third one was left unplanted as control (L2). The feeding of the system was made with greywater collected from the sinks at the university. To monitor the green wall treatment capacity treated greywater was collected and analysed regarding dissolved oxygen (OD), pH, electric conductivity, chemical oxygen demand (COD), total suspended solids (TSS) and volatile suspended solids (VSS). Figure 1a) provides a general view of the green wall.

a) b)

Figure 1. *a) view of the green wall for greywater treatment and b) packaging and pieces used in the biodegradation study inside CW.*

To study the degradation capacity of biodegradable packaging particles retained in constructed wetlands, sugarcane based packaging were cut in a set of 132 pieces with 1cm x 1 cm and submerged in an experimental CW (also located at Instituto Superior Técnico), in a full scale constructed wetland located in Barroca D'Alva, Alcochete, Portugal and in an aerated reactor containing river water, to asses different biodegradability environments. Figure 1b) provides a view of the packaging and corresponding pieces used in the study.

The pieces were previously weighted and placed inside nylon socks that acted as a porous container to allow contact with wastewater. At known time intervals a set of pieces was removed from each environment to analyse their degradation state. To remove biofilm formed in each piece samples were cleaned with KOH which acted as a mild oxidizer. After this step samples were dried at 40°C overnight before weighing with an electronic scale.

RESULTS AND DISCUSSION

Green walls for greywater treatment

DO concentrations at the inlet averaged 2.3 mg/l. There was a DO increase from inlet to outlet due to the air contact between pots, and concentration values at the outlet were close to saturation, with an average of 9,4 mg/l on all treatment lines. This situation demonstrates that treatment processes occurred mostly in an aerobic environment. pH showed a slight increase from 7.2 at the inlet to 7.9 the outlet, but without significant differences between treatment lines.

Although TSS inlet values were low, with an average of 19.9 mg/l, removal efficiencies were up to 80% in one of the planted lines, reaching an average outlet concentration of 3.8 mg/l. Most of the solids were biodegradable, as indicated by the high percentage of VSS at the inlet, with an average of 88%.

COD at the inlet averaged 129 mg/l but showed also a large variation with a standard deviation of 115 mg/l. Removal efficiencies averaged 50% for Line 1, 47% for Line 2 and 43% for Line 3 (unplanted).

Degradation of biodegradable packaging

Degradation of sugarcane based packaging in the different environments showed a weight loss of 7% in river water after 25 days. In the experimental constructed wetland the average weight loss was of 9% after 37 days and 24% after 72 days. In the full scale wetland the average weight loss was of 63% after 35 days and 72% after 63 days.

According to EN 13432 standard in order for a material to be considered biodegradable it has to reach a 90% biodegradation in 12 weeks (84 days). Considering the trends calculated for each degradation environment, a 90% biodegradation rate would be achieved in river water after 34 weeks, in the experimental CW after 21 weeks and in the real wetland after 17 weeks. Although the constructed wetland environment did not meet the standard goal, it has the potential for a faster degradation than aquatic environments. Further studies should be conducted in order to optimize this system, since one of the samples collected reached 100% biodegradability in 63 days.

CONCLUSIONS

Constructed wetlands are Nature-Based Solutions that provide a sustainable solution for a wide variety of wastewater treatment types. They can also be engineered to achieve specific goals and contribute to current challenges such as water reuse and biodegradation of new materials than can be retained inside.

The development of green walls for greywater treatment has proved to be a promising technology to provide a source of wastewater for reuse in urban areas.

The biodegradation of biodegradable packaging retained in constructed wetlands also showed the potential of these systems to remove the organic load of new pollutants that may reach water courses.

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ON THE ECOLOGICAL BENEFITS OF USING CONSTRUCTED WETLANDS FOR TREATING WASTEWATER IN SMALL URBAN AREAS IN A MEDITERRANEAN REGION

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Abstract

The performance of different types of constructed wetlands (CWs) managed by the Global Omnium Group in eastern Spain has been monitored. These CWs work either as part of secondary treatment or as polishing tertiary treatment in WWTPs. Based on the regular analytical monitoring, they generally showed a high efficiency in removing organic matter (OM) and inorganic nitrogen compounds. Additional surveys on the biochemical, microbial, and metabolic features were conducted to fully understand the pollutant removal mechanisms. The performance of CWs varied depending on its design and operational modes, but a general achievement was the occurrence of an upgrading of the ecological quality of the treated water consisting in both the production of a biochemically more stable dissolved organic matter and in the assembling of a microbial community that was functionally more suited for the receiving environment. Findings of this study demonstrate that these CWs may serve as transi-

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tional systems to further mimic the water features of the receiving environments. This can buffer the impact of discharges, particularly in water stressed regions as the Mediterranean where these spills can account for an important part of the hydrologic supply of natural environments.

INTRODUCTION

Constructed wetlands (CWs) are a well-established wastewater treatment technology used in small urban agglomerations (<2,000 equivalent inhabitants). They are designed to remove pollutants from wastewater by mimicking the biological processes of natural freshwater ecosystems. CWs allow effective purification with a minimum consumption of electrical power and chemical reagents, as well as low costs of operation and maintenance. They can perform as a secondary treatment or as tertiary polishing step when attached to conventional treatments, which may represent a great benefit for water reclamation purposes (Camacho *et al*, 2018; Peña *et al*, 2019a, 2019b; Rochera *et al*, 2021). They can also provide ecosystem services such as the maintenance of biodiversity and buffering of storms runoff and floods.

CWs are a relatively recent technology compared to the conventional wastewater treatments. In the last decades, many progresses have been made in the field of ecological engineering and phytodepuration applied to CWs, so they have reach high levels of performance (Vymazal, 2014; Camacho *et al*, 2018). However, there are many issues and new challenges that still need to be addressed. Initially, it is still needed to increase the knowledge on the microbial and biogeochemical process occurring within the CWs, which may get a boost from using novel diagnostic molecular and bioinformatic methodologies (Rochera *et al*, 2021). This would shed light on processes conducting to naturalizing the effluents of wastewater treatments, which is a way to minimize the impact in the natural receiving environments, particularly in those from deficit basins, as those of the Mediterranean, where the wastewater effluents often constitute a significant percentage of the natural flows. Other challenge gaining attention on CWs is its environmental sustainability in terms of climatic regulation. This is based in the idea of combining its primary function of water purification with an additional role as a carbon sink. This purpose goes through enhancing the sequestration of CO_2 but restricting as well wherever possible the emissions of greenhouse gases (GHG) with high warming potential such as CH_4 and N_2O (Mander *et al*, 2014; de Klein and Werf, 2014).

An enterprise-research consortium composed by Global Omnium, a Spanish based company dedicated to water management, and the Limnology Research Group from the Cavanilles Institute for Biodiversity and Evolutionary Biology (University of Valencia), are carrying out research studies in the last years on different configurations of CWs designed as part of secondary treatment or as a polishing tertiary treatment for treated wastewaters. As being functionally complex recreations of natural ecosystems, the monitoring of these CWs requires a systematic (i.e., multiparametric) approach that considers a variety of physical-chemical, biochemical, microbiological, and functional properties.

METHODS

Different configurations of CWs designed for different purposes were studied in facilities managed by Global Omnium in the eastern Spain (Fig 1). One of them, representative of the CWs that the company is operating in sparsely populated mountain areas, consisted in a set of two subsurface flow wetlands (flow direction: vertical and horizontal respectively) arranged in series, which were differently vegetated by *Helosciadum nodiflorum,* a riparian plant typical from nearby streams, and *Typha latifolia,* respectively (Camacho *et al*, 2018). This was preceded of a pre-treatment and a biological (extended aeration activated sludge) treatment and settling. A second studied, free water-surface CW, was located closer to the coast, in a more populated area (Peña *et al*, 2019b). This consisted in a surface linear watercourse of approximately 400 m^2, vegetated in the shores mainly by *Typha spp.* and *Iris pseudacorus*. This CW has been used to explore the possibilities of re-naturalizing water after being properly treated by conventional procedures but still showing some characteristics more similar to treated wastewaters than to those of the receiving natural environment. Accordingly, this CW was fed by a small fraction of the effluent from a secondary treatment.

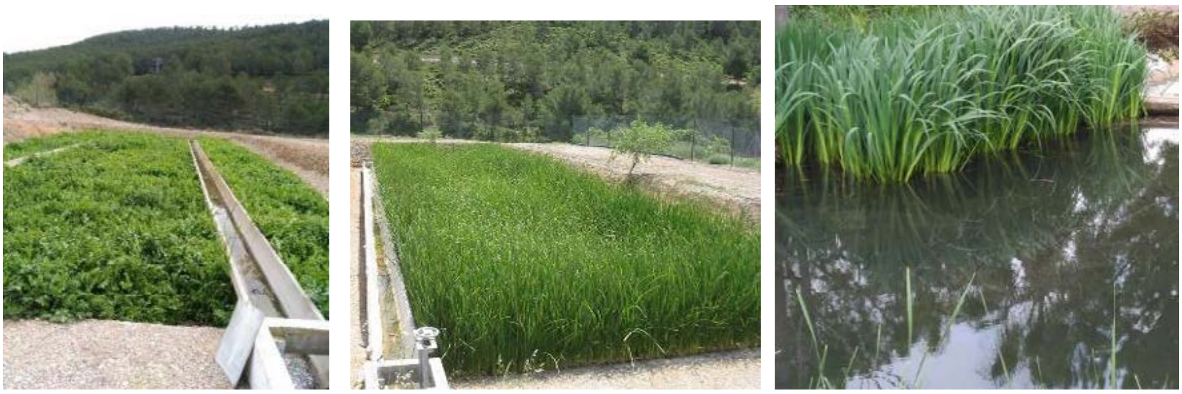

Figure 1. *Photographs of the studied subsurface and surface flow CWs.*

The performance of CWs was regularly assessed by comparing variations in the variables measured in the inlet and outlet of each subparts or whole system. Regular control variables of wastewater were analysed following standard methods (APHA, 2005). Furthermore, a characterization of dissolved organic matter (DOM) was performed by spectrophotometric techniques (Hansen *et al*, 2016) to track biological activities related with the DOM processing and transformation. The abundance and activity of the bacterial population were evaluated by cytometric methods (Picazo *et al*, 2019). Additionally, the metabolic rates related with the exchange of carbon with the atmosphere were also measured in different parts of the system (Morant *et al*, 2020). Finally, a genomic study consisting in a high-throughput sequencing of the environmental DNA (Kozich *et al*, 2013; Picazo *et al*, 2019; Rochera *et al*, 2020) was performed to determine the taxonomic and functional structure of the microbial community.

RESULTS AND DISCUSSION

The CW system from the mountain area, that combined two types of subsurface flows (vertical/ horizontal) and plant types in the two different basins (the riparian *Helosciadum nodiflorum* and the commonly used in CW *Typha latifolia*), demonstrated to be efficient in removing concentrations of major inorganic nutrients and the remaining OM as already described in previous studies (Camacho *et al*, 2018). Optical analysis of DOM demonstrated, in this case, the reduction of recalcitrant substances potentially produced during the preceding conventional treatment (i.e., fulvic acids), of up to 40%. On the other hand, the free water-surface CW from the most populated location functioned rather as a polishing system. This CW removed efficiently biodegradable OM regardless of already receiving low loads from the conventional secondary treatment. These reductions occurred in parallel to an increase of the DOM aromaticity in the outlet (Fig 2) as demonstrated by the specific ultraviolet absorbance at 254 nm ($SUVA_{254}$). This DOM was less reactive as indicated by the reduction of biodegradability (Fig 2), which led it to be more protected from further microbial degradation once arriving to the receiving environment. This highlights the role the CW as a transitional system between treated wastewater and natural ecosystems.

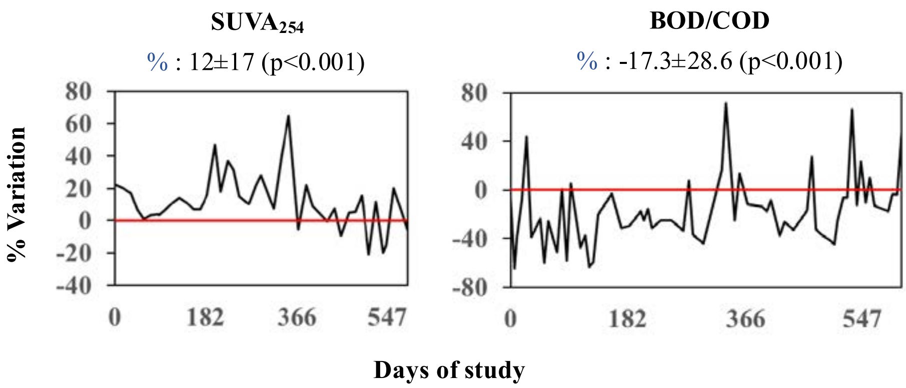

Figure 2. *Percentage variation of DOM-related parameters in the outlet compared to the inlet. Specific ultraviolet absorbance at 254 nm ($SUVA_{254}$) is proxy of DOM aromaticity and the ratio between biological and chemical oxygen demand (BOD/COD) indicates OM biodegradability. Positive and negative values indicate increases and decreases, respectively.*

The CWs from the mountain location, functioning with subsurface flow, operated efficiently as biofilters and contributed more clearly to reduce bacterial abundance in water (Fig 3). This is the case for instance of potentially pathogenic genera such as *Arcobacter*, that decayed significantly in the outlet, which may respond to the adverse environmental conditions imposed by the CW to this bacterium (Fera *et al*, 2004). On the other hand, in the surface CW, the increases of the oxygenation over the hypoxia, the exposure to solar light radiation and the interaction with vegetation promoted important changes in the microbial community (Table 1). This involved a general increase of the aerobic bacteria, including those typical from freshwater communities such as *Limnohabitans*, *Limnobacter* and *Polynucleobacter*. Bacteria able to degrade plants material (e.g., xylose, vanillin, syringate) also increased their abundance, likely by the release in the CW of OM from vegetation (Table 1). By contrast, the capacity for denitrification and dissimilatory reduction of sulfate, both predominant processes in anaerobic environments, decreased in the outlet.

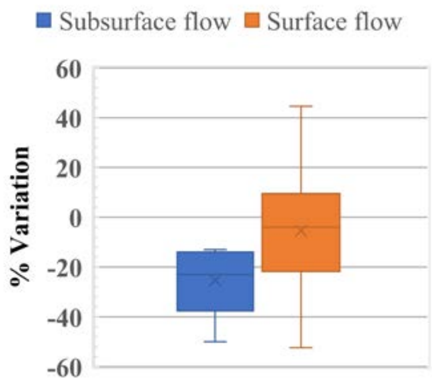

Figure 3. *Percentage variation of bacterial abundances in the outlet with respect to the inlet in contrasting CW design. Positive and negative values are increases and decreases, respectively. Data are partially extracted from Camacho et al (2018).*

More consistent measurements of metabolic rates were conducted in the surface CW. In this case, the balance between photosynthesis and aerobic respiration was, as a rule, slightly heterotrophic. The balance was only autotrophic if the plankton compartment was considered alone for calculations and under low-flow conditions. However, it should be noted that in slightly alkaline waters like those of the surface CW (pH~7.5), the carbon fluxes are mainly regulated by the calcite-carbonate system of water, in such way that an important part of the inorganic carbon released by respiration is captured into the bicarbonate-carbonate system and precipitates as calcium carbonate. Moreover, in terms of the total carbon budget of the CW, potential emissions are also counteracted by the carbon fixation triggered by the riparian vegetation. The methane emissions ranged usually low to moderate, suggesting that this was not a prominent pathway for C-GHG outflow in this CW. However, higher rates were occasionally measured during warm periods. This highlights the role of temperature in controlling methanogenesis over other environmental factors, which has also been demonstrated experimentally (Camacho *et al*, 2017), and stresses its importance in warmer geographical regions.

Table 1. *Main transformations observed of treated wastewater after passing through the surface CW. This shows the naturalizing process of DOM and microbial communities in these transitional facilities.*

Trend	Decrease in the outlet	Increase in the outlet
Nature of DOM	Protein labile fraction	Humic and less reactive DOM.
Microbial community structure	Sulfur-oxidizing bacteria, denitrifiers and other microbial guilds from wastewater treatment	Photoautotrophs and heterotrophic bacteria typical from freshwater ecosystems
Main metabolic trends	Fermentative or anaerobic pathways and degradation of diverse aromatic compounds	Aerobic heterotrophy, using compounds other than biopolymers or aromatic hydrocarbons. Skills for degradation of vegetal materials (e.g., xylanolytic and cellulolytic activities)

CONCLUSIONS

The study demonstrates the utility of different CW designs to further reduce the impact upon discharge of treated wastewater, acting in some cases as a proficient transition from the conventional treatment to the natural environment. The multi-analytical approach used proves the ecological upgrading provided to water by these CWs. Mainly, this involves an extra reduction of pollutants, a transformation to a biochemically more stable DOM, and the change into microbial communities with functional trends more harmonized with those of the microbiota of the receiving environment. Wherever it was possible to conduct a comprehensive metabolic survey, results indicate that occurs with a carbon balance close to neutral. The buffering role of CWs could be particularly mandatory in water stressed regions, where treated water discharges can be relatively high compared to the natural flows of the receiving environments.

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PHYTOREMEDIATION IN PORTUGAL: APPLICATION OF PLANTS TO DIFFERENT WASTEWATERS

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Abstract

Phytoremediation is a technology that can involve different processes such, mobilization, stabilization, and elimination of contaminants through, phytostabilization, phytodegradation, phytovolatilization, phytoextraction and rhizofiltration. It consist of the use of plants, and their associated microorganisms, to reduce the concentrations or toxic effects of different types of contaminants that are found in the environment. Is mostly applied to treat polluted soil, contaminated surface water and wastewater. Constructed wetlands and floating wetlands are examples of phytoremediation technologies for water treatment. Briefly, constructed wetlands comprise an excavated bed with impermeabilization, a substrate and vegetation. Floating wetlands comprise a floating platform with an anchoring system and vegetation adapted to hydroponic mode. In water environment the macrophytes are often used for phytoremediation, although careful selection should be carried out to cope with the specificity and concentration of pollutants. Main advantages of this technology are that it is nature-based, provides several ecosystem services and is low cost in terms of implementation, operation and maintenance compared with conventional approaches. Some limitations relate to the area needed for establishment (in the case of constructed wetlands) and toxicity effects associated to higher concentrations of pollutants.Portugal is a country that pledged to become a green country within 2050. The use of green and blue solutions is very important for climate change mitigation and adaptation. Therefore, the use of phytoremediation through floating islands and constructed wetlands can be a feasible and sustainable way to help meeting the green cities goal, in relation to water quality improvement and biodiversity promotion.The present study aims to do a bibliographic research about phytoremediation technology applied to wastewater management in Portugal, with special focus on the plant species used. The methodology will comprise the search of papers, thesis, and dissertations based on "science direct" search engine and Portugal University´s online repositories. To complement the bibliographic research, field trips will be made to two places of technology implementation in the north region: a floating island at Leixões port and a constructed wetland at "Paço de Calheiros", tourism unit placed in a rural area. Within this framework we intent to gather knowledge on the state of the art concerning phytoremediation in Portugal, types of constructed and floating wetlands in use, typologies of wastewaters, operational characteristics and specially plant species considered. We also expect that the information gathered will set a platform for phytoremediation dissemination.

Keywords: ecosystem services; water management; macrophytes; phytoremediation; green solutions

INTRODUCTION

The current demographic growth and urbanization leads to an increase in waste production and energy consumption.Legislation is often not up to date and adapted to current pollution levels (WWDR,2017).Nowadays, with the COVID-19 pandemic and the consequent economic crises,we see more than ever the necessity to invest in alternatives aiming for social, economic, and environmental sustainability. Some developing countries do not have water treatment systems and if they do, they are not adapted to remove all the contaminants that arrive there. Even some developed countries still need to improve the treatment of wastewater with certain pollutants such as heavy metals, drugs, pesticides, dye, hydrocarbons, coming from different sources. (UNEP,2015).This is because for an efficiency of more than 99% in removal of all the impurities and pathogens from sewage it is necessary to have a tertiary treatment.Most of the time, this can be expensive, requiring a high level of technical knowledge, trained operators, a steady energy supply, chemicals and equipment (UNEP,2015). A way to help this problem can be investing in cheaper secondary treatment that use bioremediation and phytoremediation. Through that, some pollutants can be degraded by biological means, extracted from the water and transformedfrom high to low toxicity (Abid Ali Ansari, 2020).

Phytoremediation can be applied as secondary or tertiary wastewater treatment. It comprises the use of plants, and their associated microorganisms, to reduce the concentrations or toxic effects of different types of pollutants that are found in the environment. It is mostly applied to treat polluted soil, contaminated surface water and wastewater. In this technology plants can use different mechanisms to mobilize, stabilize and eliminate contaminants, such as absorption, translocation, evapotranspiration and metabolization. Some processes that use these mechanisms are phytostabilization, phytodegradation, phytovolatilization, phytoextraction, phytostimulation and rhizofiltration (Yadav, B. & Akhtar, et al, 2015).

One important component that must be taken in consideration when applying phytoremediation is the type of pollutant that we aim to remove. The pollutants can be organic (mostly man-made, xenobiotics, and can be degraded) or inorganic (they occur as natural elements in the earth's crust or in the atmosphere, that cannot be degraded). The pollutant can be absorbed, translocated, evapotranspirated, degraded, accumulated or metabolized by plants. For this to occur, the pollutant must be bioavailable in the environment. Some factors interfere with the bioavailability of organic and inorganic pollutants such as their chemical properties (hydrophobicity, pH, redox state, volatility), soil properties, environmental conditions and biological activities that occur in the environment. In addition, it is important to consider the effects that pollutants can have on plants. When it comes to organic pollutants, some xenobiotic effects can occur in plants. This effect varies according to the chemical structure, concentration and accumulation site, resulting in biochemical, physiological and molecular changes (Pilon-Smits E. ,2005).

Another important factor is related to the choice of the most adequate plant for the type of treatment to be carried out. In water treatment, macrophytes and algae are the most used organisms for phytoremediation, although careful selection should be performedin order to cope with the specificity and concentration of pollutants. To choose the thisplant one must consider its ability to survive in environments with high toxicity and if the plant is native to the region to avoid further ecological problems.Another relevant point is to choose plants with some specific anatomical characteristics such as long and dense root system, high level of degradative enzymes, production of abundant root exudates, high biomass and rapid growth.There are many plants with the innate ability to remedy pollutants present in the environment. This capacity is directly proportional to the growth rate of the plant, being correlated with the total biomass of the plant.(Pilon-Smits E. ,2005). Because of this, it is necessary to identify species with high growth rate and biomass or modify the metabolism of those already known through biotechnology.

Furthermore, the plant species can be selected based on its value and on how it can be applied in the local economy. That is because the use of plants can also be lucrative if there is an intention of biomass valorization.Other advantages of technologies based on phytoremediation are that they are nature-based, provide several ecosystem services and are low cost in terms of implementation, operation and maintenance compared with conventional approaches. Some limitations arerelated to the area needed for establishment (in the case of constructed wetlands), toxicity effects associated with higher concentrations of pollutants, the pollutant removal process potentially being slower than conventional methods and the possible lack of bioavailability of pollutants for processing by the plants (Pilon-Smits E. ,2005).

Some phytoremediation applications for water treatment are constructed wetlands (CW) and floating island or floating wetlands (FI). Floating wetlands comprise a floating platform with an anchoring system and vegetation adapted to hydroponic mode. This system consists mostly of aquatic plants that grow hydroponically on floating structures set up on the surface of water bodies. The entire submerged surface of the islands and the plants roots serves as the basis for the fixation of microorganisms (biofilm), which favors the decomposition of organic matter and adsorption of suspended solids (Yeh et al., 2015).

A constructed wetland mimics the biogeochemical processes that occur in natural wetlands promoting water depuration. They are engineered systems that combine plants, substrate and associated microorganisms. Their application can be adapted for secondary or polishing wastewater treatment. (Nabais C., 2007).Some of the advantages of this system is that it is a low-cost and low technological system (Dotro et al., 2017) and versatile in terms of flow. It is also possible to have systems that bring together different flows which are called hybrid systems (Kadlec and Wallace, 2008). This system can also show a satisfactory removal of suspended solids, organic matter and nutrients, and natural decay of pathogens(WHO, 2018).

In Portugal, the interest in phytoremediation-basedtechnology to treat contaminated water started at the end of the 80´s with the visit of Keith Saidal, one of the CW technology's creators in Germany. She tried to show how this system works and its importance in wastewater treatment.After that, the first CW pilots began to be operated at Lisbon State Institute IST, Évora State University and Fábrica Anilina. However, the first full-scale constructed wetland was built at Estarreja in 1993 and was designed as a subsurface vertical flow type for industrial wastewater treatment. Besides that, the first full-scale constructed wetland to treat domestic wastewater was implemented in 1997 at Pinheiro da Cruz (Dias V.N et al.,2000). According to the same author, atthe end of the 20th century there were almost 76 constructed wetlandsin operation in Portugal.Based on the research that is carried out, there are no detailed records with current accounting for constructed wetlands and floating wetlands in operation.

Green- Blue solutionsare frequently promoted as terms for such sustainable multifunctional measures able to reduce negative effects of urbanization and adapt to a changing climate, such as heat and water regulation, air and water purification, increased biodiversity and recreation. When implemented in the urban context their aim is typically stormwater detention, infiltration and purification in a decentralized manner(Sörensen et al.,2019). According to the National report on the implementation of the 2030 Agenda for Sustainable Development published in 2017, Portugal is a country that adoptedthe*EU Strategy for Adaptation to Climate Change*and aims forthe neutrality of greenhouse gas emissions within 2050. The objective is to become more resilient to the impacts of climate change. One way to reach that is to transform cities into green cities. As specified by*Brilhante, O; Klaas, J. (2017)*a green city is “a city that promotes energy efficiency and renewable energy in all its activities, extensively promotes green solutions, applies land compactness with mixed land use and social mix practices in its planning systems, and anchors its local development in the principles of green growth and equity.”.As how constructed wetlands and floating islands are considered green solutions, the use of phytoremediation can be a feasible and sustainable way to help the green cities goal. (Sang, 2017)

For an efficient water management it is important to know what kind of effluent will be treated and for which purpose it will be reused or where it will be disposed after treatment. There are several sources that pollute the water in Portugal. For example, the large amount of drugs from residential water that cannot be removed from urine through WWTPs. Furthermore, chemicals that are byproducts from the wine, textile, tannery, mining, agricultural and tanning industries. An example of an urban source is the large production of trash and leaching from sidewalks. Other factors that can reduce the water quality are the overpopulation of seagulls and pigeons, combined with the practice that some people have of feeding aquatic animals in parks. This practice produces feces and food waste that sometimes decompose as organic material in water, contributing to the eutrophication process. (Wither, A., et al., 2005).

There are still some gaps related to water treatment through phytoremediation, such as how to solve the main disadvantages described above, which are linked to the plant’s tolerance to the pollutant and to the xenobiotic effects, the efficiency of removing organic and inorganic pollutants, treatment time, space required, and bioavailability of pollutants.With this work, it isintended to contribute to clarify some of these issues in Portugal context by conducting a data survey to fill the gap described above on the current accounting for constructed wetlands and floating wetlands in operation in Portugal. Furthermore, relating plant species that have already obtained positive results to treat a specific type of effluent in a specific system can also help to fill these gaps. Through that it is expected to avoid the loss of time, investment and biomass caused by xenobiotics effects.

METHODOLOGY

For this systematic review, the methodology will comprise a scan of the literature. To complement the bibliographic research, field trips will be undertaken to an established constructed wetland and floating wetland. It is intended to be implemented, under the present study, a pilot floating wetland in a port marina.

The process of writing a literature review will cover the following steps: Define our research question, plan our approach to our research and our review, search the literature, analyze the material we have found, managing the results of our research, and writing our review. The search of papers, theses and dissertations within the year range from 2000 to 2020, will be carried out based on science direct (https://www.sciencedirect.com/) and Google Scholar (https://scholar.google.com.br/schhp?hl=pt-PT) search engines, and Portugal University´s online repositories. Also, a parallel literature review has been made of studies published at conferences. The following terms were used: (Phytoremediation or Fitorremediação) AND (Portugal) AND (Greens technologies OR Green engineering OR tecnologiasverdes) AND (Constructed wetlands OR Wetlands construídos) AND (Floating island OR Floating wetlands OR Ilhasflutuantes).To assist the literature review, some questions will be answered, such as:

- What is known about phytoremediation in Portugal?
- Are there any gaps in the knowledge of phytoremediation in Portugal?
- Have areas of further study been identified by other researchers that we may want to consider?
- What aspects have generated significant debate on the topic?
- What methods or problems were identified by others studying in the field and how might they impact our research?
- What is the most productive methodology for our research based on the literature we have reviewed?
- What sources of information or data were identified that might be useful to us?

We will also categorizeall the literaturein aresearch log in a spreadsheet to record the theories, methods, findings of articles we have read. The categories that will be used are: Code, Title, year, macrophytes, System, effluent, source, keywords, authors and country. For the references, the*Mendley* platform will be used. Subsequently, we intend to perform a data mapping through graphs to outline the results found**.**

To contribute with the bibliographic research, field trips will be made to two places of technology implementation in the north region of Portugal: a floating island at “Leixões port”, a constructed wetland at “Paço de Calheiros”, placed in a rural area.

A Floating Wetland Island (FWI) will be set up in the Marina of the Porto Cruise Terminal in Matosinhos (*APDL-Administração dos Portos do Douro e Leixões SA*). This Marina has 2 ha. of surfacewater andis positioned at the port entrance, being influenced by tides from the Atlantic Ocean and the river mouth (*Leça river*) that flows through the Port and by the traffic of cargo ships, fishing boats, and cruise ships that pass by near the respective section in the port. The FWI pilotthat will be implemented in March 2021 will follow the previous guidelines established by *Calheiros et al.(2020).* It will consist of a platform made by cork, anchored to the Marina. These platforms will beable to receivethe plants that will be placed in a coconut fiber vase, filled with rockwool.Also, plants will be carefully selected following the criteria previously mentioned.

EXPECTED RESULTS

Within this literature review it is intendedto gather knowledge on the state of the art concerning phytoremediation in Portugal, types of constructed and floating wetlands in use, typologies of wastewaters, operational characteristics and especially plant species considered. It's also expected that the information gathered will set a platform for phytoremediation dissemination per a publication of a paper based on this work. Also, the use of graphics and tables can help an easy understanding of the data and information that we intend to show.Lastly, it´s expected to answer which plant has worked well to clean which type of effluent in Portugal based on previous research.

Doing the two field trips it is intended to observe two types of technologies that use phytoremediation in Portugal. It is also intended to analyze the latest results obtained in order to complement the literary research. Furthermore,with the implementation of a FWI it is expected to assess the level of difficulty of the process and the cost of implementation to show how accessible this type of system can be, in addition to contributing to the practical dissemination of phytoremediation through the implementation of a new prototype.

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DEVELOPING A NEW DESIGN OF ANAEROBIC DIGESTER FOR THE TREATMENT OF RAW WASTEWATER IN COMBINATION WITH CONSTRUCTED WETLANDS

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Abstract

A new model of combined digester-wetland system was used to treat municipal wastewater in a pilot plant at the University of ACoruña. The system consisted of a first hybrid anoxic digester unit (AD) followed by a vertical flow constructed wetland (VF CW)from which the VF effluent was recirculated to AD inlet. AD aimed to perform removal of solids and organic matter and support denitrification. VF CW aimed to obtain nitrification in the wastewater after achieving low levels of particulate and soluble organic matter. The system combined anoxic-anaerobic and aerobic conditions that improvedtotal nitrogen removal. The overall removals were about 96-98%of TSS, 80-95% of COD, 95-98%of BOD_5 and 79-95% of NH_4^+-N.

INTRODUCTION

Water quality problems persist in both developed and developing countries. In developing rural areas,it is estimated that 95% of wastewater is discharged into the environment without any treatment when population density is not sufficient to justify connection to the sewage system (ONU, 2018; WWAP, 2019).

A combination of anaerobic-aerobic process through non-conventional systems, e.g., an anaerobic digester-constructed wetland, may be one of the strategies to be followed. Constructed wetlands (CW) are eco-friendly systems whose removal mechanisms are based on natural processes. The main CW limitations are clogging of granular media and large land area requirements. In this regard, anaerobic digesters (AD) have been used as a CW pre-treatment in order to reduce suspended solidsand organic matter loading rate, consequently decreasing the required footprint (Álvarez et al., 2008; De la Varga et al., 2013; Ruiz et al., 2010). Moreover, combined AD-VF systems can reach simultaneous organic matter and nitrogen removal through efficient nitrification and denitrification trough recirculation from the VF effluent to the AD inlet (Gonzalo et al., 2017).

Therefore, it is required to develop systems appropriate to local conditions together with applicable design and operation criteria. Effluent recirculation to sludge bed digesters (i.e., UASB or HUSB digesters) can lead to sludge washout that deteriorates both denitrifying capacity in the AD unit and the performance of the subsequent CW unit due to clogging. The scientificliterature lacks sufficient information on the effect of VF effluent recirculation on the anoxic pre-treatment unit(Gonzalo et al., 2017; Tanner et al., 2012). The aim of this work is to develop a novel AD design suitable as pre-treatment for CW units and to study the performance of a combined AD-VF system with effluent recirculationregarding the simultaneous removal of organic matter and nitrogen from municipal wastewater. Particular attention has been paid to the parameters of TSS, chemical oxygen demand (COD), biological oxygen demand (BOD5) and ammonium.

METHODS

The combined system was located outside the Science Faculty of the University of ACoruña (A Coruña, Spain) and it was treating wastewater from one of the faculties and surrounding houses, plus stormwater and runoff.Thecombined system was constituted by two pilot units connected in series, the first one consisting of a hybrid AD followed by the VF CW unit. As the land was on a slope, the AD was placed higher than the VF CW to feed it with AD effluent by gravity. Raw wastewater (WW) and the recycled VF effluent wereintermittently pumped into AD and the pre-treated wastewater entered the CW.The final effluent was collected in a tank. According to the defined recirculation ratio (R) (Table 1), the VF effluent was pumped to the AD inlet at the same time as the WW.AD unit combined an up-flow sludge bed zone and an anaerobic filter (AF) whose active volume was 0.222 m^3 and surface area of 0.385 m^2. VF CW had an overall surface of 3 m^2 and its filtering medium consisted of an80 cm layer of 1-3 mm coarse sand and an upper 5 cm layer of 0.5-2 mm fine sand. Both units were planted with *Phragmites australis*. The AD unit was fed intermittently by pulses every 2 hours and consequently the VF CW too. A rest of 3 days on a weekly basis was applied to the AD and VF CW. As a result of the rainfall entering the collection sewer, wastewater was strongly diluted. Therefore, from period II onwards, raw wastewater was supplemented with a concentrated synthetic substratum. The influent average concentration for the overall period was (in mg L^{-1} except pH): pH 7.0±0.3, TSS 192±103, COD 342±196, BOD_5 206±124 and NH_4^+-N 34±21.

The study was carried out during 508 days of which 81 days constituted the start-up of the combined system. During the start-up, there was no recirculation, and the units were subjected to an adaption and stabilization time. Therefore, the evaluated periods began with the beginning of VF effluent recirculation and were divided into five periods, as indicated in Table 1. R ranged from 0.7 to 1.5.The overall hydraulic loading rate (HLR) ranged from 77 to 233 mm·d^{-1}(without recirculation) and the overall surface loading rate (SLR) ranged from 8 to 70 g TSSm^{-2}·d^{-1},from 12 to 123 g COD m^{-2}·$d^{-1}$and from 7 to 83 g BOD_5 m^{-2}·d^{-1}, due to variations in influent flow and concentration (Table 1). Ammonia SLR ranged from 1 to 10 g N m^{-2}·d^{-1}, whilst total nitrogen load may be slightly higher because of partial ammonification of influent nitrogen.

TABLE 1. ***OPERATIONAL CONDITIONS OF THE COMBINED SYSTEM.***

Period	Q_{ww}	R	AD HRT	Overall HLR	Overall SLR (gm^{-2}·d^{-1})[a]			
(days)	(Ld^{-1})	(Q_R/Q_{IN})	(h)	(mmd^{-1})	TSS	COD	BOD_5	N-NH_4^+
I (105)	259±54	0.94	19.8	77	8	12	7	1
II (105)	554±68	0.85	9.5	164	39	74	49	6
III (35)	790±8	0.67	6.7	233	70	123	83	10
IV (126)	342±36	1.49	15.6	101	27	52	29	6
V (56)	368±33	0.78	14.3	109	19	28	13	4

[a]The overall SLR was defined as the ratio of the influent mass (g/d) to the total area of the system (AD+ VF CW).

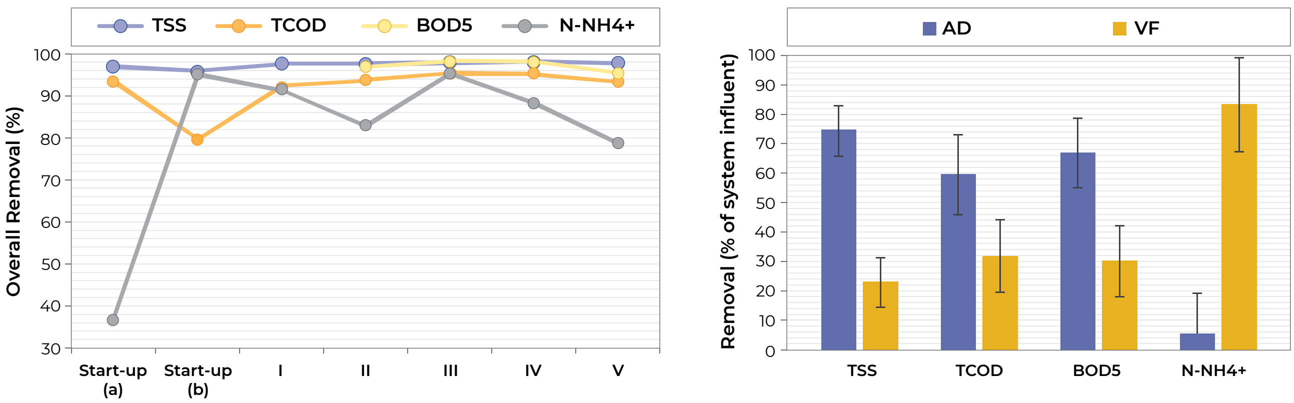
Influent and effluent composite samples from each unit were obtained integrating daily samples. Samples were analysed following Standard Methods (APHA-AWWA-WEF, 2017) for TSS,COD and BOD_5. Nitrogen compounds were determined by Ionic Chromatography. Solid accumulation in the AD as well as its specific methanogenic activity (SMA) and specific denitrifying activity (SDA) were determined periodically.

RESULTS AND DISCUSSION

The combined system AD-VF removed about 96-98% of TSS (97.5% on average), 80-95% of COD (91.6% on average), 95-98% of BOD_5 (97.1% on average) and 79-95% of NH_4^+-N (88.6% on average), as shown in Fig. 1.Most of the TSS were removed by the AD, which achieved removal efficiencies of 75±9% while the VF CW contributed removals of 23±9%, respectively to the raw influent (see Fig. 1). In terms of organic matter, the AD was also the most efficient with average removals of 60±14% in COD and 67±12% in BOD_5 whereas about 30% of COD and BOD_5 was removed by the VF CW. After start-up, the overall system produced stable effluents with low levels of particulate and soluble organic matter. The AD achieved a concentration below 88 mg TSS L^{-1} (29±16 mg L^{-1} on average), 260 mg COD L^{-1} (82±50 mg L^{-1}) and 110 mg BOD_5 L^{-1} (50±30 mg L^{-1}). Because of the pre-treatment, the VF CW treated a low suspended solid and organic matter influent, which means that the VF CW could reduce its footprint and reduce clogging of the granular media. Finally, the AD-VF combined system obtained a final effluent with concentrations below 25 mg TSS L^{-1}, 45 mg COD L^{-1} and 18 mg BOD_5 L^{-1}.

The increase in loading rates during period II caused a greater accumulation of active biomass in the AD which improved its filtering capacity.In the whole period (I toV), the AD digester removed through hydrolysis 80.8% of influent volatile suspended solids(VSS; mass balances not shown).The sludge generated represented 6.1% of the influent VSS while the remaining 13.1% was in the AD effluent. Meanwhile, the VF CW operated an SLR of 8.3 g TSS $m^{-2}{\cdot}d^{-1}$, 22.5 g COD $m^{-2}{\cdot}d^{-1}$ and 16.3 g BOD_5 $m^{-2}{\cdot}d^{-1}$.The VF CW achieved surface removal rates (SRR) of 1.9 g TSS $m^{-2}{\cdot}d^{-1}$, 7.3 g COD $m^{-2}{\cdot}d^{-1}$ and 5.6 g BOD_5 $m^{-2}{\cdot}d^{-1}$, on average.

FIGURE 1. ***PERCENTAGE REMOVAL OF TSS, COD, BOD5 AND NH4+-N IN THE OVERALL SYSTEM AND CONTRIBUTION OF EACH UNIT.***



On the other hand, ammonia was mainly removed by VF CW due to its aerobic conditions (see Fig. 1). Based on the data in Table 2, it could be seen how the environmental conditions affected the yield of nitrogen compounds. Thus, the AD was responsible for removing nitrates that were recirculated from the VF effluent due to the prevailing anoxic conditions.During period III, the VF effluent had the lowest ammonium concentration, despite the highest SLR (10 g NH_4^+-N $m^{-2}·d^{-1}$). The lowest nitrate concentration for AD effluent was obtained also during period III. This could be since period III operated at the highest influent temperature (Table 2), which could favour the activities of denitrifying and nitrifying bacteria.R appeared not to greatly affect the ammonium concentration in the AD effluent neither the nitrate concentration in the VF effluent. Theseeffluent concentrations seemed to be more related to the SLR applied in each period and temperature.Both units produced irregular nitrite concentrations according to its role as intermediary in the nitrogen cycle. In addition, the presence of plants in both the AD and the VF CW may have contributed to the elimination of the nitrogen compounds studied. It should be noted that the VF CW operated at SLR between 1.3-8.2 g NH_4^+-N and achieved SRR between 1.2-6.4 g NH_4^+-N, indicating its high nitrifying capacity.

TABLE 2. ***CHARACTERISTICS OF THE NITROGENOUS COMPOUNDS IN THE EFFLUENT OF BOTH UNITS***

Period	T_{IN}	R	AD effluent[a]			VF effluent[a]		
	(°C)	(Q_R/Q_{IN})	NO_3^--N	NH_4^+-N	NO_2^--N	NO_3^--N	NH_4^+-N	NO_2^--N
I	12.8±2.3	0.94	1.2±0.1	8.7±5.2	0.13±0.09	10.7±3.9	1.3±1.5	0.11±0.08
II	18.2±3.2	0.85	2.2±1.8	22.9±12.6	0.53±0.56	19.1±4.3	6.7±4.5	0.18±0.13
III	21.7±0.3	0.67	0.4±0.5	19.5±9.3	0.06±0.06	18.0±5.5	2.1±1.9	0.10±0.04
IV	19.6±2.2	1.49	1.8±1.3	28.6±10.7	0.30±0.23	23.1±9.2	6.9±4.9	0.07±0.03
V	15.0±0.8	0.78	2.4±1.7	19.7±8.2	0.12±0.06	14.5±3.8	7.0±6.8	0.07±0.06

aConcentration units: mg L-1

SMA showed low values (between 0.026-0.056 g COD_{CH4} g $VSS^{-1}·d^{-1}$) with short latency periods (1-2 days). SDA of the sludge bed zone was between 2.5-4.7 mg N g $VSS^{-1}·h^{-1}$ while SDA of gravel of AF was maintained about 0.9-2.6 mg N g $VSS^{-1}·h^{-1}$. Owing to a higher amount of biomass attached to the AF than biomass accumulated in the sludge bed, about 64% of the potential denitrifying capacity of the digester was provided by the AF. As a result, the AD generated an effluent with a low nitrate concentration during the whole study (1.7±1.5 mg NO_3^- L^{-1} on average), so high removal efficiencies were achieved. Moreover, surface nitrification rate (SNR) was calculated in VF CW regarding denitrification process could considered negligiblein this unit. So, SNR equalled the rate of nitrate formation in the VF CW. The SNR was higher between periods II to IV (6-8 g NO_3^--N $m^{2-}·d^{-1}$) than during periods I and V (between 2-3 g NO_3^--N $m^{2-}·d^{-1}$). In addition to the influent ammonium concentration, this behaviour was explained by the higher activity of nitrifying bacteria during periods II to IV (spring-summer season) than in periods I and V (winter season). Nitrate accumulation in the final effluent indicates that improving total nitrogen removal will require higher recirculation rates.

CONCLUSIONS

A new AD design was tested for the treatment of raw WW in combination with CW.The AD removed 61-83% of TSS, 43-75% of COD and 50-76% of BOD_5, highly reducing the influent concentration of particulate and soluble organic matter to the VF CW. Surplus sludge was produced at a rate equivalent to 6.1% of influent VSS. On the other hand, the intensification of nitrogen removal efficiency was carried out through recirculation of the nitrified VF effluent to the AD inlet. In this pathway, the AD effluent had low nitrate concentrations (1.7±1.5 mg NO_3^- L^{-1} on average) because of denitrification process under anoxic environments while the VF effluent achieved low ammonium concentrations (4.8±5.0 mg NH_4^+-N L^{-1}) due to prevailing aerobic conditions. The combined AD-VF system removed 96-98% (97.5% on average)of TSS, 80-95% (91.6%)of COD, 95-98% (97.1%)of BOD_5 and 79-95% (88.6%) of NH_4^+-N.

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SOIL AMENDMENTS TO IMPROVE NUTRIENT ATTENUATION IN VEGETATION FILTERS

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Abstract

Vegetation Filters (VFs) can be a sustainable solution to treat wastewater and to recover resources such as nutrients,water and biomass from small municipalities and isolated dwellings. However, under certain conditions, the leakage of nutrients, especially of nitrate, can represent a limitation. The addition of two sustainable soil amendments, woodchips and biochar, has been tested as a strategy to improve nutrient attenuation in VFs increasing sorption sites and microbial activity. To this end, unsaturated infiltration and batch experiments have been carried out at laboratory scale. The systems for infiltration experiments contain natural soil, natural soil amended with woodchips and natural soil amended with biochar. To determine the sorption capacity of NH_4^+, batch tests were performed using an amendment/SWW ratio of 1:20 and an NH_4^+initial concentration ranging from 30 to 600 mg/L. Results from the infiltration experiments show a high attenuation (~95%) of total phosphorous (TP) independentlyof the amendments. Different behaviour is observed for total nitrogen

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(TN). The removal of this species is obtained only in the soil amended with woodchips (>85%) whereas the natural soil alone and the soil with biochar have no impact on TN attenuation. In these two porous media, all the NH_4^+input concentration is transformed to NO_3^-that infiltrates without further reactions. According to batch experiment results, the potential-role of biochar in the nutrient attenuation is limited to sorption processes (K_d (NH_4^+) = 21.35-193.18 L/kg). Woodchips act primarily as a labile source of carbon promoting biodegradation, being more effective for nutrient attenuation than the sorption capacity of biochar.

INTRODUCTION

Vegetation filters (VFs) are a non-conventional water treatment technology where wastewater and/or treated water is applied for the irrigation of a forestry plantation. The treatment is carried out by the mutual action of soil, microorganisms and plants. In this system, part of nutrients and water are recovered from wastewater by plant uptake to generate biomass. The main advantages are: i) the low maintenance costs; ii) the production of biomass as an extra economic value; iii) the low energy consumption; iv) the contribution to mitigate climate change as CO_2 sink; v) the increase of groundwater resources by irrigation returns and vi) the creation of an ecologic niche that fosters biodiversity.

Different studies have demonstrated that VFs represent a robust environmental management application since they are able to reduce contaminant loads (de Miguel et al., 2014; Martínez-Hernández et al., 2018). However under certain conditions, the leakage of nutrients, especially of nitrate, can represent a limitation even when this technology is properly operated. Already in the soil, phosphorus is retained by precipitation with calcium, iron and aluminium. Whereas nitrogen (mainly as NH_4^+) is sorbed onto soil through cation exchange processes. In these forms, nutrients can be stored in the soil and used by plants when necessary. Although the plants recover part of the nutrients, there is the possibility of their leaching towards deeper levels. For example, when NH_4^+ is transformed to NO_3^- it becomes very mobile in the subsurface and, therefore it can reach the aquifer and contaminate groundwater resources. Our research arises from the necessity to overcome this limitation by seeking a sustainable solution that can be easily integrated in a VF. To this end, and based on research published in recent years, we selected two soil amendments, biochar and woodchips, to improve nutrient attenuation in VFs by increasing sorption sites and promoting microbial activity (Meffe et al., 2014).

METHODOLOGY

The soil used in the experimental tests comes from a pilot scale VF that treats wastewater from an office building. The collected calcareous soil is a sandy clay loam soil (50.9% sand, 22.5% silt and 26.6% clay), contains 1.69% of OM and its cation exchange capacity is 11.1 $cmol_c/kg$. A stock solution of synthetic wastewater (SWW) mimicking the real composition of the wastewater used to irrigate the pilot scale VF was produced in the laboratory for the experimental tests. Regarding amendments, woodchips were obtained from the pilot scale VF and biochar by pyrolysis of poplar woodchips. Pyrolysis was carried out in a Microsynth Microwave oven (Batch) from Milestone following the methodology described in Martín et al. (2017).

To investigate the attenuation of nutrients during vadose zone infiltration, three experiments were performed(Martinez-Hernández et al., 2020) using stainless steel columns (L 30.0 cm, Ø 8.49 cm)connected to a vacuum chamber to simulate infiltration under unsaturated conditions. The systems contain natural soil (Column S), natural soil amended with 3% w/w of woodchips (Column S+W) and natural soil amended with 3% w/w of biochar (Column S+B).The porous materials were packed with increments of 2 cm avoiding the formation of stratified layers and preferential flow paths. Vacuum pressure was adjusted to adapt the drainage of the experimental set-up to that observed at field conditions. The irrigation was simulated by manually applying 500 ml (dual pulse of 250 ml each) of SWW at the upper end of the columns once per week. Once a week, SWW and the three column outlets were sampled after every irrigation event (n=12 for each column) to analyse chemical oxygen demand (COD), dissolved ions (NO_2^-, NO_3^-, PO_4^{3-}, Cl^-, SO_4^{2-}, NH_4^+, Na^+, K^+, Ca^{2+}, Mg^{2+}), alkalinity, total nitrogen (TN) and total phosphorous (TP).

The sorption capacity of each amendment was determined in batch experiments following OECD guideline 106 (OECD, 2000). SWW (50 mL) in 100 ml plastic vessels containing 2.5 gamendment was spiked with NH_4Cl at different NH_4^+ concentrations (30, 60, 100, 300 y 600 mg L^{-1}) per triplicate. To measure the current NH_4^+ concentration in the amendments, and to exclude the possibility of NH_4^+ sorption onto the vessels and degradation, control (without amendment) and blank (without NH_4^+) samples were prepared in triplicate and analysed along with the others.After 24 h, samples were collected from each vessel and centrifuged at 4,000 rpm for 20 min to separate the liquid phase from the amendment. The supernatants were then stored at 4°C during 24-48h until analysis.The desorption isotherms were determined placing amendment (50 g) with previously sorbed NH_4^+ (at all concentrations) in contact with 0.01 M $CaCl_2$ (50 mL) solution for 24 h. The distribution coefficient $K_d$wasdeterminedforeachconcentration.

RESULTS AND DISCUSSION

COD, TP and TN removal percentages at the column effluents are shown in Table 1 for the three infiltration experiments.Columns S and S+B exhibit a better performance in terms of COD removal compared to S+W. Although the microbial activity seems to be more developed in the

S+W column (see nutrient removal), only the biodegradable OM is actually treated. Indeed, the obtained results suggest that the addition of woodchips as a source of OM is responsible also for the leachate of non-degradable OM. The addition of woodchips increases both biodegradable and non-biodegradable OM reducing its removal efficiency in terms of COD.

Table1. *Averageremovalpercentages and standard deviationsof allirrigationevents in terms of nutrients and OM.*

	Soil (S)	Soil + Woodchips (S+W)	Soil + Biochar (S+B)
COD	86±3	70±5	88±4
TP	96±1	97±2	95±1
TN	-5±15	85±15	-8±16

Amendments have no effect on TP attenuation.The obtained removal of TP in all cases reaches values ≥95% indicating that this nutrient does not imply a concern in a treatment technology where calcareous soils are present.In fact, the precipitation of orthophosphate PO_4-P in the presence of calcium, which is abundant in the tested soil (~35 g/kg), could explain its removal from the infiltrating water(Duchafour, 1984). TheTP accumulation in the three column soils at the end of the experiment (S: +49 mg/kg; S+W: +76 mg/kg; S+B: +70 mg/kg) corroborates the presence of sorption and/or precipitation processes.

Concerning TN, results were different among the columns. Negative values of removal percentages reported for column S and S+B indicate that instead of attenuation, lixiviation of nitrogen from the inlet and from the soil is occurring. Indeed,at both columns,TN mainly in the form of NH_4^+ in the inlet, is transformed to NO_3^- by nitrification processes occurring during its infiltration(Fig. 1). Different behaviour was observed for the S+W column.Regardless of the predominant nitrogen species at the effluent during the first 9 irrigation events was NO_3^-, its average concentration (N-NO_3: 3.49±2.33 mg L^{-1}) reflects that NO_3^- is further transformed. The removal of NO_3^- can occur by denitrification, dissimilatory nitrate reduction to ammonia (DNRA), anaerobic ammonium oxidation (ANAMMOX) and/or biomass incorporation. DNRA is excluded since there is no increase in NH_4^+ concentrations in the column effluent. ANAMMOX is unlikely to occur since it is a process inhibited when the concentration of OM is high (Nordström y Herbert, 2018) and the S+W column has a high input of OM due to the addition of a labile carbon source. Biomass incorporation may happen as there is a slight increase in N-NO_3 in the amended soil with woodchips (S+W) after irrigation in the upper part of the column (+63 mg/kg). This accumulation also occurs in S (+26 mg/kg) and, to a higher extent in the S+B column (+222 mg/kg) due to its higher sorption capacity. However, it has not an important impact in the removal of N-NO_3. According to approximated balance calculations, the nitrogen accumulated as TKN+N-NO_3 accounts for only 2.61%, 12.13% and 14.35% for columns S, S+W and S+B, respectively.

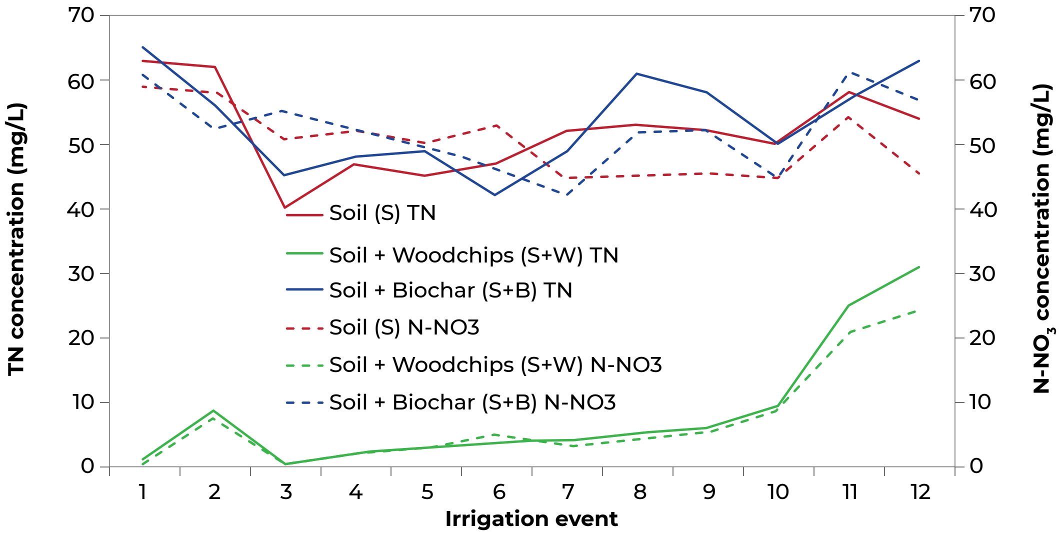

Figure 1. *Totalnitrogen (TN) and N-NO_3concentrations in allinfiltrationexperiments at thecolumnoutlets.*

Therefore, the obtained results indicate that in the column amended with woodchips, NO_3^- is removed by denitrification as corroborated by data measured during the last 3 irrigation events. In the S+W column, there is a decrease of TN removal due the appearance at the column effluent of higher N-NO_3 concentrations (Fig. 1).Data observed during the last 3 irrigation events show a decrease of alkalinity as a consequence of the inhibition of denitrification.This inhibition can be derived from an OM limitation. The C/N ratio and the type of OM are important factors controlling the activity of denitrifier bacteria(Fang et al., 2018, Narkis et al., 1979).At the end of the experiment almost half of the initial OM in the S+W has been consumed, besides the organic carbon that the SWW is supplying.It seems that once all the biodegradable OM is degraded, the denitrification is inhibited and NO_3^- starts leaching. The removal efficiencies of nitrogen are very high (> 90%) when the system has enough biodegradable OM.

Calculated sorption K_d value ranges are 5.76-10.81 L kg^{-1} and 21.35-193.18 L kg^{-1}, for woodchips and biochar, respectively. Results show that sorption of NH_4^+ onto biochar is one order of magnitude higher than onto woodchips being the average percentage of NH_4^+sorbed 2.6 times higher. This difference increases after desorption up to 5.9 times, indicating that biochar is a better sorbent of NH_4^+. Despite biochar sorb more NH_4^+, itsretention capacity is less pronounced when added to soil in a 3% (w/w). The selected amount of amendment is insufficient to render sorption processes as important as the biological attenuation occurring in the presence of woodchips.

CONCLUSIONS

COD is well-attenuated by biodegradation in soil, while biochar only incorporates sorption as an additional sink process. Results suggest that the addition of woodchips increases both biodegradable and non-biodegradable OM reducing its removal efficiency in terms of COD. Amendments have no effect on TP attenuation since sorption and precipitation taking place already in the soil almost completely remove phosphorous species. Woodchip amendments reduce TN leachate by denitrification processes. However, higher content of amendments should be added in order to increase both sorption and degradation processes and to assure a higher durability of the treatment.Obtained results indicates that woodchips used as amendments are more effective than biochar. This labile carbon source has also the advantage that it is easily provided by the VF without further post-processing.

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CONSTRUCTED WETLANDS FOR THE TREATMENT OF WASTEWATER IN SMALL CITIES ON THE PERUVIAN COAST

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J. Morato[1]

Abstract

Peru has the largest amount of the tropical glaciers of the world. Nevertheless, 10% of the population lacks water service and 25.5% lacks drainage (sewerage or sanitation). It's also important to notice that Peru is divided into the coast, mountain, and jungle. Being the coast, the region with the least area but the major population. To wit, Lima, a coast department, has 29.7% of the population of the country. To magnify the problem, 72% of wastewater is treated and discharge to the Pacific Ocean. Thus, it isn't difficult to foresee that this situation is unsustainable. Reuse of treated wastewater could be an excellent alternative to help solve this problem. Accordingly, the current study aims: I) to evaluate the physicochemical and biological quality of UNALM constructed wetland effluent (SDG 6) and II) to know the potentialities of the reuse of this effluent for afforestation irrigation (SDG 11 y SDG 15).

Keywords: Constructed wetlands, wastewater reuse, French system, afforestation.

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INTRODUCTION

In Peru, 10% of the population lacks water service and, 25.5% lacks sewerage service (MVCS, 2017). In addition, more than 70% of the population lives on the coast (Ioris, 2012), where there's a water deficit. The coverage of domestic wastewater in Peru reaches 72% but, if Callao and Lima (capital city) are excluded, the rest of the country only reaches 48% (GWI, 2016). In this framework, is considered to contribute:

SDG$_s$ 6. Target 6.2
"By 2030, achieve access to adequate and equitable sanitation and hygiene for all and end open defecation, paying special attention to the needs of women and girls and those in vulnerable situations".

Moreover, it is worth mentioning that in Peru, the prime technologies used for the treatment of domestic wastewater consist of the combination of primary and secondary facultative lagoons (ANA, 2016).

As of 2018, the MVCS (Ministerio de Vivienda Construcción y Saneamiento) issued a regulation that considers to constructed wetlands for wastewater treatment. One of them is the so-called constructed wetlands (CW), technology that simulates the natural processes of pollutant removal; and is characterized by low cost, easy operation and maintenance, and potential for application in decentralized situations (Dotro, Langergraber, Molle, Nivala, Puigagut, Stein & Von Sperling, 2017). Furthermore, aquatic plants and chemical, physical, and biological mechanisms are optimal for treating wastewater (Von Sperling & Chernicharo, 2005). On the other hand, investigations carried out in wetlands built in high Andean areas, showed potentialities of using treated water for afforestation (Rosado, Paredes, Joachin, Morató, & Rosario, 2019).

However, constructed wetlands require pre-treatment to remove solids. Against this background, vertical flow wetlands have been successfully tested to treat raw wastewater, known as the "French System", and includes two stages: French cell (1st stage) + vertical flow subsurface wetland (2nd stage). Both provide integrated sludge and wastewater treatment in a single system (Molle, Liénard, Boutin, Merlin & Iwema, 2005).

In 2011, an experience with the French System was carried out in a coastal city of 60 inhabitants (Chinc ha-Peru). The results showed that the pollutant removal in the French cell (1st) is more efficient compared to other treatment technologies: septic tank, ABR, Imhoff tank, UASB (Platzer, Hoffmann & Miglio, 2016).

Also, in 2011, 2 pilot plants were built in the facilities of the Universidad Nacional Agraria La Molina -UNALM (Lima-Peru), to promote research work (Pastor, Miglio, Suero, Arias & Morató, 2016). The system consists of 2 lines, and each of them has been dimensioned for 30 PE, which generates a flow of 6 m^3d^{-1} per line. Two types of pre-treatments have been built, an improved septic tank (Baffled Tank or ABR) and a vertical wetland or "French System".

The research results aimed at optimizing the operation of the French System (León, 2020) are shown in this work and are intended to contribute to SDG 6. Goal 6.3, SDG 11. Goal 11.7 and SDG 15. Goal 15.2.

SDGs 6. Target 6.3
"By 2030, improve water quality by reducing pollution, eliminating dumping and minimizing release of hazardous chemicals and materials, halving the proportion of untreated wastewater and substantially increasing recycling and safe reuse globally".

SDGs 11. Target 11.7
"By 2030, provide universal access to safe, inclusive and accessible, green and public spaces, in particular for women and children, older persons and persons with disabilities".

SDGs 15. Target 15.2
"By 2020, promote the implementation of sustainable management of all types of forests, halt deforestation, restore degraded forests and substantially increase afforestation and reforestation globally".

METHODS

Location

The research was carried out in the pilot plant for the treatment of domestic wastewater (PTAR) located at the Universidad Nacional Agraria La Molina (UNALM), Lima - Peru.

This pilot plant has three lines. Line one is made up of the French System - FS (French cell + 02 vertical flow subsurface wetlands).

Work was done on the SF line (Figure 1), that is, on the French cell (1st stage) and the 02 vertical flow subsurface wetlands (2nd stage).

Figure 1. *French System of UNALM. Left: 1st stage (French cell); Right: 2nd stage (vertical flow subsurface wetlands)*

The FS is made up of a French cell (1st stage) and two vertical flow subsurface wetlands (2nd stage) (Figure 2). The domestic wastewater that enters the French System (raw water) comes from a domestic sewerage network and passes through a 24.5 mm opening grate that is manually cleaned. The wastewater is diverted to a pumping chamber (CB1) and is pumped from there to the French cell using two submerged pumps (B1 and B2). The French cell has a surface area of 36 m^2 and is subdivided into two subunits or lines of 18 m^2 of surface each. The two lines operate alternately to guarantee the rest period in each of the lines after a time of use of 72 h (3 d). Each line is fed by one of the pumps; the wastewater enters through two vertical PVC pipes with a diameter of 76.2 mm per line (from 1.1 to 2.2). To ensure an equitable distribution of the wastewater on the surface of each line, the outlet pipes are surrounded by circular concrete plates.

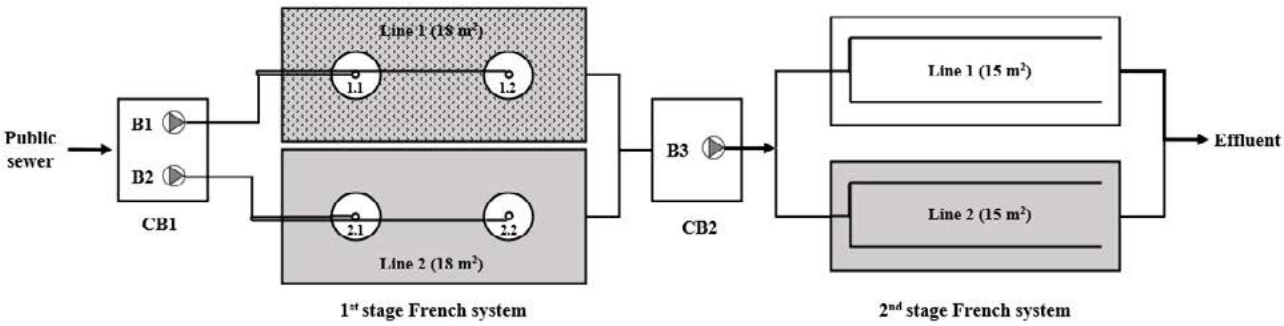

Figure 2. *French System Scheme*

After passing through the filtering material of the 1st stage, the wastewater is collected through perforated drainage tubes located in the lower part of the filter (diameter of 110 mm), which discharge by gravity to the pumping chamber CB2, from where they are pumped with a submerged pump (B3) to the 2nd stage of the French System. The B3 pump, unlike the other pumps, is controlled by a float to maintain a constant level of water in the pumping chamber. The 2nd stage of the FS has a surface area of 30 m^2, subdivided into two units or lines of 15 m^2 of surface each. The treated water enters and is distributed uniformly over the entire surface through pipes, with a diameter of 50.8mm; which are diametrically opposite perforated. After passing through the filter medium, the residual water is captured by drainage tubes (diameter of 101.6 mm) located at the bottom of the cell and finally, the treated water is discharged.

The 1st stage of the French System and one of the subsurface vertical flow wetlands of the 2nd stage have been considered. In Figure 1 the systems studied are shaded in lead colour. The 1st and 2nd stages of the French System are planted with umbrellas (*Cyperus alternifolius*) and vetiver grass (*Chrysopogon zizanioides*), respectively.

The B1 and B2 pumps are controlled by an automated SCADA (Supervisory Control and Data Acquisition) system to ensure a pumping and resting sequence.

The 1st stage has a total filter bed depth of 1m and is divided into three layers of different filter material. From top to bottom, these layers are: 0.6 m gravel Ø 4.75 - 19.0 mm; 0.25 m transition layer Ø 12.7 mm crushed stone; 0.15 m drainage layer Ø 25.4 mm crushed stone. The 2nd stage of the French System has a total filter bed depth of 0.9 m and is divided into 3 layers of different material filters. From top to bottom, these layers are: 0.1 m gravel Ø 4.75 - 12.5 mm; 0.6 m transition layer sand Ø 9.53 mm; 0.2 m drainage layer Ø 4.75 - 12.5 mm (Rotaria del Perú, 2012).

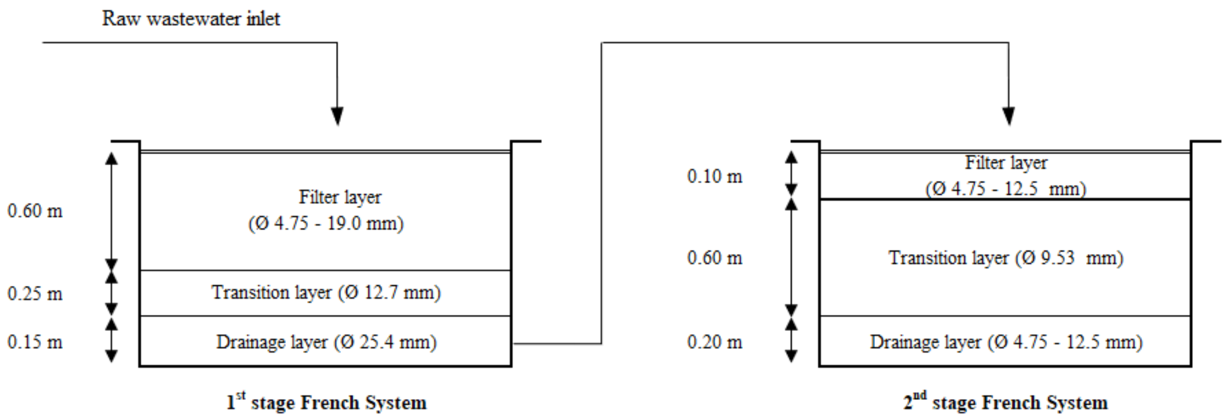

Figure 3. *Cross-sectional view of the French System installed at UNALM by Rotaria del Perú (2012)*

Operating loads

Four hydraulic load increments were applied to the 1st stage of the SF, with a duration between 6 - 12 weeks each. Table 1 shows the 4 applied load campaigns. In each campaign, B1 and B2 ran alternately for three days each, so that line 1 and line 2 were fed for three days and then rested for the same time.

Throughout the investigation, the system was fed with raw domestic wastewater in hydraulic batches of 6 minutes each. The hydraulic load increased by increasing the number of batches per day, starting 3 batches d-1, and increasing to 6, 8, and 12 batches d^{-1}.

Table 1. *Research period for 4 campaigns*

Campaign	Station	Date	Research period (week)	Batch duration (min)	Batch number (d^{-1})	Pumping time (min d^{-1})
I	Spring	Oct - Nov 2017	6	6	3	18
II	Summer	Dic - Jan 2017	8	6	6	36
III	Summer	Feb - May 2017	12	6	8	48
IV	Winter	Jul- Aug 2018	8	6	12	72

Parameters analysed

The temperature (T) and pH parameters were measured in the field. Chemical Oxygen Demand (COD), Biological Oxygen Demand (BOD_5), Total Suspended Solids (SST), Turbidity, Total Nitrogen (N_{Total}), Ammonia Nitrogen (NH_4-N), Nitrate (NO_3- N), and total phosphorus (PO_4-P) were analysed in the Sanitation and Environment laboratory of UNALM. NO_3-N was not analysed in the influent samples, since the raw wastewater was assumed to be anaerobic. Thermotolerant coliforms (CT) and helminth eggs (HH) were analysed in an accredited external laboratory.

RESULTS AND DISCUSSION

Characterization of untreated wastewater

Table 2 shows the characteristics of the concentrations of raw wastewater during the 4 research campaigns (October 2017 to August 2018).

The average pH value of the influent wastewater was 7.5, which is slightly alkaline, and the average temperature was 24.9 °C. The average concentration of the N_{Total} was 55.9 mg L-1, and the NH_4-N represents 70.3% of the N_{Total}, which is 29.7% of the nitrogen that enters the 1st stage was in organic form. The average PO_4-P concentration was 10.1 mg L^{-1}. The average concentrations of COD and BOD_5 (biological oxygen demand) were 699.9 mg L^{-1} and 344.3 mg L^{-1} respectively, which turn in a BOD_5 / COD ratio of 0.49 that indicates that any biological treatment can be applied to wastewater (Von Sperling & Chernicharo, 2005). The COD, BOD_5, SST, and Turbidity present high standard deviation, which means the data is dispersed concerning the average, and therefore there is a higher standard error.

Table 2. *Characteristics of the concentrations of raw wastewater during the 4 research campaigns.*

	Unit	Concentration
COD	mg L^{-1}	699.9 ± 276.6
BOD_5	mg L^{-1}	344.3 ± 149.7
SST	mg L^{-1}	584.9 ± 471.9
N_{Total}	mg L^{-1}	55.9 ± 14.0
NH_4-N	mg L^{-1}	39.3 ± 8.3
PO_4-P	mg L^{-1}	10.1 ± 2.8
Turbidity	NTU	476.6 ± 239.5
pH	-	7.5 ± 0.5
T	(°C)	24.9 ± 2.4

Hydraulic loads

The hydraulic load results applied to the 1st and 2nd stage of the French System are shown in Table 3. For practical purposes, the hydraulic load applied in the 1st stage was the average of the two lines; instead, the hydraulic load applied in the 2nd stage was only the one calculated in line 2 (see Figure 2).

The hydraulic loads applied in campaign 3 for the 1st and 2nd stages were 0.329 m d^{-1} and 0.395 m d^{-1} respectively. These values are similar to those proposed by Dotro, Langergraber, Molle, Nivala, Puigagut, Stein, and Von Sperling (2017) (maximum design hydraulic loads) related to one of the active lines in the 1st and 2nd stage under temperate conditions (0.37 m d^{-1}).

The small variations in hydraulic load applied in the 1st stage are because the power of the pumps was not controllable. The pumps were operating at maximum capacity during the operation periods. Although the pumps manufactured are similar, they do not generate the same volumetric flow generating different hydraulic loads for each of the lines. Other factors were the pipe connections and valves of the installed system.

Table 3. Hydraulic loads applied to the 1st stage (French cell) and line 2 of the 2nd stage (vertical flow subsurface wetland) of the French System.

Campaign	*Unit*	**1st stage**			**2nd stage**
		Line 1	**Line 2**	**Average**	**Line 2**
I	m d^{-1}	0.125	0.113	0.119	0.143
II	m d^{-1}	0.254	0.247	0.251	0.301
III	m d^{-1}	0.338	0.319	0.329	0.395
IV	m d^{-1}	0.449	0.483	0.466	0.589

Treatment efficiencies

Table 4 show the pollutant removal efficiencies of the 1st and 2nd stages, and the whole system (1st + 2nd stage) during the 4 research campaigns. The removal efficiencies in the 1st stage were > 79.9%, > 83.3%, > 96.0%, > 58.2%, > 53.0%, and > 39.3% for COD, BOD_5, SST, N_{Total}, NH_4-N, and PO_4-P respectively. The results obtained are higher than those registered by Molle, Liénard, Boutin, Merlin, and Iwema (2005) for moderate climates and similar to those registered by Lombard and Molle (2017) for temperate climates regardless of the operating hydraulic load.

The removal efficiencies in the 2nd stage were > 84.5%, > 76.7%, > 79.2%, > 38.1%, > 81.6%, and > 22.5% for COD, BOD_5, SST, N_{Total}, NH_4-N, and PO_4-P, respectively. The results obtained are higher than those recorded by Molle, Liénard, Boutin, Merlin, and Iwema (2005) for moderate climates and lower than those recorded by Gomez (2017) for temperate climates, regardless of the operating hydraulic load.

The removal efficiencies of the whole system (1st + 2nd stage) were > 97.7% in all Research Campaigns for COD, BOD_5, and SST, and NH_4-N > 91.5%, these results are even higher than those registered by Molle, Liénard, Boutin, Merlin, and Iwema (2005) for moderate climates and similar to those established by Platzer, Hoffmann, and Miglio (2016) for temperate climates regardless of the operating hydraulic load. Likewise, for the N_{Total} and PO_4-P they present significant efficiencies in all campaigns > 76.2%, without considering the result of PO_4-P in campaign 4.

However, during field observations, it was noted that in campaign 4 of research, there were signs of clogging and slower passage of residual water over the filter in the 2 stages of the French System. These indications must be confirmed by working with a longer operating time under the operating loads indicated in each investigation campaign.

Table 4. *Efficiency of removal of pollutants from the whole system (1st + 2nd stage) during the 4 research campaigns.*

	Campaign	Affluent	Effluent 1st stage	Effluent 2nd stage	Efficiency 1st + 2nd stage
		mg L^{-1}	mg L^{-1}	mg L^{-1}	%
COD	I	497.6	100.0	10.3	**97.9**
	II	827.6	92.1	11.2	**98.6**
	III	856.3	81.2	12.6	**98.5**
	IV	465.1	73.1	10.6	**97.7**
BOD_5	I	254.1	42.5	5.0	**98.1**
	II	546.0	38.2	7.9	**98.6**
	III	372.0	29.5	6.9	**98.2**
	IV	220.1	28.4	3.1	**98.6**
SST	I	378.2	15.2	1.8	**99.5**
	II	540.3	15.9	3.3	**99.4**
	III	1019.2	19.0	1.5	**99.9**
	IV	207.3	8.3	1.6	**99.2**
N_{Total}[3]	I	55.0	23.0	10.0	**81.8**
	II	65.5	23.7	10.2	**84.5**
	III	64.2	19.0	9.8	**84.8**
	IV	47.0	18.1	11.2	**76.2**
NH_4-N	I	42.8	20.1	3.0	**92.9**
	II	42.6	19.6	3.6	**91.5**
	III	37.0	16.2	1.8	**95.2**
	IV	36.7	16.7	2.6	**92.8**

[3] N_{Total}: 10 mg N.L^{-1}, maximum allowable value (VMA) for recharge of aquifers by localized percolation through the ground or by direct injection. RD 1620/2007 Legal regime for the reuse of treated wastewater in Spain.

PO_4-P	I	7.2	4.3	1.6	**77.8**
	II	10.2	4.5	1.2	**87.9**
	III	12.6	6.2	2.9	**76.6**
	IV	8.2	5.4	4.2	**49.1**

Wastewater reuse in forestry

The Urban forest can be considered a goods solution (Randrup et al., 2020) which has several benefits: regulating urban microclimates, filtering air pollution, providing shade, capturing CO_2, and regulating temperature. Within the framework of SDGs 11, they can help citizens have access to safe, inclusive, and accessible green areas and public spaces for the development of social, cultural, and sports activities. However, as can be seen in Figure 4, in Lima, since 2005, more than 200ha have been reforested each year but from 2015 to 2018 this number has decreased considerably.

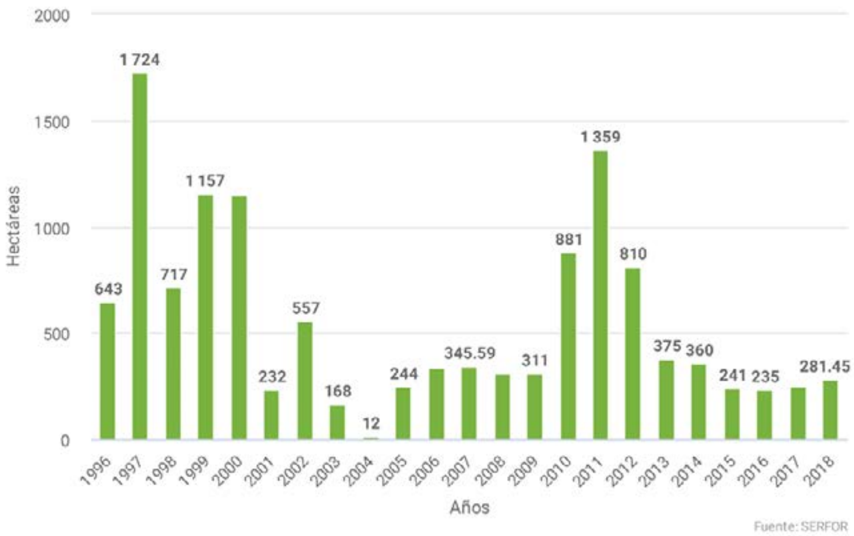

Figure 4. *Area reforested annually in the city of Lima-Peru. Source: MINAM, 2021.*

On the other hand, the urban forest requires “a planned development and maintenance approach” (Schwab, 2009), where water is the key element for its proper functioning. Although reforestation is undoubtedly beneficial, each tree consumes water and in this coastal city water is not an abundant resource. That is why it would be optimal to use treated wastewater for irrigation of afforestation in the city of Lima.

Evaluation of microbiological quality for the reuse of wastewater in forestry

Table 5 shows the microbiological quality parameters for irrigation reuse purposes. The results in the tributary fluctuate between $2x10^8$ - $4x10^8$ NMP mg L^{-1} for the CT, and at the same time, these are similar to those found by Platzer, Hoffmann, and Miglio (2016) for a similar design of the French system under the same climatic conditions. Likewise, the HH fluctuates between 205 - 2200 N ° L^{-1}.

The high removal of CT in the 1st and 2nd stages of the French System is due to sedimentation and filtration in the first 20 cm of the filter medium (Sleytr, Tietz, Langergraber & Haberl, 2007) and the elimination of HH, possibly due to the porosity of filters (Jimenez, 2007).

Table 5. *Characteristics of the concentrations of thermotolerant coliforms (CT) and helminth eggs (HH) in wastewater treated in the SF during 4 campaigns.*

	Campaign		effluent	Effluent 1st stage SF	Effluent 2nd stage SF	WHO regulations Restricted irrigation	WHO regulations Unrestricted irrigation
CT	I	NMP ml^{-1}	$2x10^8$	$4.5x10^5$	$2.5x10^3$	≤ 1	≤10^3
	II	NMP ml^{-1}	$4x10^8$	$6.7x10^6$	$3.4x10^3$		
	III	NMP ml^{-1}	$3.4x10^7$	$3.9x10^6$	$1.6x10^4$		
	IV	NMP ml^{-1}	$2.8x10^6$	$1.6x10^6$	$4.7x10^3$		
HH	I	N° L^{-1}	2200	0	0	-	≤ 1
	II	N° L^{-1}	485	0	0		
	III	N° L^{-1}	205	0	0		
	IV	N° L^{-1}	113	0	0		

CONCLUSIONS

The characterization of the physical, chemical, and microbiological properties of the effluent from the "French System" as a whole (1st + 2nd stage), indicates an efficiency superior to 90% of elimination of COD, BOD_5, N_{Total}, P_{Total}.

Regarding the N_{Total}, the values (9.8 ± 2.9 - 11.2 ± 2.1 mg N / L) indicate that the effluent can have an environmental use both for recharging aquifers by localized percolation through the ground and by direct injection (RD 1620/2007 - Catalunya). This indicates that a post-treatment would not be necessary for its use in forest irrigation and that it would not affect the quality of the aquifers.

The microbiological evaluation carried out during 4 campaigns showed values of Thermotolerant Coliforms (1.6 $x10^4$ - $7x10^3$ NMP ml^{-1}) and of Helminth Eggs equal to 0 N ° egg L^{-1}, showing that French System produce AR with optimum quality for use in afforestation irrigation.

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APPLICATION OF FLOATING WETLAND ISLANDS FOR WATER AND HABITAT PROMOTION IN TWO CONTEXTS: URBAN RIVER AND SMALL FISH FARM

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Abstract

Despite the proven vital importance of freshwater ecosystems for humanity, those continue to be subjected of accelerated ecological degradation. Floating wetland islands (FWI) - one of bioengineering technologies classified as nature-based solutions – have shown ability to assist the reduction of nutrient concentrations, improving water and habitat quality for wildlife. Therefore, if properly used, FWI can be important tools for assisting the sustainable management and the rehabilitation of these ecosystems. Herein, are presented two proposals for FWI installation: one concerning the water and habitat quality improvement of an urban river section (Case 1); the other aiming the reduction of the small fish farm outflow impact on downstream water quality and the improvement of reared fish welfare (Case 2).

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INTRODUCTION

Societies and ecosystems are highly dependent on water resources. Therefore, a strategic water management is critical to achieve sustainable development, in all its dimensions (UNESCO, UN-Water (2020)). When considering specific water bodies that are subject to certain types of contamination or pressures, like some rivers, lakes, and aquaculture production sites, the water quality assessment and mitigation plan pose a major concern. It is thus important to find sustainable solutions to pollution mitigation and biodiversity promotion. Several contaminants can be biologically degraded, or uptake by plants, through the application of phytoremediation technologies, such as wetland systems (constructed or floating) (Calheiros 2015, 2020). Floating wetlands islands (FWI) are an innovative variant of the traditional constructed wetlands, being a man-made ecosystem intending to mimic the depurative processes that naturally occur in wetlands. Their main applications have been related with the treatment of stormwater, sewage, eutrophic lake water and water supply reservoirs (e.g., Colares et al, 2020; Bi et al, 2019). However, there is a great unexplored potential for FWI application in other situations, such as urban rivers and fish farms.

Urban rivers and streams are crucial to cities, because they provide environmental, cultural and aesthetical services that are essential to maintaining urban environmental quality (Hua & Chen, 2019). Nevertheless, in urban areas there is a range of stressors impacting the integrity of these ecosystems, namely high nutrient levels and contamination resulting from point and diffuse loadings arriving from several sources. Besides, many of the so called “urban river rehabilitation actions”, carried out either the past, either recently, promoted the regularization of the riverbeds and the riverbanks leading to riparian wood elimination. The environmental consequences of fish farming can be potentially negative. Reduction in dissolved oxygen, increase in biochemical oxygen demand (BOD) and in nutrient concentrations downstream are the main consequences if fish farms are carried out without effluent treatment (Fidalgo, 2002; Crispim et al 2009). Besides, another issue is fish welfare (Braithwaite and Salvanes, 2010).

The main objective of this study is to present two proposals for FWI installation: one concerning the water and habitat quality improvement of an urban river section -Fervença River- (Case 1); the other aiming the reduction of the fish farm outflow impact on downstream water quality and the improvement of reared fish welfare (Case 2).

METHODS

Case 1: The Fervença River is located into Portuguese part of the Douro basin (latitude: 41°47’N; longitude 6°46’W), is about 25 km long, and it is regarded as an urban river, because flows through the city of Bragança (21 853 inhabitants in 2011). Along its course it encounters nonpoint sources of pollution, originated by agricultural activities, and point sources of pollution from some villages located in the vicinity. Further downstream, in the core urban area, along 630 m riverbed was regulated with five weirs, an artificial concrete riverbank was created and the riparian wood was partially eliminated (Fig 1).

Figure 1. *Fervença river section located in the core urban area (A-C). Point source of pollution (D); algal summer blooms (E,F).*

Case 2: Posto Aquícola de Castrelos, is a small fish farm unit of governmental services: the Portuguese Conservation of Nature and Forest Institute (ICNF). Located in northeastern Portugal (41 50’ N; 6° 53’W) in the right bank of Baceiro River, a mountainous oligotrophic river, is mainly used to rear brown trout (*Salmo trutta*) and endemic cyprinid for fish stocking and conservation. Fish are grown in outdoors concrete tanks, in a flow through system, and the effluents are discharged untreated directly into River Baceiro (Douro basin) (Fig 2).

Figure 2. *Posto Aquícola de Castrelos facility (the red arrow points the outdoor tanks)*

For both cases, the following steps were carried out: i) Diagnosis of the current environmental state; ii) Evaluation of which areas could be targeted for the installation of FWI; iii) Presentation of a preliminary proposal for the installation of FWI. The necessary information supporting both proposals was obtained from literature.

RESULTS AND DISCUSSION

Case 1: Large variation range was observed in all studied parameters related to water quality. The concentrations of total phosphorus and phosphates were the highest in the summer when the flow was reduced (Nogueira, 2020). In this area, mainly during the summer, algal blooms, generally occur. The water quality as well as the visual quality of the landscape is negatively impacted when these blooms occur (Fig 1). Nevertheless, this river section still supporting fish species, including endemic species of river Douro basin, as well as water birds and, amphibians. Besides, the otter (*Lutra lutra*) occasionally visits this river section. Therefore, the main goal of the present plan is to recreate environments for habitat, assist pollution and algal blooms reduction in the river and ultimately to promote recreational and educational activities in the area. According to the available literature the FWI installation could allow to reach the mentioned goals. However, the present proposal should consider the following issues:

a) FWI should be placed along the urban section of the river nearby the fully artificialized riverbank in order to increase habitat availability and to prevent the dragging by the high winter river flow;
b) The local ecotypes of macrophytes, whose ability to remove nutrients is well known (e,g., *Juncus effusus*, *Iris pseudocorus*, *Typha* sp. and *Phragmites australis*) should be preferentially used.
c) The extent of coverage area, in a first approach, should be at least 10% of urban riverbed area (e.g., Grosshans et al, 2019). Nevertheless, a future design study taking into account different scenarios of polluting loads and expected removal rates should be carried out in order to adjust the extent of surface coverage.
d) Vegetate the vertical concrete wall using vertical fencing structures, allowing the future installation of local riparian shrubs and trees. This will introduce new hydrological patterns promoting sediment retention and new habitats.
e) Periodical and long-term monitoring should be essentially focused on (1) the integrity of the beds and monitor the development of vegetation .and of the associated biofilm; (2) the effects of FWI on water quality and aquatic ecosystem integrity methods (e.g., nutrient concentrations macroinvertebrate and diatoms indices); (3) monitoring fish, amphibian and bird populations;
f) Involvement of municipality and other stakeholders;
g) Promotion of educational activities in schools and in other public places concerning the importance of freshwater ecosystems for humanity.

The expected environmental and aesthetic outcomes are presented in Fig. 3. Socio-Economic outcomes are also expected (more walkways, bring more life to this area of the town, revitalize small business along the river edge and more educational, cultural and recreational activities along the urban riverine area.

Figure 3. *Simulation the environmental and aesthetic effects of FWI placement.*

Case 2: Increase on BOD, ammonia, total phosphorus and chlorophyll a and suspended solids was observed by Fidalgo (2002) in the riverine sampling site located near fish farm outlet discharge. Nevertheless, such effects were not persistent for a long distance downstream. Since the "modus operandi" of this facility did not change in time it is expected that mentioned effects still occurring, nowadays. Besides, mainly in summer, when water temperatures are higher, fish mortality in outdoor tanks can be significant. Recently, in a very preliminary way, FWI and fluctuating macrophytes were placed the tanks to create refuge for fish and to control water temperature in summer (Fig 4). Crispim et al (2009) verified that macrophytes and root associated biofilm could be a valuable to preventing eutrophication in small-scale fish farming. Therefore, the implementation FWI in Castrelos should be design both in order to create adequate refuge for fish and to effectively remove the nutrient excess, preventing negative impacts in water quality of tanks and in Baceiro River. Nevertheless, before FWI implementation research should be carried out in order to evaluate:

a) The adequate macrophyte coverage area to effectively remove nutrients considering food requirements, fish density and N and P excretion rates;
b) The adequate macrophyte coverage area to effectively increase fish welfare;
c) How fish can influence the growth of plant root associated biofilm;
d) The efficiency of the combination of submerged and /or fluctuating plants with FWI.

The expected outcomes with the FWI installation are the reduction in mortality and the increase of fish condition due to the improvement of water and habitat quality leading to better levels of welfare.

Figure 4: *Tanks with fluctuating macrophytes (A) and (B) with FWI in February 2021.*

CONCLUSIONS

FWI are nature-based solutions that provide an array of ecosystem services, and are characterized by being multifunctional. As both case studies had shown, FWI can be used in different scenarios, being extremely flexible technologies, assisting to the sustainable water resource management, uploading the depuration capacity, improving water, habitat and landscape quality and ultimately promoting biodiversity or, in case of fish farming, fish welfare. Nevertheless, there is still lack of knowledge concerning ecological fundamentals of aquatic ecosystems and fact that “each aquatic ecosystem has its own nuances” can limit the use and the efficiency of this technology if previous research and good monitoring design are absent.

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SUPPORT STRUCTURE FOR HELOPHITE AQUATIC PLANTS. AQ3M +

Ownership: **QUARQ ENTERPRISE SA (100.0%)**
Inventor: **CARBONELL ESPIN, Francisco Javier.**
Utility model: **ES1225763**

1 DESCRIPTION

Support piece for helophyte aquatic plants (AQ3M +, MODULE or TILE).

2 FEATURES

– Measurements: 720 x 588 x 62mm.
– Opening between pieces: 1 mm.
– Volume: 936.1 cm^3.
– Material: Polypropylene + expander.
– Allowable planting density (44 plants per square meter).
– Weight: 633 g / piece.
– Density: 0.68 g / cm^3.
– Maximum weight supported: 303 g per piece
– T-section.

3 TECHNICAL SECTOR

The present invention refers to the support for helophyte aquatic plants, intended for the natural purification of water.

The support structure is formed by a plurality of modules laterally coupled to each other to form a laminar surface that is placed on top of the water to purify it.

The object of the invention is to provide a modular structure, which is characterized by its low weight, rigid and flexible thanks to the collateral coupling between modules.

4 BACKGROUND OF THE INVENTION

As is known, helophytic aquatic plants are used for the purification of water being situated on floating structures.

There is a suspension system for helophyte aquatic plants, based on modules that can be coupled. Where each module includes three parts, one corresponding to the floating structure itself, another corresponding to the supports of the receptacles where plants are to be arranged helophyte aquatic plants, and another corresponding to said receptacles. In such a way that the manufacture of these modular systems is expensive, regardless of the problem that results in stacking, transporting, storing and even handling.

In the patent of invention ES 2490515, owned by the applicant himself, a series of improvements is described on floating structures for helophyte aquatic plants. Improvements that are fundamentally based on the fact that each module is a one-piece body, including perimeter and in certain parts of it, complementary inter-coupling elements, to form a structure with the desired dimensions to place it on the water, and the seedlings directly on this structure.

The problem of this system is that the inter-coupling between modules is rigid, which in situations of waves, strong winds, etc., can affect the structure of the modules, damaging them, and may eventually split due to fatigue.

Furthermore, the structure of each module is tubular in order to achieve an efficient and correct buoyancy, which obviously implies an excessive expenditure of material in the face of non-tubular structures.

5 EXPLANATION OF THE INVENTION

The proposed structure, based on the modular suspension structure of helophyte aquatic plants for natural purification of water described in the patent of ES 2490515, presents a series of improvements from which substantial advantages and new benefits.

For this, and more specifically, the new system presents the particularity that instead of being tubular, it presents a “T” section, with the consequent material savings, achieving high buoyancy as a consequence of the material, polypropylene base, internally incorporates an expander, which allows optimal buoyancy even though the section is in “T”.

Another novelty feature is that the inter-coupling means established between modules are mobile, for which it has been envisaged that some sections have cylindrical portions in which complementary elements fit as a clamp provided in other sections of the perimeter of each module, forming a tilting joint, like a hinge, which confers flexibility to the whole. In this way, it is possible to mitigate risk of breakage of the structure in the event of inclement weather, waves, or the very tension that the development of the plant supposes on the plastic structure itself. All of this in such a way that by virtue of this structure, mobility between modules makes it possible to establish rotation angles between modules, improving buoyancy and implying greater security that the aerial part of the plants will remain above the level of water, mitigating the risk of gaps.

Likewise, the admissible weight of the plants is increased at the time of planting, which allows the use of more developed plants or with a larger root ball, which limits the risk of gaps and increases the resistance of the plant to stress derived from transplantation, also achieving a higher performance of the installation in less time.

Finally say that the section in “T” of the sections that form the structure of each module sees increased resistance of said structure with a lower weight of it.

Besides, it should be noted that the structure of each module includes two floats that allow the density to be decreased and the structure’s resistance to breakage increased.

6 DESCRIPTION OF THE DRAWINGS

To complement the following description and in order to help a better understanding of the characteristics of the invention (Patent), according to a preferred example of practical realization, a set of plans where, with an illustrative and non-limiting nature, the following has been represented:

– **Figure 1.-** Shows a perspective view of the structure of a support with seedlings located on the receptacles that incorporates the structure, all of this made in accordance with the object of the present invention.
– **Figure 2.-** Shows a detail from the lower part of one of the vessels and the inter-coupling system between supports.
– **Figure 3.-** Shows a plan view of several modules inter-coupled with each other.
– **Figure 4.-** Shows a perspective view of a detail corresponding to the inter-coupling between modules.
– **Figure 5.-** Shows a perspective view of the intercoupling system of the previous figure.– **Figure 6.-** Shows a perspective view corresponding to the angulation that two supports or modules can form with each other.

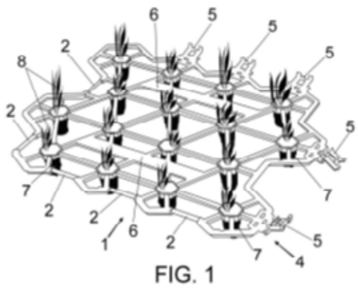


FIG. 1

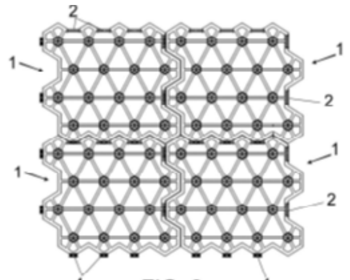


FIG. 3

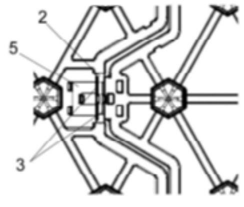


FIG. 5

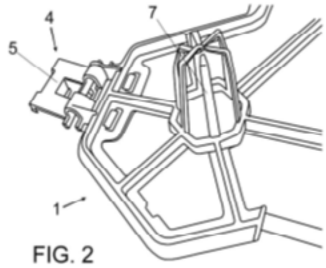


FIG. 2

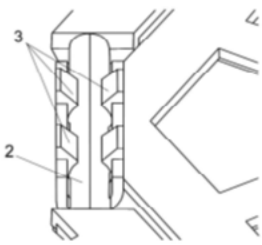


FIG. 4

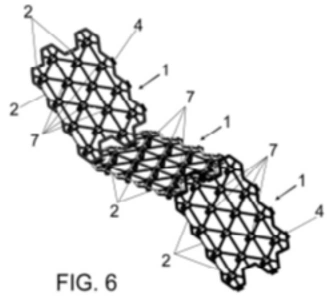


FIG. 6

7 PREFERRED EMBODIMENT OF THE PATENT INVENTION

In view of the figures outlined, it can be seen how the support for helophyte aquatic plants for natural water purification is intended to participate in a system that comprises a plurality of modules (1), made in one-piece bodies obtained by polypropylene injection with an expander inside, presenting a reticular structure based on “T” section profiles.

Each module or one-piece body has a hexagonal mesh-like configuration, including on certain sides of the perimeter cylindrical sections (2) that fit together in forks (4) as wedges, being retained by teeth (3) established in the upper ends of said forks (4), said forks (4) and cylindrical sections (2) being established on opposite and complementary sides in the formation of the mesh from the modules, as clearly represented in figure 1, obtaining thus a flexible and mobile inter-coupling between modules (1), said inter-coupling being secured by the inclusion of the forks (4) of folding closing elements (5), visible in Figures 5 and 6, which completely close the mouth of said forks (4) once the cylindrical sectors (2) have been inserted.

Finally, it only remains to point out that each module includes a pair of floats (6), counting with the appropriate receptacles (7) where the root balls of the helophyte aquatic plants are arranged (8), so that they can purify the water.

8 CLAIMS

1st.- Support for helophyte aquatic plants, usable in water purification and that in combination with other supports of the same type, inter-coupleable collaterally with each other, allow to form a floating structure carrying a series of vessels (7) for aquatic plants helophytes, the set presenting a mesh-like structure that is arranged on the water, each support or module constituting a one-piece body made of injected plastic, equipped on its perimeter with complementary elements for inter-coupling with other modules, characterized in the support is made of polypropylene with an expander, presenting a structure based on "T" section profiles, and equipped on the perimeter with cylindrical sections (2), intended to be coupled on wedges or forks (4) established in perimeter sections, opposite and complementary, defining a means of articulated coupling between modules (1).

2ª. - Support for helophyte aquatic plants, according to claim 1, characterized in that the structure of each module (1) includes a pair of floats (6).

3ª. - Support for helophyte aquatic plants, according to claim 1, characterized in that the wedges or forks (4) include at their upper end teeth (3) for retaining the cylindrical sections (2), as well as folding closing elements (5) for its embouchure.

9 COMPLIANCE WITH THE MINIMUM REQUIRED CHARACTERISTICS OF THE PIECE AND FLOATING STRUCTURE IN ORDER TO BE A NEW TECHNOLOGY OUTSTANDING SMAGUA 2016

Each support piece or support module of the AQ3M + floating structure allows the stability and buoyancy of the seedlings during the initial phases of their development, being a single piece that, once assembled to the others, achieves flexibility, ergonomics and variable rigidity, sufficient to resist any inclement weather.

This piece includes the system for fixing the plants, which facilitates their placement, the growth of the plant upright and that of its rhizome by breaking the flooded part of the vessel of the structure in a natural way with the only root growth of the helophyte.

The assembly of the individual modules (tiles) is sequential and occupies the entire surface of the water sheet to avoid preferential flows.

The union between tiles is mobile, hinge type, thus achieving the necessary flexibility in the structural assembly, mitigating the risk of breakage of the floating structure in the face of inclement weather, waves, and variations in the water sheet or the tension that the development supposes of the plant on the plastic structure. AQ3M + additionally has notches, necessary for the orderly positioning of cable ties as a safety reinforcement.

The module is reticular allowing the passage of light and solar radiation to the rhizomes of the plants.

It includes an expander in the material of its composition to reduce the density of the polypropylene structure without reducing the resistance to breakage.

It includes floats that decrease the density of the piece and increase its resistance to breakage, implying greater security so that the aerial part of the plants remains on the water sheet, mitigating the risk of gaps and increasing the admissible weight of the plant at the time of its placement in the support piece, which allows the use of more developed plants or those with a larger root ball, mitigating the risk of gaps and increasing the resistance of the plant to the stress derived from transplanting the plant to the water surface.

The module has a solid section and an expander and its T-shaped cross section increases the resistance of the structure with a lower weight.

10 SUMMARY AND CONCLUSIONS

– **AQ3M +** JOINS IN 10 POINTS WITH CLICK PRESSURE HITCH AND HINGE.
– **AQ3M +** HAS THE SUPPORT OF PLANTS INCORPORATED TO THE STRUCTURE AND WITH BREAKING THE SAME AT THE GROWTH OF THE PLANT TO FAVOR ITS VERTICALITY AND RADICULAR JOINT.
– **AQ3M +** IS SOLID with expander, T-PROFILE and high buoyancy as it has two floats integrated into the structure itself.
– **AQ3M +** Covers the entire sheet and allows tilting without breakage of up to 45° in both directions.
– **AQ3M +** It is stackable. Quick and easy to assemble.
– **AQ3M +** Allows up to 16 plants per piece, which allows a rapid formation of the sieve.

PROCEDURES FOR THE PRODUCTION OF AQUATIC PLANTS FOR THE PURIFICATION OF POLLUTED WATER BY MEANS OF POLLUTED WATER BY FLOTATION SIEVES OF HELOPHYTES OF THE GENUS THYPA

Property: QUARQ ENTERPRISE SA (100.0%).
Author: CARBONELL ESPIN, Francisco Javier.

1 BACKGROUND

Much is known about the optimal seed germination conditions of typha genus species, however, not so much in terms of plant production processes and there are some direct planting protocols. In such conditions a significant amount of seeds are lost and plants obtained from root deficits.

2 OBJECT

It is the establishing the procedure for the production of aquatic plants for the purification of contaminated water, of the genus Typha, as raw material applicable to the water bio depuration sector by sieves of helophytes in flotation, in a simple, effective, economical way, at any time of the year and anywhere in the world.

In the pictures, you can see three plants selected based on their plant growth, root and infertility and higher resistance to cold.

3 PROCESS OF CONSERVATION, GERMINATION SEED AND PRODUCTION OF PLANTS

The procedure that is preconscious allows obtaining in a simple and economical way plants of the genus Typha spp. It based on an effective and rapid process of seed germination, obtaining plants with good root development and aerial development, with lower needs of time and nutrients, and with a lower cost in obtaining the plants finally, in order to make it extendable to any species of genus typha and develop it anywhere in the world.

The procedure includes:

– **Conservation** of seeds intended for the industrial production of Typha seedlings.

– **Germination** of seeds for the industrial production of Typha plants.

– **Root diversification** for industrial production of Typha plants.

– Empowering **foliar development** in the early stages of Typha seedlings.

<u>**Conservation**</u>

Seed conservation conditions are protocolized according to standard and basic environmental conditions for seed stabilization and durability over long shelf life periods and which can reach more than 30 years.

In the case of Typha seeds, found in a membranous fruit with a feathery appendix called “ginoforo”, mechanical removal of the membrane and ginoforo is projected with a sieve system. Once the seed has been removed from the fruit, the process of preparing the seed for its conservation, proceeding at first to a prior drying of the seeds once their maturity and viability have been confirmed with germination percentage above 80%. Drying occurs under controlled temperature conditions, between 20 and 25°C and 2% humidity, under an air current with an average speed of 3-8 m/s. The drying process ends when the seeds reach a moisture value below 6%.

The seeds dried in the conditions and with the preparations mentioned, are placed in airtight, opaque bottles, in an amount ranging from 500-600 g/bottle, allowing an average of between 18-25 x 10^6 seeds/bottle.

These bottles with dried seeds are stored in a chamber where environmental conditions are controlled, with a temperature at 5°C and a constant humidity below 20%.

Germination

The seed germination stage includes first the rescue of stored seeds, obtaining seeds in number of 1.5 times the expected needs since the average weight of the seeds is around 0.02 mg. The extracted seeds are passed through a laminar flow chamber to proceed with their irradiation, in order to avoid the presence of germs that prevent germination or contaminate the materials. The seeds then temper the prevailing environmental conditions, which in no case can be found outside a 35% to 60% humidity, 15° to 25° C, and a brightness between 1,000 and 2,500 lux. In these conditions the seeds are submerged for 24 hours to moisturize them and move on to the germination process, developing wetting in slightly saline water and with pH between 6.5-7.5.

Once the seeds are moistened, the seeds are then forced to germinate, in order to have between 70-90% of the seeds germinated (with rootlets of more than 2 mm), in at least two days.


Root diversification

The root diversification stage includes first the transfer of germinated seeds to a container with special substrate intended for the diversification of seedling roots, in order to obtain plants with a high number of secondary roots that allow better grip and subsequent development in the areas of lagoon to which they are intended.

That special substrate will contain 25% black peat, 10% blonde peat, 25% fine sand, 20% thick sand and 20% natural sludge, with the particularity that the containers on which the substrate will be placed and seedlings will be of a volume ranging from 50 to 100 cm^3, with dimensions that will in no case exceed 4 cm in height.

These containers shall have perforated bottoms to ensure that water and nutrients are added to nourish the seedlings in order to stimulate development and especially root diversification.

Substances that will stimulate root development will be found in solution with other substances and will be offered deferred with a frequency of one dose each week of 4-8 times, so that the solution that stimulates root diversification will be composed of humic acid, Indole-3-butyric acid (IBA), ammonium, potassium and sodium.

For 4-8 weeks the seedlings activated in root development and diversification, so that after that time they will be assessed, having been estimated as optimal values to achieve root development which plants should have more than 8 secondary roots per plant and an average root weight of 6 g per plant. Where sampling of the seedling population allows at least 75% of plants to be available at the optimum level, transplantation shall be performed in containers for foliar development.

Foliar development

The stage of enhancing foliar development includes first the passage of seedlings to a new container with a volume ranging from 200 to 350 cm^3, and where the dimensions of the container must in no case exceed 12 cm in height (length), the bottom of the container perforated at the base must be up to 1/4 of the length of the container.

These seedlings shall be placed on a nutrient-rich substrate and under irrigation conditions with high concentrations of nutritious substances, in order to stimulate foliar development and facilitate the rapid production of a plant intended for commercialization.

This substrate on which the seedlings will be planted will have 40% black peat, 25% blonde peat, 20% sludge and 15% fine sand. In addition, irrigation with nutritious substances will be irrigated in an aqueous solution having urea, liquid fertilizer (nitrogen, potassium and phosphorus) and Indole-3-butyric acid (IBA).

The application of the nutritious substances will be organized following a protocol that depends on the prevailing environmental conditions, with applications being necessary two to three times in the months of maximum illumination (summer-spring) and once to twice in the months of medium or low lighting (winter-autumn).

Together with the application of nutritious substances, plants will have periodic preventive applications with phytosanitary substances, in order to eliminate potential pathogens, applying mainly pyrethrins and antifungals, with a monthly frequency.

Optimal foliar development conditions in plants shall be reached between 1 and 2 months and shall be estimated, following the established protocol. The optimal level is where the plants will be at least 50 cm tall and at least 80 g of foliar dry weight.

It is estimated as an optimal value for having populations of plants with adequate development, when at least 80% of the plants are above the optimal level established, so that when these levels are reached, commercial quality plants are available suitable for their exit from the industrial production process.

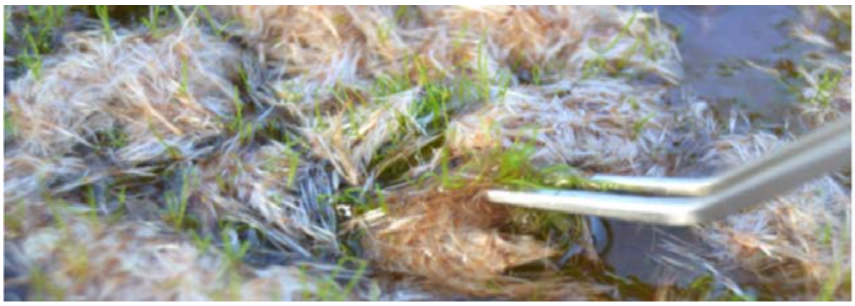
4 PRACTICAL AND AGRONOMIC GUIDE TO PLANT PRODUCTION

1. Locate the inflorescence of the Typha, in it are the fruits with the seeds.
2. Once the inflorescence begins to crumble, it is time to extract the seed contained in the fruit.
3. Detail of the fruit of the Typha. It consists of seed and a feathery structure called ginoforo, which allows its dissemination by the wind.

4. Once the seeds have been extracted together with the ginoforo or vilano, they are deposited in a clean tray for humidification. Water is deposited on the seeds until they are floated. With the tweezers the seeds are removed until they are soaked, this action is repeated for several days. The temperature is maintained between 20-28°C by using a heating bed in addition to covering the tray with clear plastic to increase moisture. Trays are placed in an illuminated place, without direct sunlight.

5 . After 4-6 days fine green strands begin to be appreciated, the seeds have begun to germinate. Keep the tray unchanged for another week until it is sized enough for transfer to substrate trays. Week after germination, the small groups of strands are carefully removed with long tweezers. Threads are deposited in trays with growth substrate. The trays must be perforated at their base, may or may not be divided into sockets. The substrate composed of peat, fine sand and natural mud must be porous, so that moisture is contained but not formed.

6. The trays must always be moistened, for this they are placed on a surface with several centimeters of water so that it ascends to the substrate by capillary thanks to the perforations.



7. Trays should be placed in a well-lit place, but without direct sunlight. The temperature is maintained between 20-28°C by using a heating bed in addition to covering the tray with clear plastic to increase moisture. These conditions are maintained by providing water when necessary until the seedling reaches about 5 cm in length, the approximate time is 20-30 days.

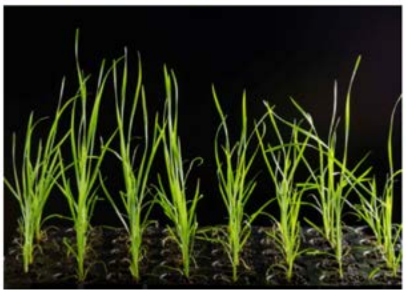
8. Once the plant reaches 5 cm, the whole plant is removed and with the hands are carefully separated, obtaining groups of 3-4 plants that are transferred to another tray where they are planted by inserting the roots into the substrate. The same conditions of humidity, lighting and temperature are maintained.

9. When the plants have acquired about 15 cm in height, after approximately 15 days, a new rinse and transplantation of the young women to the individual seedbeds is carried out and placed in the greenhouse.



In these conditions, after another 15 days, its development is completed until it has a population of 80% of plants 50 cm high, this being the optimal size for its exit from the greenhouse.






5 CLAIMS OF THE PLANT PRODUCTION PROCEDURE

The procedure of the invention is intended to obtain industrial quantities of typha spp. seedlings, used as raw material for the biodepuration of contaminated water, forming sieves or green filters of plants Aquatic.

The process includes the stages of seed conservation, seed germination for the industrial production of Typha plants, root diversification and an enhancer of foliar development in the childhood stages of seedlings, all according to a simple, economical process of making and with total efficiency in obtaining the plants.

A SUSTAINABLE APPROACH FOR DOMESTIC WASTEWATER TREATMENT IN RURAL AREAS USING NATURE-BASED SOLUTIONS

A.I. Ferraz[1] • J.M. Alonso[1] • F. Pereira[2] • P. Alves[2] • C. Calheiros[3]
A.C. Rodrigues[1]

Abstract

Based on the assessment and benchmarking of the wastewater services quality according to Portuguese standards and the identification of improvement opportunities, the present study aimed to develop a proposal for a sustainable solution for domestic wastewater treatment supported on natural systems. The main focus was to increase physical accessibility to the service and enable treated wastewater reuse in rural areas. For a small community in a Portuguese Municipality located in the Alto Minho region, with no available sanitation service, it is presented both a treatment system design, consisting on a bar screening, a septic tank, a subsurface horizontal flow constructed wetland and UV disinfection, and its localization definition supported by geographic information analysis. Regarding the relevance to develop similar approaches in the future, this study highlights the need and opportunity to develop spatially explicit models as decision support in the definition of sustainable solutions for the treatment of wastewater in low density areas.

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INTRODUCTION

According to the Portuguese legal framework the urban wastewater service is considered an essential public service (Law n.º 23/96, of 26 June), and recognized by the Water and Waste Services Regulatory Authority (ERSAR) as a structural service, fundamental to improve the quality of life, foster economic activities, and provide environmental protection (ERSAR, 2021). Despite the positive development trend, reflected by the increase of the service accessibility, quality and efficiency, currently monitored according to the 3rd generation of the assessment system defined by ERSAR, there is a clear need for evolution and improvement in order to ensure availability and sustainable management of water and sanitation for all as foreseen by the objective for sustainable development goal (SDG) nº. 6 of the UN 2030 Agenda "Clean water and sanitation". Wastewater retail utilities (WRU) performance assessment and benchmarking in 2019 points out 60% of good and average service quality evaluations, 33% of unsatisfactory evaluations and 7% with no reported evaluation. Considering the WRU from predominantly rural intervention areas in the NUTS North, the indicator that expresses the service coverage through sewage networks revealed an unsatisfactory evaluation in 15 of them (28.8% of the total), 4 of which located in the Alto Minho region. This indicator average value reached 85% for WRU in mainland Portugal, 66% for the ones located in predominantly rural intervention areas, and 58.7% in the Alto Minho region (ERSAR, 2021), situation that highlights the need to reinforce investment in sanitation infrastructures.

The implementation of conventional wastewater (WW) collection, drainage and treatment systems in low density areas, with complex orography pose limitations regarding the installation of gravity sewer networks and landscape planning challenges. Therefore, the present work aims to develop a proposal for a sustainable WW treatment solution, based on natural systems (constructed wetlands, CW), in alignment with technical, environmental and socio-economic conditions of predominantly rural areas, in order to increase physical accessibility to the service and treated wastewater (TWW) reuse. Constructed wetlands are recognized as nature-based solutions that provides multiple ecosystem services (Calheiros et al, 2020, 2018), with low costs, and suitable as WW treatment plants sized at less than 2000 population equivalents (p.e.) (Monte et al, 2018) located in rural areas (Rodrigues et al, 2020). The approach to develop the solution here presented intends to meet sustainability criteria through i) its capacity to effectively treat urban WW, ensuring the contamination reduction of receiving water bodies; ii) the production of TWW with quality grade for reuse, reducing anthropogenic stress on water resources and contributing to water-use efficiency and circularity within the urban-water cycle; iii) low operation and maintenance interventions needs; iv) low energy consumption and v) the reduced landscape, air quality and noise impact.

METHODOLOGY

The proposal development adopted the following methodological steps: i) data collection for the calculation of 2019 wastewater quality service indicators for a WRU from the Alto Minho region, and the identification of possible improvement opportunities to implement; ii) analysis of the sectoral legislation, strategies and policies and definition of the Strategic Reference Framework (SRF), supporting the measures and actions suggested to improve the quality of the service provided to users and the validation of the proposed WW treatment solution; and iii) analysis of the municipality sanitation network project, selection of the area to be considered for the proposal development focused on a sustainable WW treatment solution based on the SRF, aligned with the service quality assessment indicators and adequate to areas with a predominantly rural profile sustained on criteria for their location and design supported by Geographic Information Systems (GIS).

RESULTS AND DISCUSSION

The analysis of data reported to ERSAR by the WRU and the calculation of the most relevant service quality indicators for the present study, namely AR01 – Service coverage through sewage networks, AR06 - Connection to the service, AR11 - Accessibility to wastewater treatment, and AR13 - Compliance with the discharge permit, reveals an unsatisfactory quality for indicators AR01 and AR06, good for AR11, and not applicable for the AR13 (no treatment plants with valid discharge permit). The characterization of the WRU profile highlights the lack of TWW reuse. The improvement opportunities identified, regarding both a better wastewater service quality assessment (SQA) and the SRF, are presented in Table 1.

Table 1. *Improvement opportunities and proposed implementing measures: relation with Strategic Reference Framework (SRF)*

Improvement opportunity	Implementing measure	SDG	SQA	SRF
Increase service coverage through sewage networks	i) Extend the sanitation network to areas with no available WW service; ii) WWTP construction: WW treatment and TWW reuse		AR01 dAR12b dAR20b	PENSAARP 2030 Law nº 58/2005, 29 December D.L. nº 236/98, 1 August D.L. nº 157/97, 16 June D.L. nº119/2019, 21 August
Increase the compliance with discharge permit	i) Provide the construction of a WWTP with monitoring and control systems		AR13 dAR46b dAR47b dAR48b dAR49b	
Promote TWW reuse	i) WWTP designed to produce TWW with quality compatible with WW reuse; adequate treatment level for irrigation water: disinfection		dAR56b	

Note: SDG: sustainable development goal; SQA: service quality assessment; WWTP: wastewater treatment plant; TWW: treated wastewater; WW: wastewater

Analysing the municipality sanitation network project, a group of households with no connection to the WW service, located at a low altitude area requiring pumping stations to elevate WW to the main collector connected to a centralized wastewater treatment plant (WWTP), was identified (Figure 1a). From this diagnosis and the technical, environmental and socio-economic conditions to be considered within this project context, the solution to be developed will focus on the installation of a WWTP consisting of a screening system, a bi-compartmented septic tank (ST), a biological treatment (2 horizontal subsurface flow CW operated in parallel), and a disinfection system by UV radiation in a closed channel (no contact, automatic cleaning system, with photovoltaic panels to electricity supply), to ensure compliance with TWW quality criteria for reuse. The selection of the study area (Figure 1b), which includes 16 households and 1 warehouse, resulted from the GIS analysis, identifying locations meting the following criteria: i) no WW sanitation service available; ii) land use compatible with the installation of a WWTP and easy access; iii) land availability; iv) land orography enabling the installation of a gravity collector network; v) proximity to a water body for TWW discharge (when reuse is not possible); vi) proximity to users with the potential to reuse TWW (e.g. dairy farm < 500 m away, greenhouses for horticulture < 250 m away, surrounding fields).

Figure 1. *a) Representation of the sanitation network project and areas with no physical accessibility to the service; b) area considered for the project development (households and WWTP location).*

Table 2 presents the data considered for the project design and Table 3 the results obtained for the septic tank and the CW design.

Table 2. *Data for project design*

Project parameter	Reference	
Population, *Pop*	Censos 2011 (INE), considered the municipality ratio for residence population/household	37 p.e.
Capitation, *Cap*	DR n°23/95, 23 August	80 L hab^{-1} d^{-1}
Inflow Coefficient, $Coef_{in}$, (rural area)	DR n°23/95, 23 August	0.65
Per capita loading		60 g O_2 hab^{-1} d^{-1}
Per capita loading for primary effluent		40 g O_2 hab^{-1} d^{-1}
Average flowrate, Q_{av}	$Q_{av} = Pop \times Cap \times Coef_{in} + Q_{inf} + Q_{industrial}$ (Eq. 1)	2.12 m^3 d^{-1}
Infiltration flowrate, Q_{inf}	10% of Q_{av}, (assuming new and technically efficient network; DR n° 23/95, 23 August)	0.19 m^3 d^{-1}
Peak flow, *PF*	Health Education Services Division (2016)	3
Peak flowrate, Q_{peak}	$Q_{peak} = Q_{av} \times PF$ (Eq. 2)	6.35 m^3 d^{-1}

Table 3. *Septic tank (ST) and constructed wetland (CW)design results*

Considerations for system design	Data for ST and CW design	Reference
Septic tank with 2 compartments $V_{1st\ compartment}$ = 2/3 total volume total HRT = > 12 h Sludge accumulation = 80 L person^{-1} year^{-1} Desludging interval: 1/year Depth = 1.5 m + 0.3 m free board; Length = 1.5 m	V_{max} = 6.75 m^3 $V_{1st\ compartment}$ = 4.5 m^3 (2x1.5x1.5) $V_{2nd\ compartment}$ = 2.25 m^3 (1x1.5x1.5) V_{total} = 8.10 m^3 $V_{sludge\ accumulation}$ = 2,96 m^3 year^{-1} $HRT_{minimum}$ = (6.75-2.96)/Q_{peak} = 14.3 h	UN-HABITAT, 2008
Subsurface flow CW 2 parallel basins Organic superficial load 25 g m^{-2} d^{-1} Substrate: medium sand, = 0.35 wastewater level, *h* = 0.5 m HRT: 3 - 15 d Hydraulic load: 2 - 20 cm d^{-1}	Organic load = 1.48 kg O_2 d^{-1} Influent BOD_5 from Septic Tank effluent = 0.5 g O_2 L^{-1} Minimal area considering Q_{peak} = 127 m^2 HRT = = 3.5 d (Eq. 3) Hydraulic load = Q_{peak}/A = 5 cm d^{-1} 2 basins: 5 m x 13 m (racio Length:Width = 2.6: Area = 65 m^2 x 2 A_{total} = 130 m^2	Monte *et al*, 2018

CONCLUSIONS

From this project development several opportunities to improve wastewater service quality provided by retail utilities where highlighted, resulting in the proposal for a sustainable solution that suits to predominantly rural areas characteristics and areas with constraints concerning the connection to sanitation networks. Considering the challenges associated to the optimization of natural-based wastewater treatment systems location, this study points out the need to carry out further studies aiming to develop spatially explicit analysis models to decision support regarding the definition of sustainable solutions for WW treatment facilities in low density areas.

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WETLANDS (NATURAL, RESTORED AND ARTIFICIAL) AS STRATEGIES FOR ENVIRONMENTAL EDUCATION AGAINST CLIMATE CHANGE WITHSTUDENTS OF PROFESSIONALIZINGHIGH SCHOOLSIN THE SOUTHWEST OF CASTILLA Y LEÓN (SPAIN)

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Abstract

Different types of wetlands are an essential part of the natural environment. Despite being largely unknown, their ecological importance is due, among others, to the fact that they absorb pollutants, improve water quality, protect coasts from the impact of waves, reduce flooding and are reservoirs of CO2 in the fight against Climate Change. Due to the pandemic, the students educational tasks and actions included in this work were not face-to-face but virtual and were disseminated using on-line methods, through MOOC courses that contain, among other resources, short videos as educational pills that will be shown in different collaborating Spanish secondary education institutes, as well as synchronous seminars introducing the analytical control and monitoring of the different types of wetlands as regulated laboratory practices of the secondary institute, etc. With this work, it is expected to achieve a better knowledge and a greater environmental awareness of the students about wetlands and their importance for biodiversity and the environment, stimulating them to promote conservative and sustainable actions that have a positive impact on wetlands among their relatives and acquaintances. In this sense, secondary school students from IES Fray Luis de León and Martínez Uribarri in the city of Salamanca (Spain) have received environmental education and knowledge related to the characteristics, functions, quality and importance of these water ecosystems and their role regarding Climate Change in the Southwest of Castilla y León. It is hoped that through these MOOC courses, which will be incorporated into Miriada X, as well as through their appropriate dissemination by governmental and educational institutions, a better knowledge of wetlands at a global level will be facilitated.

INTRODUCTION

Wetlands are ecosystems represented by permanent or seasonal water bodies with a depth of no more than 6 metres and can be fresh, brackish or salt water in a lotic or lentic system (Ramsar, 2013). Their ecological importance is due to the fact that they absorb pollutants, improve water quality, protect coasts from the impact of waves and reduce flooding, as well as providing a suitable habitat for animals and plants, giving rise to a great biodiversity. Three main classes of wetlands are known: marine and coastal wetlands, inland wetlands and artificial wetlands. In 2020, 353 wetlands with more than 50000 wintering waterbirds were identified in Castilla y León (Spain) (Sánchez, 2020). Currently, there are 2041 Natura 2000 Network protected areas in Spain, 1445 Sites of Community Importance (SCIs) and 596 Special Protection Areas for Birds (SPAs), which means that 30% of the country's surface area is within a protected area. Only 10% of Natura 2000 Network sites have a specific management plan, despite the fact that all sites should have a management plan approved before 2011, according to the Spanish Natural Heritage and Biodiversity law(RD 42/2007). Despite the importance of this network, 85% of Spaniards surveyed said they knew nothing about these sites. Of the remaining 15%, only 8% acknowledged that they knew "something" about them. Furthermore, of this 15%, 10.5% are unable to name a site belonging to the Natura 2000 Network (www.natura2000day.eu). In addition to these recognised and protected areas, there are many ecologically important wetlands whose water quality is not controlled, posing a danger to both local and migratory flora and fauna. Due to a lack of knowledge, the local population misuses them, ignoring their socio-economic, cultural and environmental potentials. Thus, in the area of Southwest Castilla y León, specifically in the provinces of Zamora. Ávila and Salamanca (area of influence of the University of Salamanca), there are 53 catalogued wetlands, many of which are unknown to the majority of the local population.(http://www.jcyl.es/web/jcyl/MedioAmbiente/es/Plantilla100/1284151755866/_/_/_).

The Paris Agreement on Climate Change recognises that wetlands play a key role in regulating the amount of carbon present in the atmosphere, as these ecosystems act as efficient sinks mainly through peatlands (Ramsar, 2019). It is estimated that peatlands sequester one third of terrestrial carbon, even though they represent only 3% of the earth's surface (Aquae, 2021), making wetlands a valuable natural or artificial space for addressing the effects of Climate Change.

Education is a timely but underutilised tool to address Climate Change. This toolallows for knowledge acquisition related to climate change, environmental and social issues, disaster risk reduction and sustainable consumption. In order to generate significant changes towards more sustainable habits and actions, environmental education must focus on an institutional environment in which such content can be taught to generate environmental awareness, and thus ensure that schools and education systems themselves are training centres for climate change adaptation in new generations (Anderson, 2021).

This project comprises an educational package based on the knowledge of wetlands in Castilla y León and has been designed to promote environmental awareness among secondary school students in Salamanca and to strengthen their knowledge of water resources and their role in the face of Climate Change. This project develops a website as a virtual pedagogical platform containing audio-visual material with information on wetlands and waters preservation in the framework of Climate Change problem.

METHODOLOGY

This project is a pilot experience involving professional training centres together with Water Interpretation Centres and the Salamanca University, through its Centre for Research and Technological Development of Water (CIDTA), with the aim of stimulating groups of professional training students who have traditionally been distanced from these concerns, in order to initiate them in researching wetlands. Their initiation to research is developed through activities, workshops, theoretical and practical sessions, adapting the contents and the scientific method to the measurement and data interpretation on water bodies, especially temporary rivers and natural and artificial wetlands,making them aware of the importance of controlling these water resources, so affected in the Mediterranean basin by Climate Change and on compliance with the requirements of the European Union established in its Water Framework Directive in relation to their chemical and ecological status. Due to the current pandemic prevention measures, Information and Communication Technologies (ICT) have been used in this work and through online platforms, short educational videos, no longer than 10 minutes, have beenshooted and edited as well as live connections, through its Youtube channel, to the Oso steppe lagoons (Ávila). The script, recording and editing of these videos have been carried out by CIDTA staff, teachers from Fray Luis de León and Martinez Uribarri IES, environmental educators and experts from the collaborating interpretation centres, Tormes E-BFundation, Interpretation Center of Tormes River Mouth (Monleras City Council, Salamanca) and the Interpretation Center of the steppe lagoons of El Oso (Ávila) and may serve as a model for similar actions of environmental dissemination to other schools and educational centers. The dissemination project runs from September 2020 to September 2021 and is funded by the Spanish Foundation for Science and Technology (FECYT) of the Ministry of Science and Innovation (project: "*FCT-19-15200 CON-CIENCIA de ríos y zonas húmedas del Sudoeste de Castilla y León"*) and by the Scientific Culture and Innovation Unit (UCCI) of the Salamanca University (III Call Grants forScientific, technological and innovative disseminations).

Three wetlands were chosen to learn about the different types of wetlands: 1) the Natural Wetland of the Steppe Lagoons of El Oso, whose interpretation centre and museum, located in the municipality of El Oso, are dedicated to the dissemination of the richness of the wetlands of the La Moraňaregion (Ávila), which receives and shelters more than 50000 birds of different species throughout the year; 2) as a model of Restored Wetland, the gravel pits of Tormes-EB Fundationin Almenara de Tormes (Salamanca) were chosen, a wetland that also houses a study centre and museum of this SCI area, not only for the development of dissemination activities but also for different research activities which are transferred to other areas of Spain involved in the restoration of this type of wetlands (Andalusia, Murcia, Castilla y León, etc) and 3) as an artificial wetland, the low-cost wastewater treatment plant based on natural passive methods in the municipality of Monleras (Salamanca) was chosen, which consists of a primary treatment carried out in an Imhoff tank, a secondary treatment that is initially carried out in a horizontal wetland of macrophytes in flotation with other aquatic plant species, after which the water is sent to a secondary treatment plant in a horizontal wetland of macrophytes in flotation with other aquatic plant species, after which the water is sent to three parallel vertical wetlands with subsurface flow and finally reaching an artificial wetland that functions as an aerobic lagoon, due to its shallow depth, for the disinfection of the water by the effect of sunlight (Alainez, 2014). This plant was designed by the authors of CIDTA and this design was chosen by the Duero Hydrographic Confederation (Spain) to finance its construction in the framework of its call for pilot projects "Singular experimental treatment of wastewater discharges in small towns in the Duero river basin" (García Prieto et al. 2008).

Students from collaborating secondary schools also participate in the development of dissemination activities, water quality control, interviews with waterexperts, etc. These videos on different topics are uploaded as educational resources to the virtual platform as a pilot virtual course to be subsequently disseminated in the form of MOOC-type educational pills to other students from other schools and educational centers. Furthermore, these videos and other educational resources will be uploaded to the Con-Ciencia-Tewebsite and, as they are freely accessible, they can be viewed by all those interested, providing knowledge of tools and instruments to support and manage the main current sources of scientific documentation and dissemination on the world of water and developing the necessary skills to find new ones.

RESULTS

The content of the videos is focused on the different types of wetlands and the Climate Change's impact on wetland conservation. Amongthemaincontentsto be covered are:

- The ecological, tourism and economic importance of rivers, wetlands, reservoirs and fresh water bodies.
- The fauna and flora that inhabit these ecosystems, invasive species, pollution, etc., with an emphasis on the problems arising from Climate Change.
- Procedure for sampling and analysis of water quality and interpretation of results.
- The importance of tangible and intangible traditional cultural heritage associated with the uses of water as didactic objectives of different educational disciplines.

The aim of this work is to raise the environmental awareness of students and the general public about wetlands and their importance for biodiversity and the Environment, in order to promote conservative and sustainable actions and habits in relation to wetlands (Figure 1). In this way, the expected results will be a remarkable degree of interactive learning and self-learning, using both the virtual tools described above, the contents of the educational e-bookandthe information search strategies on the Internet:

a) By Reflection: preparation of summaries of the laboratory practices, scientific articles.

b) By Exploration: tutor-guided visits on Internet, videos.

c) By Incidental or Interested Exploration: thanks to databases or virtual libraries, there are many possibilities to search for documentation in different servers and formats.

d) Become familiar with Internet: Some course activities have been designed so that those who follow them are able to increase and update their skills in the beneficial use of Internet.

FIGURE 1. *INFOGRAPHIC POSTER OF THE PROJECT.*

In addition, it is expected to create a virtual learning environment by:

a) Forums: Virtual channel that allows the connection between the components of the e-book. In turn, working groups will be opened with student groups and their teachers for the orientation of common problems.

b) E-book board: Permanent message board page of the e-book to which any of the e-book participants can send messages.

c) Real and daily monitoring of all the student's teaching activity, which is automatically recorded on the teacher's computer (server) through Studium (https://studium.usal.es/), the Virtual Campus of the Salamanca University, which allows continuous assessment and early detection of learning problems that can be quickly solved with the teacher's individualised response, using e-mail or other telematic means such as synchronous or asynchronous virtual tutoring sessions. In the case of general interest questions, the electronic bulletin board would also be used.

d) Potential accessibility of these educational media and tools to a large number of students from different regulated educational specialities at the same time (Environmental Health, Chemistry and Process Control, Safety and Environment, etc.).

CONCLUSIONS

Environmental education allows for the creation of spaces for environmental awareness among secondary school students, using as thematic axes the preservation of water resources in particular wetlands, and the importance of appropriate actions to face the Climate Change effects. On-line didactic tools such as thematic videos, explanatory talks, and the implementation of web pages allow for the appropriate dissemination of these topics due to the advantages that these telematic pedagogical strategies and tools possess. Advantages include their permanent availability on web platforms, their easy dissemination and more agile forms of educator-student interaction and knowledge transmission.

Secondary school students from Fray Luis de León and Martínez Uribarri IES in Salamanca receive education and knowledge related to the characteristics, functions, quality and importance of water ecosystems and their role in the face of Climate Change in the Southwest of Castilla y León. In addition, they will assess the impacts generated by Climate Change on fauna and flora species, as well as on natural resources, adequately approaching students closer to this current environmental issue, both regionally and globally. It is necessary to continue with the formulation and implementation of this type of environmental and educational initiatives in order to attract the interest of young people in the conservation of water resourcesand the environment and the understanding of the phenomena already visible as a result of Climate Change.

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APPROACH TO MODELS OF BACTERIAL DISINFECTION BY PHOTOLYSIS OF TREATED WASTEWATER IN ARTIFICIAL WETLANDS

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Abstract

The simulation of the inactivation processes of coliform bacteria and Clostridium sulfite-reducing bacteria (vegetative species and spores) taking place in the water of a maturation pond of a low-cost wastewater treatment plant using artificial wetlands was studied by means of photolysis in a pilot photoreactor.A discrimination has been made between the different mechanisms of inactivation by photolysis of these bacteria according to the criteria of different statistical and kinetic models. The results obtained show an approximation to the models of bacterial disinfection in maturation ponds. Disinfection by photolysis is not total but a good bacterial inactivation performance is achieved, with all that this implies considering the risks that non-disinfection represents for aquatic ecosystems.

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INTRODUCTION

The inactivation of microorganisms by ultraviolet radiation has been known for more than 100 years (Downes and Blunt, 1877). The germicidal action of UV-C light (photolysis) depends on the absorption by the micro-organisms of the appropriate electromagnetic energy (wavelength and irradiation power), which in turn depends on the properties of the fluid itself and the substances present in it, such as suspended solids, which can absorb part of this electromagnetic radiation (Lorch, 1987). This electromagnetic energy affects the genetic material of the organism (DNA or RNA), so microorganisms cannot replicate and therefore die (Bolton and Linden, 2003). An important aspect in the effectiveness of this process is the existence of nucleic acid repair mechanisms called photoreactivation or photorepair, in which a photoreactive enzyme, after absorbing radiation, is able to repair the damage caused (Groocock, 1984). This regenerative capacity occurs in bacteria and other microorganisms, but not in viruses, and its performance is related to the extent of UV-C damage, exposure to the reactivating light, pH and water temperature (Masschelein and Rice, 2002). The photoreactivation phenomenon will require that the exposure of the microorganism to the reactivating light does not exceed two to three hours after inactivation, taking into account that the degree of reactivation is a function inversely proportional to the radiation dose used.

Nowadays, Nature-based wastewater treatments are an important alternative in the field of wastewater disinfection strategies to reduce pathogenic species affecting health (Vymazal, 2005; Wu et al. 2016) and their reuse for agricultural irrigation (Masi and Martinuzzi 2007; Nan et al. 2020). In addition, disinfection is required in some areas to comply with certain directives, such as the Habitats Directive (92/43/EEC) and the Bathing Water Directive (2006/7/EC). Thelevelofdisinfectionisinsufficient in constructedwetlands (López et al. 2019), thussomeauthorsproposethe use of UV photolyticreactors (Azaizeh et al. 2013; González et al. 2019) and othersthe use ofmaturationlagoons (Tanner et al. 2005; Russo et al. 2020).Bacterialcellshavealwaysbeendescribed as targets forthestudyofwaterdisinfection, withcoliform and clostridialmicroorganismsselected as processcontrols, beingthemaingroupsof bacteria coveredbydrinkingwaterregulations(Gorchevy Ozolins 1984).

Disinfection in constructed wetlands depends on different factors, such as water composition, seasonal fluctuations, local vegetation and whether or not excreta from wild birds and other animals adds significantly to the total human excreta (Rahman et al. 2020). This means that low-cost Nature-based wastewater treatment plants cannot offer a standardised performance, unlike conventional treatment plants. Pathogen removal mechanisms are also complex, as they often include natural death by starvation or predation, sedimentation and filtration, and adsorption (Al-ufasi et al. 2017). Therefore, it is difficult to assess the performance of wetlands. In this work, we studied the feasibility and performance of UV photolysis in maturation ponds coupled in series with artificial wetlands for the inactivation of coliform and clostridial bacteria and their spores. For this purpose, a photolytic pilot reactor simulating the photolysis process in such lagoons was used.

METHODOLOGY

The photoreactor used consists of a tank for the real water sampleswith a capacity of 200 L, a 1 hp pump for the recirculation of the sample in the photoreactor system, in whose outletthe water passes through a 50 m solid filter and then through a rotameter to measure the flow rate. The UV lamp is installed in a transparent quartz tube inside a stainless steel tubular reactor. The experiments were carried out with total recirculation at a Q= 1000 L/h and a temperature of 20°C.

For the study of the inactivation of Coliform and *Clostridium* bacteria, aliquot samples were taken from the effluents of the Wastewater Treatment Plant (WWTP) of the municipality of Monleras, in the province of Salamanca, Spain. The primary treatment is carried out in an Imhoff tank for the decantation of solids and digestion of organic matter. Secondary treatment is initially carried out in a horizontal wetland of floating macrophytes together with other aquatic plant species, then the water is sent to three parallel vertical wetlands of subsurface flow, where the water percolates vertically through an inert substrate of sand and gravel; and finally the water reaches an aerobic maturation lagoon, due to its shallow depth, for the disinfection of the water by the effect of sunlight. This WWTP was designed by the authors of CIDTA and this design was chosen by the ConfederaciónHidrográfica del Duero (Spain) to finance its construction in the framework of its pilot project “Unique experimental treatment of wastewater discharges in small towns in the Duero river basin” (García Prieto et al. 2008).

The coliform bacteria count was carried out by APHA method 1995 9215 B (Standard Methods, 2002). The culture and count of vegetative cells and spores of *Clostridium perfringens* was carried out following the procedure described in the Spanish Standard UNE-EN 26461.

RESULTS

The susceptibility of microorganisms to photolytic processes is affected by many factors. Previous studies have shown that it is very difficult to extrapolate the results obtained using synthetic waters since,atthe bacterial growth stage, there is a synergy between the different microbial species present in the effluent. Therefore, an annual monitoring study was carried out on the main physicochemical characteristics of the wastewater from the Monleras WWTP and the global UV radiation (direct and diffuse) corresponding to the solar irradiation that reached the maturation lagoons (Figure 1). For the simulation of these processes, a UV lamp was installed in the photolytic reactor to simulate the conditions described (Figure 2).

FIGURE 1. ***MAIN PHYSICOCHEMICAL CHARACTERISTICS OF WASTEWATER SAMPLES FROM THE MONLERAS WWTP (SALAMANCA, SPAIN) AND THE OVERALL UV RADIATION ON THESE WATER SAMPLES THROUGHOUT THE YEAR.***

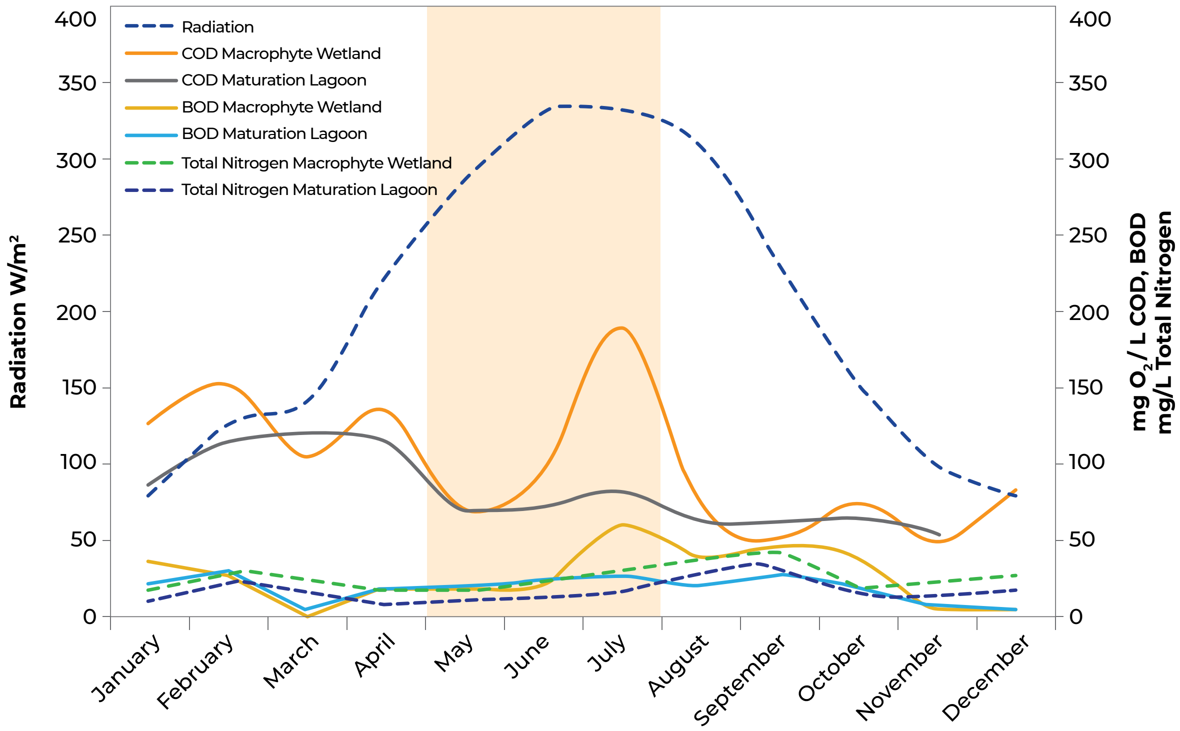
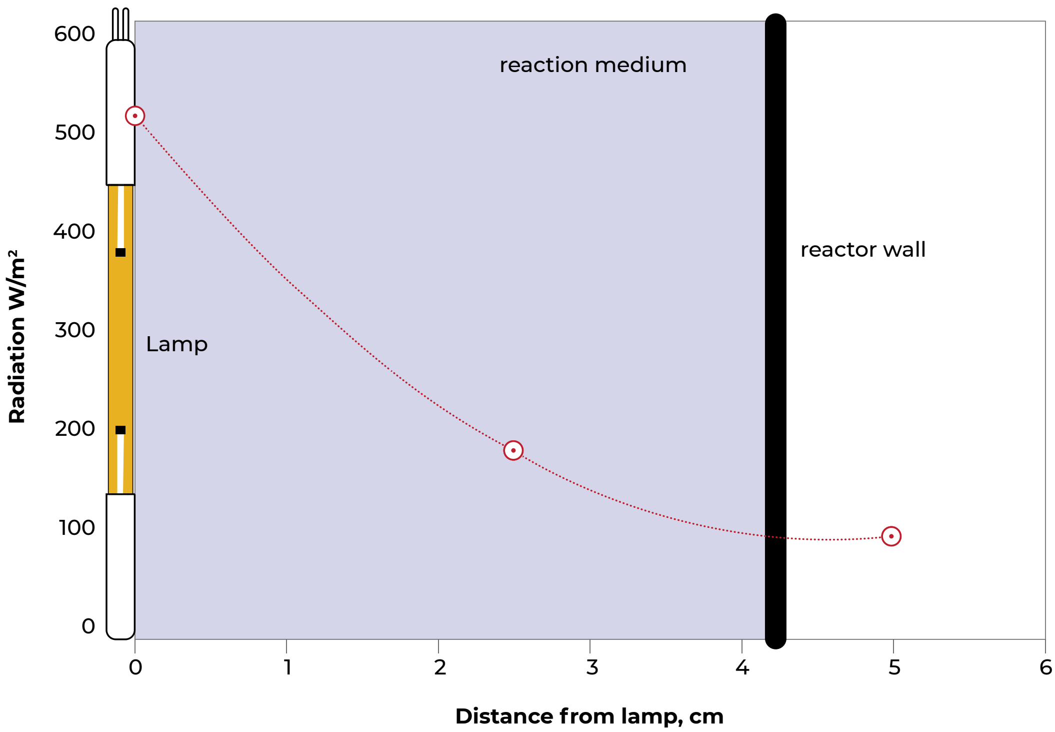


FIGURE 2. ***ELECTROMAGNETICI RADIATION POWER CHARACTERISTICS OF THE LAMP INSTALLED IN THE REACTOR.***



Under these conditions, the wastewater samples were taken in the months of May to July, when the climatic conditions are optimal for the reproduction of these microorganisms and the general conditions of UV radiation are the most similar to the radiation emitted by the lamp of the photolytic reactor. The bacterial inactivation processes in the maturation pond are simulated in the photolytic reactor, using effluents from the macrophyte wetland as samples. Two types of models, one statistical and the other kinetic, were used to study the disinfection processes in the Monleras WWTP maturation lagoon and a preliminary study was carried out to discriminate between the different models representing the different possible microbial survival curves under UV radiation (Gyürek and Finch, 1998).

Weibull-type statistical models propose that cell survival patterns can change with the physiological state of the cells and their adaptation to stress (Coroller et al. 2006). It is assumed an initially large subpopulation that is more sensitive to stress (first part of the inactivation curve) and a smaller subpopulation that is more resistant to stress (second part of the curve).

Its integrated rate equation is:

$$Log\, N_t = \log(\frac{N_o}{1+10^{\propto}}\left[10^{-\left(\frac{t}{\delta_1}\right)^P+\propto}+10^{-\left(\frac{t}{\delta_2}\right)^P}\right])$$

Where Nt represents the bacterial concentration at time t, N_o is the initial concentration of microorganisms (CFU/mL), d_1 andd_2 parameters correspond to the time required to reduce the first logarithmic decimal cycle of the sensitive and resistant bacterial population, p indicates the shape of the equation curve and α is the fraction of the first remaining subpopulation of the total population, defined as α = log (f/(1-f)), where f is the fraction of the initial population after the rapid inactivation reaction and (1- f) is the fraction of the initial population following the second stage of the inactivation reaction.Among the kinetic models considered that allow describing non-linear survival curves, two different models have been used:

The inactivation processes of *Clostridium* sulphite-reducing bacteria are adapted to a two-phase kinetic model that considers two groups in the microbial population, one with initial resistance to stress, with an initial protection that is gradually destroyed, and a second more resistant population group based on vitalistic or mechanistic models (Geeraerd et al. 2005). This model is defined by the equation:

$$Log\,\frac{N_t}{N_o} = \log(\left(f\,.\,e^{-k_1 t}+(1-f)\,.\,e^{-k_2 t}\right).\frac{e^{-k_1 S}}{1+(e^{-k_1 S}-1)\,.\,e^{-k_1 t}})$$

where k_1 is the rate constant of the sensitive population and k_2 is the rate constant of the resistant population. The parameter S is the initial stress resistance before bacterial decay.

The inactivation of coliform bacteria does not fit a two-phase model but fits a logarithmic-linear model with a shoulder (Geeraerd et al. 2000). Curves typically showing an initial shoulder-type deviation indicate that a fraction of surviving micro-organisms remains constant in the first instants of treatment, followed by a linear decrease in the number of surviving microorganisms. This behaviour is attributed to an inadequate distribution of the UV light in the sample, a delay in the diffusion of the UV light to the bacterial sites of action or an initial resistance of the microorganisms to the attack of the disinfecting agent. This model is defined by the equation:

$$N_t = (N_o - N_{res})\left(\left(e^{-k_{max}t}\right) . \left(\frac{e^{k_{max}S}}{1 + (e^{k_{max}S} - 1) . e^{-k_{max}t}}\right) + N_{res}$$

Where k_{max} is the specific inactivation rate constant, N_{res} is the residual population density and S is the initial stress resistance.

The results obtained for the inactivation study of *Clostridium* sulphite-reducing bacteria are shown on Table 1:

TABLE 1. ***KINETIC AND STATISTICAL PARAMETERS OF THE MODELS FITTED TO THE EXPERIMENTAL KINETIC DATA FOR THE INACTIVATION OF CLOSTRIDIUM BY PHOTOLYSIS.***

BifaseModel	k_1 (min^{-1})	k_2(min^{-1})	*f*	S
	0.24 ± 0.07	0.11 ± 0.05	0.94 ± 0.11	5.3 ± 2.1
Double Weibull Model	*p*	δ_1 (min)	δ_2 (min)	α
	1.4 ± 0.3	14.5 ± 1.2	31.2 ± 5.7	1.3 ± 0.4

Bacterial regrowth of encysted species was observed over time. To verify this, the samples submitted to photolytic treatment were taken after a time of more than 4 logarithmic cycles (4d) which is the classical value considered as a guarantee of hygienic-sanitary quality.

The results obtained for the inactivation study of total coliform bacteria are shown on Table 2:

TABLE 2. ***KINETIC AND STATISTICAL PARAMETERS OF THE MODELS FITTED TO THE EXPERIMENTAL KINETIC DATA FOR THE INACTIVATION OF COLIFORMS BY PHOTOLYSIS.***

Log-linear model with shoulder		*kmax*(min^{-1})	S	4d (min)
		0.63 ± 0.03	2.62 ± 0.75	± 17.4
Double Weibull Model	*p*	d_1 (min)	d_2 (min)	α
	1.7 ± 0.2	4.2 ± 0.9	8.3 ± 1.1	0.43 ± 0.36

It was observed in the monitoring of the maturation lagoon that the total elimination of coliforms is not reached, suggesting processes of photorepair or bacterial photoreactivation.

CONCLUSIONS

Anaerobic sulphite-reducing *Clostridium* bacteria have been used as disinfection indicators because of their high resistance due to the formation of resistant spores. Their inactivation by photolysis fits models in which two types of bacterial populations coexist, one sensitive (vegetative species) and the other (spores) resistant to treatment, the inactivation kinetic constants for the sensitive one being (94 %) k = 0.24± 0.07 min-1 and for the resistant,(6 %) k = 0.11± 0.05 min^{-1}. The differences found can be explained in morphological terms: the spore differs significantly from the vegetative cell, because the cell wall is thicker in spores than in vegetative cells, the disinfection process according to this mechanism should be different for vegetative species (sensitive population) and spores (resistant population). Total *Clostridium* inactivation is not achieved and reactivation of sporulated species occurs, which implies a risk of transmission from one wetland to another.

In the inactivation of coliform bacteria, the shoulder phase shown by a coliform bacteria concentration (N_o) with a very gentle decay, is attributed to the loss of cell viability following the accumulation of damage during the photolytic process. The first part of the curve shows the resistance of the population to an electromagnetic radiation attack during the shoulder period. 2.62 ± 0.75 minutes. The statistical fit confirms that the curve is convex. (p = 1.68 ± 0.20) with two populations of coliforms with different physiological status and with times required to reduce the first logarithmic decimal cycle of one type of bacteria to different physiological states of$d_{1\,=}$ 4.15 ± 0.88 y d_2 = 8.29 ± 1.13 minutes respectively. According to the suggested model, total inactivation is reached at 17.4 minutes. It was found during the study time that the total elimination of coliforms in the maturation lagoon is not reached, which indicates that there is photoreactivation by sunlight. Therefore, when such treatments for the disinfection of treated wastewater by low-cost natural processes are implemented, it is the cumulative dose of ultraviolet radiation and not the treatment time, which should be considered more important at the time of implementation. In this sense, with proper maintenance of the plant, low vegetation and water level in the maturation lagoon, and a good pretreatment, good disinfection performance values can be obtained, even in the less favourable months, due to the fact that in these months the treatment time tends to increase, due to a decrease in the wastewater flow to be treated in winter, since during these periods a higher dose of radiation is necessary to achieve bacterial inactivation.

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MODELING OF WWTP SLUDGE DEHYDRATION IN DRYING BEDS

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Abstract

The by-product of wastewater treatment is called sewage sludge and must also be treated for final disposal. Its treatment encompasses three main objectives: Stabilization of organic matter, reduction of volume, and reduction of pathogens load. Sludge dehydration is an essential step in the process since the reduction in volume translates into greater ease and lower costs of transportation and landfill. Some fast and compact methods used in dewatering sludge have high energy consumption and are expensive. Contrary to that are the drying beds, which due to the simplicity of construction and operation and the low cost, are a priority alternative. The objective of this work was the formulation of mathematical models that describe the dewatering of WWTP sludge in drying beds, depending on the climatic conditions. The results of the models were satisfactory and a significant reduction in volume was observed, even in cold and humid weather.

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INTRODUCTION

Many conferences related to the theme of wastewater treatment have been carried out in the last decades (UN, 2017), highlighting a problem related to the subject and the need for debate and research.

Effluent treatment generates a by-product, the called sewage sludge, which must undergo appropriate treatment before final disposal.Sludge is considered solid waste from wastewater treatment, however, it has approximately 95% water (Sperling and Franci, 2001). A large volume is generated continuously, so its dehydration is an essential step in the process, and the reduction in volume translates into greater ease of operation and a lower cost, both for transportation and landfill.

There are several techniques for sludge dehydration, with the selection of the most favorable method depending on several factors: area available for its implementation, the volume of sludge generated, availability of technology, and climatic factors. Mechanical methods are faster and more compact but are associated with high energy costs, in addition to the complexity of installation and operation. In the drying beds (DB), on the other hand, the sludge is exposed to the environment, resulting in the evaporation and leaching of water by natural effects. In this way, they are exempt from spending on electricity and are easy to install and operate, requiring only time and space superior to energy methods (Sperling, 2007).

The choice of drying beds takes into account the volume of sludge generated, the available space, and the weather patterns of the region. Thus, the existence of mathematical models that relate these data is essential, allowing the evaluation of the technical feasibility of the process for each situation. This study intends to develop a mathematical model that describes the dehydration of sludge from Wastewater Treatment Plants - WWTP in drying beds, relating meteorological data in the water mass balance.

MATERIALSAND METHODS

The experimental study consisted of the design, construction and application of drying beds for sludge dewatering at the WWTP in Bragança, Portugal. The sludge treatment station using drying beds was set up at the Campus Santa Apolónia, Polytechnic Institute of Bragança - IPB. Built using four tanks (TK), with a square section (1m x 1m) and a useful volume of 1000 L, perfectly watertight, with a sampling port on the base for collecting leachate. All TKs had a graduated rod inside, to monitor the thickness of the sludge layer.

The draining bed of each DB consists of three overlapping layers of inert material; at the base, a layer of 30 cm of coarse gravel (20-32 mm), followed by 10 cm of fine gravel (8-12 mm), and finally, 10 cm of sand (granulometry up to 0.6 mm). Subsequently, the drying beds received a certain volume of sludge: 20 cm high in TKs 1 and 2 and 30 cm for tanks 3 and 4, as shown in Fig 1. These thicknesses were used following the recommendations of Metcalf and Eddy (1995). For

TKs 1 and 3, the sludge was mixed and homogenized daily, with a metal shovel, in order to assess the influence on the dehydration process.

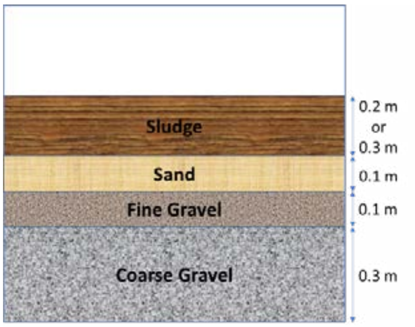

Figure1. ***Materials and composition of the drying bed***

Two dehydration cycles were carried out, each lasting 30 days, corresponding to different seasons (dry season and rainy season). The parameters monitored daily during the tests correspond to: sludge Solid Content (SC); sludge layer thickness; and drained volume of water. SC analyzes, in duplicate, were performed according to Standard Methods (APHA, 1998). The meteorological data were collected at the IPB meteorological station.

The mathematical modeling was elaborated from: the estimate of evaporated water (daily), by energy conversion equations, using meteorological data; the volume of water drained (daily), relative to the system's initial water; precipitation (volume of water) over the bed area. With these data, the water mass balance can be estimated. Fig 2 shows the flowchart for the elaboration of the models, where correction factors were established, in order to correct the evaporation and drainage of water, according to the instantaneous SC each day (Lima, 2020, p. 46 - 55).

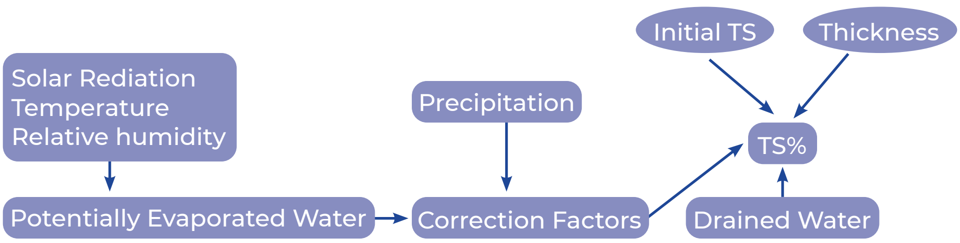

Figure 2. ***Flowchart of mathematical modeling***

For the preparation of the mass balance, initially, the water potentially evaporated was found for each day of the experimental test, and then the volume of precipitated water was determined. Finally, it was estimated the water drained from the beds, referring to the initial water content in the sludge. By the trial-and-error method, observing the approximation of the theoretical curve to the experimental SC curve, some correction factors were attributed to the correction of evaporation and drainage, since these factors are influenced bythe instantaneous SC value of the sludge.

RESULTS AND DISCUSSION

Since each TK had different characteristics, 4 mathematical models were obtained. Fig 3 shows the experimental results (blue line) and the theoretical results (orange line), found from the models, for the 4 DB (cycle 1 and 2, left and right side, respectively). The figure also shows the occurrence of precipitation during the testing period.

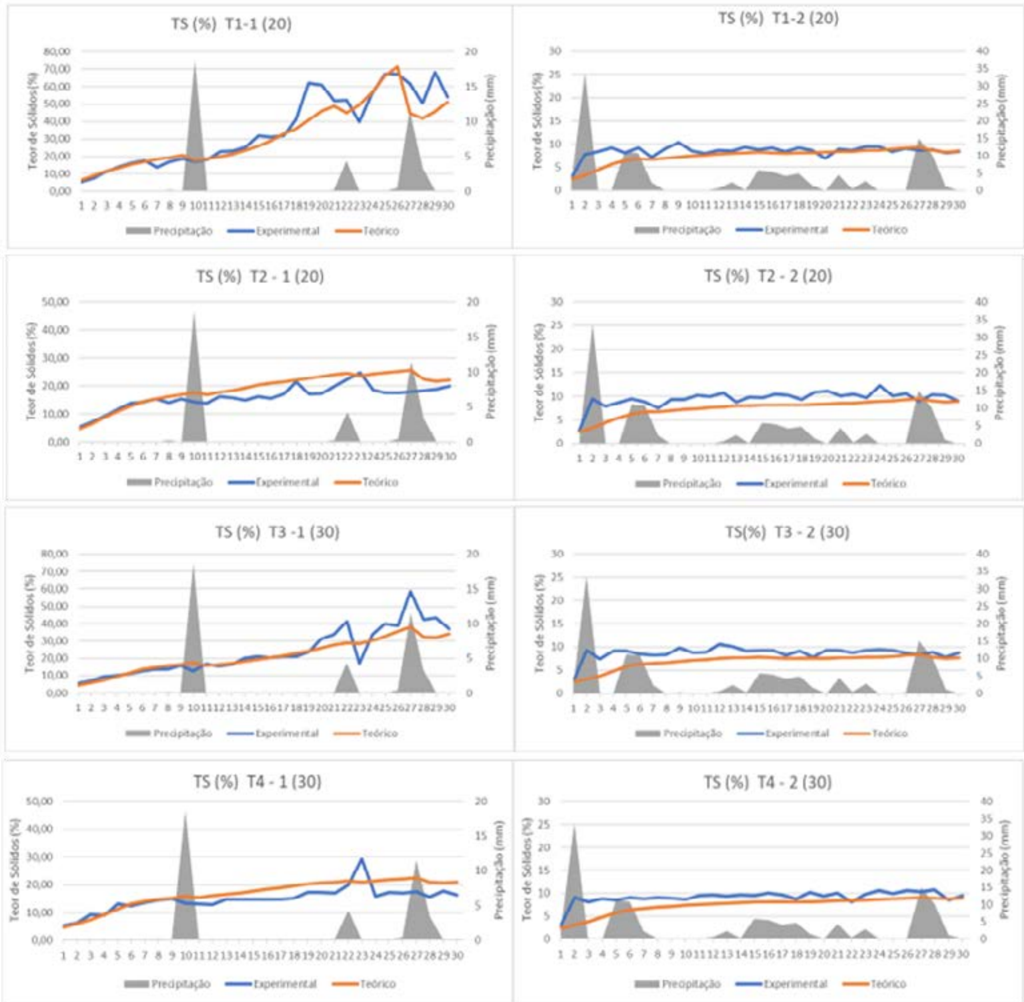

Figure3. *Mathematical models and experimental data (Solid Content)*

An approximation of the theoretical and experimental curves is observed in most of the period, for both cycles, showing that the model is adapted to different climatic conditions. A peak of decrease in SC in the experimental curves occurred after the occurrence of precipitations, when the SC was close to or greater than 20%. This peak is less accentuated in the theoretical curve;however, the two curves return to similar values soon after the event. This factor happens, because experimentally a portion of the precipitation is incorporated in the upper part of the sludge and evaporates later, since the model does not foresee this effect and only considers that it is drained immediately.

In cycle 1, the TK 1 and 2, whose initial layer of sludge was 20 cm, showed better dehydration in relation to the other tanks, as a result of the lower thickness of the sludge layer. It is also observed that for beds where daily homogenization was increased, the SC values were higher than those without turning, such as for DB1, which recorded SC values close to 70% and the TK 2 of only 30%.

In cycle 2, the results showed no significant differences between the tanks, and the SC values at the end of the tests were approximately equal, in the range of 10% - 15%. But it was observed that even in cold and humid weather, the volume reduction was significant, reaching values that exceeded 75%, of reduction of the initial volume, tank 3.Tanks 1 and 2 had volume reduction efficiency of 70% and 75%, respectively. Tank 4, on the other hand, with the lowest performance, obtained a 2/3 reduction. All tanks achieved volume reduction within the range of 50% and 80%, which is expected in sludge dehydration (Hazel, 2015)

CONCLUSIONS

The mathematical modeling of the drying sludge dehydration process proved to be quite satisfactory, since it adequately represented the experimental results in different climatic conditions, following the experimental concentration of SC in practically the entire period of the tests. The drying beds, even in unfavorable climatic conditions for water evaporation, showed a good performance, significantly reducing the volume of sludge from the water that was drained from the system.

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AERATED CONSTRUCTED FLOATING WETLANDS FOR SMALL COMMUNITIES

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Abstract

One of the main disadvantages of constructed wetlands, is the need for a large area to be able to meet the water quality parameters. Within the framework of the LIFE INTEXT project: "Innovative hybrid INTensive – EXTensive resource recovery from wastewater in small communities", it has been designed a system to be able to intensify the process and therefore to reduce the land use.

According to a preliminary experience using an aerated constructed floating wetland in the headquarters of PROJAR Group, the system was monitored and successfully accomplished the discharge requirements. However, despite of the positive results, it cannot be considered concluding due to short period of follow up of the water parameters.

Therefore, a longer and deeper study of the system is needed to be able to adjust, calibrate and validate this technology for commercial purposes. In the experimental plant of CENTA (Centro de las Nuevas Tecnologías del Agua) in Carrion de los Céspedes (Sevilla), it has been designed a pilot case where the follow up can be carried out with quality enough.

INTRODUCTION

One of the main disadvantages of constructed wetlands is the need for a large area to satisfy the limits of discharge, which ends up generating a relevant economic cost and land use compared to conventional systems. With the objective to avoid the these problems, two ways for reducing the space required are explored in this study. First, the elimination of the granular medium through systems with floating vegetation and second, the introduction of artificial aeration.

In 2014 a free-water surface constructed wetland consisting of 8 cells of 8.8 m2 each, with a water depth of 0.4 m was built. The floating system was made of volumetric mesh, polyethylene bars and coconut fiber mattress. The wastewater to be treated came from the offices and warehouses of the company Projar S.A., with a flow of 773 l/d; the inflow came intermittently from a septic tank with two compartments thus, the water reached the wetlands with a previous sedimentation process. This floating system has been already adjusted and it has been found the best adapted vegetation for these conditions.

Preliminary results obtained by including aeration in the wetland system cannot be considered concluding. Therefore, a longer and deeper study of the system is needed to be able to adjust, calibrate and validate mathematical models for its use with guarantees.

After this preliminary study, it has been designed a layout within the framework of LIFE INTEXT project. The pilot case to be implemented in the CENTA (Centro de las Nuevas Tecnologías del Agua) facilities, has been designed to accomplished the water quality standards for the outlet by using the aerated constructed floating wetland system.

Due to the technical and economical limitations of small communities, it has been proposed a robust system that allows small communities its installation and maintenance. This pilot case will treat the water flow for a 150 heq, with approximately 250 m2 of constructed wetland and 1.5 m depth.

The wetland cell has been divided into two longitudinal compartments and at the same time, each of these compartments is also divided in several cells. This design has three objectives:

- Increase the length / width ratio to improve the hydraulic efficiency of the system
- Increase the number of compartments to improve hydraulic efficiency
- Test two different aeration sequences

METHODS

The project of the constructed aerated wetland will be implemented in the municipality of Carrión de los Céspedes (Sevilla), where the CENTA is located. The climate of the area is Mediterranean with an average annual rainfall of about 485 mm. The annual average temperature is 18.8 °C maximum and minimum average temperatures were 5.9 °C and 35.3 °C, respectively.

The existing facilities at CENTA has an Imhoff tank from where the water to be treated will be pumped to the secondary treatment (the aerated constructed wetland). The layout of the constructed wetland treatment is designed in a way that the water can go through two parallel treatment lines. Each individual line can be controlled in terms of water flow, aeration flow and the aeration sequence.

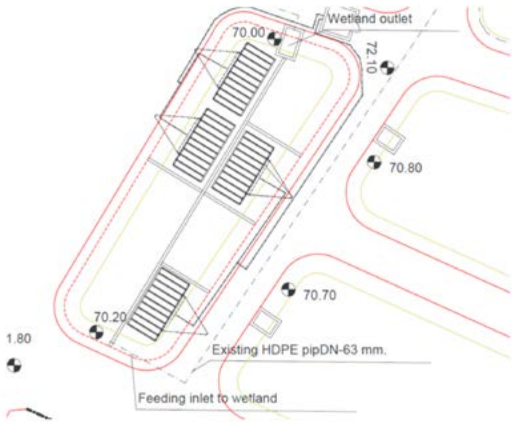
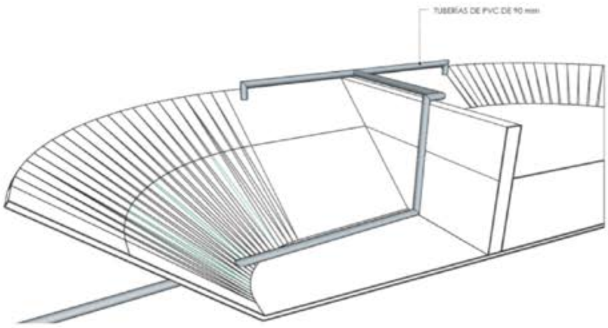


Figure 1. *Constructed wetland layout*

The installation parts are as follow:

- Effluent discharge network through PVC pipes
- Effluent distribution network between the different compartments through flexible PVC pip



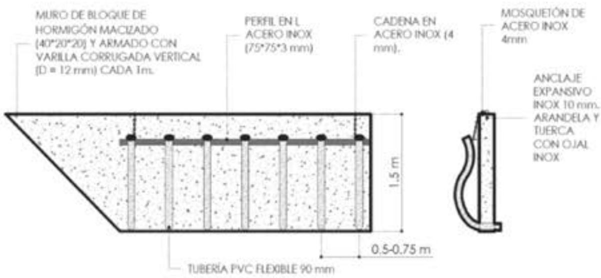


Figure 2. *Input power supply*

Figure 3. *Compartment unit*

- Effluent collector and a levelling system by a manhole with a PVC pipe
- Aeration grid by a 16mm irrigation pipe anchored to a PVC frame.

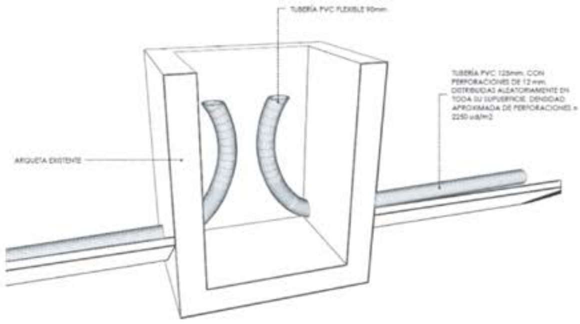


Figure 4. ***Leveling and sampling manhole***

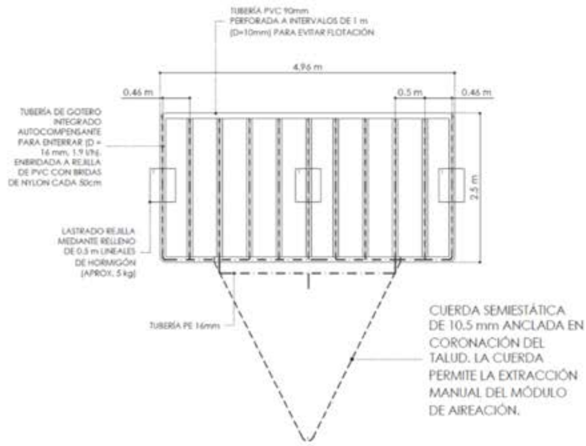


Figure 3. ***Compartment unit***

- Aeration system:
  - a. 2 x Air pump 1.5 kW, 230 V, 210 m3/h, -190/200 mbar
  - b. Electrical distribution panel with controlling system.

The inlet flowrate will me manage according to the needs of the constructed wetland which it is estimated in 40 m3/day. Once the water goes through the wetland, the outflow will be poured to the main water collector of the CENTA facilities.

As a starting point, the design of the constructed wetland will take into consideration the following data:

	Table 1. Population equivalent water parameters	
Parameter	**g/peq*d**	**mg/l**
BOD_5	60	300.00
COD	90	666.67
TSS	90	300.00
Nt	15	40.00
P	4	10.00

The main objective of the project is to obtain a valid and simple mathematic model for the design of aerated floating wetlands which will improve performance and economic viability of these systems.

The water quality model to be tested is:

Where:
Ct, outlet concentration (mg/l)
Co, inlet concetration (mg/l)
k', adjusted constant for one contaminant (1/d)
kmax, maximum constant (1/d)
OD, oxygen concentration (mg/l)
kOD, semisaturation constant (adimensional)
t, reaction time (d)

RESULTS AND DISCUSSION

This multicompartment aerated constructed floating wetland will provide the data to be able to obtain a valid and simple mathematic model for its design. Therefore, it will be improved the performance and the economic viability of this system. Some of the parameters it is aimed to get are:

- Characterization of kinetics constants for urban wastewater.
- Distinction between different aeration levels, trying to find a Monod relation between processes and aeration levels.
- Test different quality water models (plug flow, tanks in series...)
- Characterize oxygen diffusion performance of the aeration system.
- Characterize maximum areal and cross-sectional organic loading rates (gBOD/m2*d)
- Validate non-aerated wetlands previous data.
- Experiment deeper ponds as primary and secondary treatment.
- Experiment recirculation systems to enhance nitrification and denitrification
- Sludge production

According to results obtained in the previous experience in the facilities of Comercial Projar SA headquarters, we observed a remarkable performance comparing it with a traditional constructed wetland system.

In the first instance, it was not applied any aeration system to the constructed wetland. Under these conditions, the discharge requirements of BOD5 (25 mg/l), COD (125 mg/l) and TSS (35 mg/l) were satisfied on average, but BOD5 presented a relative high percentage of non-compliances in specific days.

After this first test condition, it was introduced simple and inexpensive system of aeration to the wetland. Under the same organic and hydraulic load conditions and after three samplings, the results improved considerably, going from the removement 61.5% to 81.7% for COD, and from 83.2% to 93.8% for BOD5.

Table 2. Comercial Projar SA aerated floating wetland results				
Parameter	**Inlet**	**Outlet**	**Ut**	**Performance**
COD	382.00	70.00	mg/l	81.7%
BOD5	200.00	6.00	mg/l	93.8%
Pt	5.60	2.80	mg/l	50%
N amonia	144.00	108.00	mg/l	25%
N nitric	5.80	0.30	mg/l	95%
Nt (estimated)	149.80	108.30	mg/l	28%
TSS	140.00	28.00	mg/l	80%
pH	7.03	7.57	ud de pH	
Conductivity	2500.00	2100.00	µS/cm	

CONCLUSIONS

Once the aerated construction wetland is implemented in the CENTA facilities, the layout is designed in a way that it can be tested different conditions. Therefore, it will be possible to stress the system to check the maximum load capacity of this technology and the most efficient setting.

This experience will last three years, where it will be tested different configurations. The first configuration will last twelve months due to the 6 months of bacteria and vegetation implementation. After this first test, if the results are positive, the aeration flow will be reduced to cut down the energy consumption. Each configuration will be tested a minimum of three months.

In the case of the results of the quality parameters don't meet the expected water parameters, it will be possible to reduce the inflow or increase the aeration flow hours per day. Moreover, it will be possible to switch the order of the aerated phases and de depth of the water level, which it will increase or reduce the hydraulic retention time.

The system was preliminary characterized in the experience at Comercial Projar SA headquarters, and the main conclusions of the experience is that floating aerated systems can be competitive and reliable for secondary and tertiary treatment compared with conventional wetland systems. Nevertheless, this hypothesis is still to be confirmed in the experience that will be carried out in the CENTA installation due to the higher resources for testing and to try different settings.

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CONSTRUCTED WETLANDS FORDOMESTIC WASTEWATER TREATMENT: FROM CLASSICAL "FRENCH" SYSTEMS TO OPTIMIZED INTENSIFIED TREATMENT WETLANDS

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Abstract

Constructed wetlands (CW) for domestic wastewater treatment (also call Treatment Wetlands - TW)are renowned for their robustness and reliability, thus most suitable for small communities. Widely applied especially in western Europe, they are facing the need for more restrictive reject level and decreased footprint. This work reviews the efficiency of classical "French" two-stage vertical flow constructed wetlands (VFCW) and two optimized versions developed by Syntea: Biho-Filter® and Rhizosph'Air® designed for higher removal efficiency and lower footprint. The challenge lies in boosting the system while keeping it simple and resilient. This work shares feedbacks fromSyntea on classical French design and intensified CWs.

INTRODUCTION

"French" constructed wetlands treating raw domestic wastewater ([3]) have reach international recognition these last decades ([4], [8]). These systems are especially well suited for small communities and in the decentralized sanitation context, regardless of the climate ([15]). Integrated sludge management, robustness and simplicity of operation are among the main qualities of French constructed wetlands. However, high land requirements and limited total nitrogen removal efficiency restrict their application in areas with high land pressure or when higher outlet water quality is required ([10]).In this context, Syntea, French SME leader in CWs, has developed two types of intensified treatment wetlands:Biho-Filter® and Rhizosph'Air®. Biho-Filter® combines 2 stacked vertical flow treatment zones, respectively unsaturated (likewise classical systems) and water-saturated (targeting enhanced carbon and TN removal efficiency). Patented system Rhizosph'Air® is an intensified version of French CWs with forced bed aeration. As activated sludge processes, sequential air supply allows simultaneously high ammonia and total nitrogen removal efficacy.

METHODS

The feedback on these processes has been capitalized using the SYNTEA reference database with:

· more than 500 classical French CWs monitored for 20 years ([4]).

· 40 Biho-Filter® systems implemented in France and abroad (from 40PE to 3700PE) since 2012. About 140 24h-composite samples have been proceeded to fulfill municipalities regulatory obligations.

· around 15Rhizosph'Air® systems for domestic wastewater treatment (from 800PE to 4000PE) were built by SYNTEA in France since 2016. About 40 24h-composite samples have been proceeded to fulfill municipalities regulatory obligations. Data has been processed following the methodology described in [4].

RESULTS AND DISCUSSION

The performances of French 2 stages of VFCWsare well documented through SYNTEA database in [4] and [17]. The Syntea'sBiho-Filter® system has been extensively studied in [5] and benefits from European recognition through the Environmental Technology Verification (ETV,[1]). Forced bed aeration constructed wetlands have been applied to a wide range of effluents: from pre-treated domestic effluent ([6], [9] and [16]), to wastewater from petroleum industry ([18]), agroindustry ([13]) or airport runoff containing deicing products ([7] and [17]). They allow to reach lower discharge levels with lower footprint compared to systems with passive aeration. Coming from the benefits of the French design, Rhizosph'Air® has the particularity to also handle raw wastewater with integrated sludge management through surfaceaccumulation and mineralization (composted sludge removal every 10-15 years).Syntea's data on various designs of CW has been processed and allow us the following comparison in terms of outlet concentrations and removal efficiencies (Table 1).

Table 1. *TW characteristics statistics for outlet concentrationsand removal efficiency.*

	Classical French treatment wetland[4]		BiHo-Filter		Rhizosph'Air®	
Design	2 stages VFCW		Single stage VFCW with recirculation		Single stage vertical and horizontal flow CW	
Sizing	1.2 + 0.8 m²/PE		1.5 m²/PE		0.9 m²/PE	
Mean+/-X (N)*SD**	Outlet concentration	Removal efficiency	Outlet concentration	Removal efficiency	Outlet concentration	Removal efficiency
	mg/l	%	mg/l	%	mg/l	mg/l
COD	**74**+/- 16 (417)	**87**+/- 2 (409)	**50**+/- 6 (141) *38*	**92**+/- 1 (141) *7*	**27**+/- 5 (38) *15*	**95**+/- 1 (36) *4*
BOD5	-	-	**11**+/- 2 (138) *13*	**96**+/- 2 (138) *9*	**5**+/- 2 (34) *5*	**98**+/- 1 (32) *2*
TSS	**17**+/- 6 (418)	**93**+/- 1 (411)	**15**+/- 3 (142) *16*	**95**+/- 1 (142) *5*	**5**+/- 1 (34) *3*	**98**+/- 1 (36) *2*
NK	**11**+/- 2 (409)	**84**+/- 2 (357)	**17**+/- 3 (126) *15*	**79**+/- 3 (124) *15*	**7**+/- 2 (37) *6*	**92**+/- 2 (36) *15*
TN	-	-	**32**+/- 3 (126) *18*	**60**+/- 3 (117) *19*	**19**+/- 3 (33) *8*	**74**+/-6 (28) *20*

**Values for 24h composite samples N: number of values X=1.96*SD/√N*

Table 1 gives principal system configurations and results in terms of outlet concentrations and removal efficiencies. These values have been set-up after statistical analysis of SYNTEA database. These results show thatslight betternitrification is achieved in the classical two-stages VFCWs compared toBiHo-Filter, while the latter allows partial TN removal and a decreased land requirement. Classical 2 stages French system achieves almost a full nitrification. but requires a sand filtration layer for which quality and resources are less and less available. This underlines the relevance of Biho-Filter®, especially when only partial nitrification is needed.

Rhizosph'Air® results are far better than other TWs with a reduced size, due to the forced aeration and better oxygen transfer rate. These results for nitrification and carbon removal are consistent with literature on forced bed aeration constructed wetlands ([9], [11] and [16]). Concerning TN removal, a specific adaptation of the sequential air supply makes it possible to reach outlet consentlower than 15 mg N/L ([12]).

Furthermore, EU LIFE INTEXT project, leaded by Aqualia,aims at developing technologies combining advantages of intensive and extensive processes. Syntea has implemented at demonstration-scale high TRL technologies of aerated constructed wetlands aiming at**(1) optimizing TN removal in CWs**. The lack of carbon source is often reported to be a limiting factorfor denitrification ([11], [13]).; **(2)Fostering pathogen removal in CWs.** Pathogen removal in conventional constructed wetlands is very low with 1 log unit removal per stage of treatment([2]) while forced bed aeration shows promising results ([2], [14]).**(3) Demonstrating the combination of intensive anaerobic pre-treatment (biogas production) developed by Aqualiaand CWs intensified or not implemented by SYNTEA**.

CONCLUSIONS

This work focuses on the comparison between classical and intensified TW using the full-scale treatment plants data base ofSyntea. Intensified TW proved to offer an optimized land requirement while preserving outlet water quality. Rhizosph'Air® particularly shows very high ammonium and TN removal thanks to forced aeration but increasing the same way the energy costs. This will be address with LIFE INTEXT research program through the development and validation of smart monitoring and operation control tool. Thiswill offer a very high added value to next generation constructed wetlands.

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WATER QUALITY ASSESSMENT OF URBAN PONDS AND REMEDIATION PROPOSALS (SMALLWAT21)

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Abstract

The objective of this study is to assess the quality of the water of six urban ponds in the city of Lisbon, Portugal, to determine the factors that influence it and provide remediation measures for them.

The main factors that contribute to the poor quality of the water were identified, and are the following: excess nutrients, origin of water, waterbirds, vegetative debris that fall in ponds and contamination with sewage.

Three of the ponds had floating treatment wetlands installed before the start of the study, but additional measures such as removal of bottom sediments and leaves in the fall, may be necessary.

INTRODUCTION

Lakes and ponds are one of the landscape features that significantly contribute to increasing the quality of life in urban centers, by increasing amenity, providing recreational and educational activities, and even contributing to mitigate the urban climate [1], [2].

However, they are extremely vulnerable landscape elements, sensitive to anthropogenic pressures since their watershed is part of the urban fabric and, therefore, tend to emphasize the

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environmental problems that affect metropolitan areas, by collecting and accumulating large amounts of nutrients that can lead to the proliferation of toxic cyanobacteria [3] and pollutants, including microbial contaminants [4], [5].

In the urban landscape excessive concentrations of nutrients have different origins, being partly attributed to land use, runoff from construction sites [6], buildings and impervious paved surfaces [7], runoff from recreational areas (eg. golf courses) [8], feeding ducks [9] and runoff from parks and gardens [10]. Sediments from the bottom of lakes have long been recognized as a potential source of phosphorus in surface waters and are an important factor in regulating eutrophication in shallow lakes [11].

The objective of this study is to evaluate the water quality of six ponds in the city of Lisbon and to propose appropriate remediation techniques for them. To this end, several water parameters were measured over a 6 month period between July and December 2020 and chemical analyzes carried out. Lisbon Municipality provided additional data on the water quality of these ponds. Three of the six ponds had FTWs installed before the study began.

METHODS

The case study includes six ponds in the metropolitan area of Lisbon: Quinta das Conchas (QC), two ponds in Parque Oeste (PO3 and PO5), Doca da Caldeirinha (DC), Jardim da Estrela (JE) and Estufa Fria (EF). All of the ponds are artificial, with a small surface area (between 465 and 17000 m^2), shallow, and with the exception of DC, they are inserted in public gardens, so they are surrounded by lawn areas and trees. Ponds DC, JE and EF had floating treatment wetlands (FTW) prior to this study, installed between November of 2019 and March 2020. The area covered with the FTW was 9 to 12% of the total surface area of the lakes.

To assess the quality of the water of the ponds, sampling occurred over a 6 month period as follows: 22/07/2020, 26/09/2020, 10/10/2020, 14/11/2020, 12/12/2020, 19/12/2020. Surface water was collected with a bucket near the margin of the ponds and in the field four parameters were measured with probes: temperature, pH (HACH Sension+ 5051T), electric conductivity (HACH Sension+ 5060) dissolved oxygen (YSI ProODO). Total suspended solids and chemical oxygen demand were determined in the laboratory according to standard methods [12]. The Lisbon Municipality provided reports from previous years with other parameters which were also taken into account for this analysis.

Since there is no legislation that controls the quality of the water for small ornamental ponds like the ones in the city of Lisbon, reference values were adopted the decree-law nº 236/98 of 1 of August, Anex XXI, which defines the minimum quality to be attained in surface waters. The reference values for each parameter studied are presented in Table 1.

RESULTS AND DISCUSSION

The study reveals a clear picture of the status of water quality of the different ponds. Table 1 presents the minimum and maximum values obtained for each parameter when data was available.

Table 1. *Minimum and maximum values registered for each parameter for the 6 ponds.*

Parameter	Reference Values	QC	PO3	PO5	DC	JE	EF
pH	5.0-9.0	8,2 - 8,81	6,8 - 8,65	6,8 - 9,28	7,71 - 10,4	7,5 - 8,4	7,4 - 9,51
Chlorides (mg Cl/l)	250	82	126	54 - 126	246 - 20413	21 - 26	16 - 41
Nitrates (mg NO_3/l)	-	1,2	12,87	<0,22 - 20,02	0,05 – 4,2	0,33 - 1,24	1,91 - 10,18
AN (mg N/l)	1	-	-	-	0,12 - 16,1	<0,1	<0,1
KN (mg N/l)	2	-	-	-	0,44 - 18,32	0,98 - 2,82	2,47 - 5,27
BOD5 (mg O_2/l)	5	-	-	-	3 - 114	6 - 14	5 - 20
COD (mg O_2/l)	-	30 - 67	17 - 56	25 - 59	32 - 97	37 - 99	5 - 136
TP (mg P/l)	1	-	-	-	0,05 - 3,53	<0,125 - 0,34	<0,125 - 0,17

All the ponds show pH above neutral and the sampling period allowed to identify a trend of higher pH during the warmer months with a decrease in colder months. This variation could be explained with algae photosynthesis in the summer increasing the pH, and leaf litter decomposing in the autumn decreasing the pH [13].

All the ponds are contaminated with nutrients, this could be due to the presence of waterbirds as well as leaf litter that decomposes in the water. For PO3 and PO5 the well water that feeds these ponds is contaminated with nitrates (12,7 to 50 mg NO_3/l) which is not conducive to good water quality. In pond PO5 a sewage discharge was identified. Waterbirds and their feeding are known to be sources of nutrients for waterbodies [9]or thought to be relatively unimportant. However, when bird populations are large relative to the size or volume of the waterbody, a substantial fraction of the nutrient pool may cycle through birds. In this report, we estimate the mass of phosphorus in bird droppings cycling through Green Lake, a productive lake in urban Seattle, between January 1992 and December 1994. Phosphorus loading from bird droppings is compared to the mass of phosphorus entering the lake from other sources measured during a concurrent limnologic study. Waterbirds spent 528,355 bird-days (the number of bird-use days. In JE feeding the ducks is common practice among visitants with children and should be discouraged.

BOD5 exceeds the limit for all the ponds in which it was measured, and COD presents a wide range of values from 5 to 136 mg O_2/l, with higher concentrations in summer. The lower concentrations could be due to rainfall and consequent dilution, however this is not enough to explain it since the concentrations decrease before rainfall events.

TSS were especially high for JE (in September 2020) and PO5 (in December 2020). Since chlorophyll-a was not determined it is impossible to know for sure if the high TSS observed in JE could be due to higher levels of phytoplankton, which is possible since the temperatures were still high by that time. PO5 in the other hand had high concentrations of TSS most likely due to sewage discharge and stormwater runoff contributions.

DC's surrounding environment is very different from the other ponds, since it is not included in a garden, all pavement surrounding it is impermeable. It's very close to a road and the Tagus river estuary, receiving water from it. This explains the high salinity and could also be the main source of pollutants to this lake.

JE, EF and DC had FTWs installed before the start of the study. JE and EF had different plants from DC, since the later had higher salinity, therefore it received native plants to the Tagus estuary. Different growth rates were observed, with the FTW in DC growing significantly more from July to December when compared to the other 2 ponds. DC experienced a visible reduction of nutrient levels in 2020 compared to previous years possibly due to the FTW. For the other 2 lakes more data is needed.

B. Masters [14] suggests dredging and removal of sediments at the bottom of lakes as a complementary measure to FTWs if phosphorus removal is expected in the medium/long term, since the main removal of phosphorus is achieved by sedimentation and not plant uptake (20-40% vs 6% [15]).

CONCLUSIONS

The results obtained were as expected for ponds in urban landscapes: artificial, small and shallow, impermeable surface surrounding them, high nutrient loads that contribute to eutrophication, sewage discharge, bird populations that gather in these places where they are fed by visitors, cloudy water with an aesthetically unpleasant aspect and devoid of macrophytes.

The main contaminants of these waters are nutrients, and their possible sources were identified as being leaf litter that decomposes in water, surface runoff, waterbirds, feeding of waterbirds by visitants, origin of water and sewage discharge.

The new goals for the water quality of these ponds haven't been achieved for the time being but the FTW showed promising results in DC. Dredging and sediment removal might be necessary as a complementary measure to the FTWs. Nutrient loads reduction should be the priority action for all the ponds before pursuing other remediation techniques.

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02

ANAEROBIC TREATMENTS

ANAEROBIC TREATMENTS: ADVANCED TECHNOLOGIES FOR WASTEWATER TREATMENT IN SMALL URBAN SETTLEMENTS

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Abstract
Anaerobic technologies can be adopted in order to reduce the investment and operating costs of wastewater treatment plants of small urban settlements. Though anaerobic technologies are more adapted for industrial wastewaters or solid waste with high biodegradable organic matter content, special designs of anaerobic bioreactors are available to deal with low-strength wastewaters at sub-optimal temperatures. The Upflow Anaerobic Sludge Blanket (UASB) and the Anaerobic Membrane Bioreactor (AnMBR) technologies are presented and discussed as alternatives for wastewater treatment in small urban settlements.

FRAMEWORK OF CONVENTIONAL WASTEWATER TREATMENT

The conventional paradigm for urban wastewater treatment involves the following stages:

- Wastewater is collected within a wide area, from homes and businesses served by a sewage infrastructure, and is directed to centralized Waste Water Treatment Plants (WWTP).
- Once in the WWTP, the water is subjected to a treatment process consisting of several interconnected unit operations and unit processes
  - The main objectives of the process are the elimination of biodegradable organic matter and suspended solids
  - At the "heart" of the process is an aerobic biologic process such as activated sludge
  - Excess nutrient removal and thorough disinfection are only applied to the effluent of the biologic process if required by the discharge license
- The treated effluent from the WWTP is discharged to a receiving water body (river, creek, lake, ocean...)
- Waste sludge generated by the process is stabilized on site or shipped somewhere else for management.

The main disadvantages of this concept are:

- Sewage collection systems which have many kilometers in length, need lift stations and other facilities, are costly to build and maintain, and are sometimes unreliable
- Expensive treatment on the WWPT, especially concerning power and maintenance and repair demands
- There is a potential negative environmental impact of the discharge of treated water into receiving water bodies, due to incomplete treatment or the presence of traces of pollutants of emerging concern.
- Waste sludge is generated in high quantities and needs stabilization on site or shipped somewhere else for that objective,involving elevated investment and operation costs or transportation costs and treatment fees.
- There is no recovery of resources from the wastewater, such as energy, or nutrients that can be cycled back to the food chain, or the treated water itself.

Small urban settlements cannot afford the costs of such treatment solutions, because they involve the use of skilled labour and intense material and technological demands, and the mobilization of significant financial resources to build and operate such facilities. The economic viability of these solutions can only be assured by taking advantage of economies of scale. The choice has to fall upon less demanding, extensive technologies which are still able to meet the treatment demands. In the extreme case of isolated dwellings, the septic tank system (a typical anaerobic process) and its modern variants are still the best solutions available.

ANAEROBIC PROCESSES:GENERAL CHARACTERISTICS

Anaerobic wastewater treatment is carried out by a diverse consortium of microorganisms able to carry out sequentially1) hydrolysis of complex organic matter; 2) fermentation of hydrolysed material into methanogenic substrates and 3) methanogenesis with generation of biogas (a mixture of CH_4, CO_2 and traces of other gases). These processes carry out, under controlled and confined conditions, the biodegradation metabolism that would occur on the natural environment in oxygen-deprived water bodies impacted by wastewater, under septic conditions. The biodegradable organic matter on wastewater is mainly converted to biogas, with conservation of most of the chemical energy in the form of methane, and only a small faction being incorporated into biomass. Anaerobic processes bring the promise to reduce treatment costs and resource demands by presenting the following advantages:

- No need for oxygen to be transferred to the water, therefore saving most of the electric power consumption of wastewater treatment
- Low production of waste sludge, around only one third of the production of aerobic systems, (Chernicharo 2007), because biomass growth is minimal under anaerobic conditions
- Anaerobic reactors are compact and require low area for installation
- These processes are able to withstand large variations in organic load, and can even operate without feeding for limited periods
- Operation cost is kept low, minimal intervention on the process when stable operation is achieved
- Biogas can be recovered and used as energy source to produce heat, electric power, transportation fuel or injection on the gas grid infrastructure

The main disadvantages of anaerobic processes, which prevent their widespread use for domestic wastewater treatment are:

- Kinetics of the process is highly dependent on temperature. Optimal temperature is around 35ºC, which would require heating of the bioreactors. Sub-optimal temperatures, preferably above 20ºC, are acceptable at the expense of system compactness.
- The efficiency for BOD_5 removal is limited.This is especially so at sub-optimal temperatures, and also because settling of anaerobic biomass is much more difficult compared with aerobic processes.
- Long start-up time. It takes long to establish an equilibrated and adapted consortium of microorganisms able to carry out all the metabolic tasks required.
- Very poor nutrient (N and P) removal. As biomass grows less, assimilation of these elements is limited.
- Sensibility to toxic chemicals and other effects. Besides temperature, the anaerobic consortium is usually very sensitive to the presence of toxic chemicals such as sulphide and ammonia, that can be generated endogenously, or excess salinity, the presence of toxic

organics and heavy metals or the lack of alkalinity which would lower pH within the system due to the production of large amounts of CO_2.

– There is the potential for the release of odorous and corrosive gases, such as sulphide, ammonia and other sulphur on nitrogen containing gases

– Need for enclosed reactors and biogas handling and storage facilities. Direct contact with the atmosphere has to be prevented in order to minimize oxygen entrainment, avoid methane emissions to the atmosphere, and to be able to recover, store and process the biogas evolved.

This significant set of disadvantages dictates that the niche for the use of anaerobic wastewater treatment processes is the pre-treatment of industrial wastewaters with high organic loads (COD > 2000 mg/L), because under these conditions the large amount of biogas generated can be used to keep the reactors warm, near the optimal temperature of 35ºC. These processes are also used to stabilize biodegradable waste, such as waste sludge, food waste or agricultural, animal production, forestry or gardening wastes, often carrying out co-digestion of different waste materials. The typical processes for wastewater treatment, such as complete-mix, contact process, sequencing batch reactors, up flow or down flow biofilm processes can only be used efficiently on these applications.

ANAEROBIC PROCESSES: ADVANCED SOLUTIONS FOR SMALL URBAN SETTLEMENTS

Anaerobic processes have to be adapted in order to be used for domestic wastewater treatment, so that sub-optimal temperatures do not lower removal efficiency. The solution for this problem lies on designs able to keep a high inventory of anaerobic biomass inside the bioreactor while preventing the escape of biomass flocs with the treated reactor effluent. Two such designs, are:

– The Upflow Anaerobic Sludge Blanket (UASB) reactor

– The Anaerobic Membrane Bioreactor (AnMBR)

These reactors can be operated both at large and at small scale, and thus are adaptable to the use in small urban settlements generating wastewater with broad variations in instantaneous flowrate and composition.

UASB process

The UASB reactor (scheme in figure 1) consists of a tall equipment with the following characteristics:

– A dense granular sludge bed is present on the bottom of the tank, containing the microorganisms needed for the treatment. A very high content of solids (50 – 100 g TSS/L) is present on this layer. Above, is the sludge blanket, a more diffuse layer (5 – 40 g TSS/L). Such high concentrations of biomassensure fast biodegradation kinetics, even at temperatures around 20ºC, and allows high volumetric organic loading to be fed to these reactors. Long SRT is obtained, which helps remove slowly biodegradable organics.

– The affluent to the reactor is fed from the bottom and has to be uniformly distributed along the entire cross section area of the reactor to avoid fluid channelling and the existence of dead zones.
– The upflow velocity of the liquid phase has to be controlled to keep a suitable mixing without dragging excessive amounts of solids.
– On the top of the reactor a carefully designed triphasic separator is installed to ensure that biogas is recovered and no excessive solids are dragged by the effluent water. Liquid phase discharge has to avoid excessive linear velocities also to avoid dragging floating solids.
– Safety of the gas-handling facilities has to be ensured in order to avoid oxygen entrainment, methane emission to the atmosphere and fire and explosion risks.

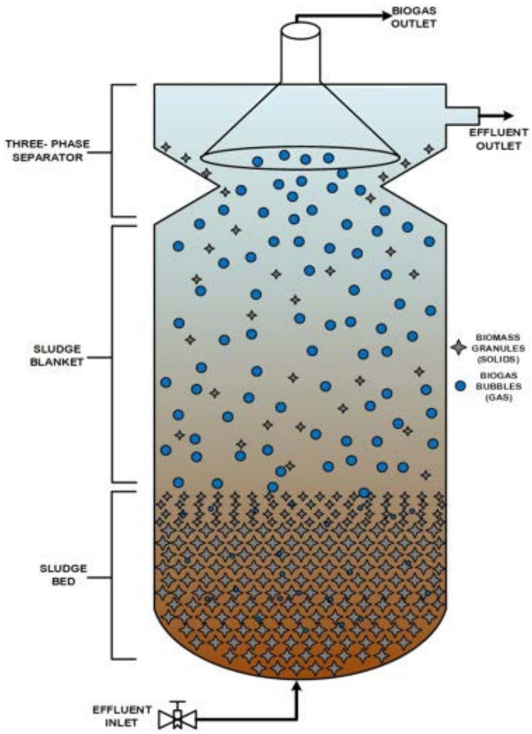

Figure 1. *Schematic of Up flow Anaerobic Sludge Blanket (UASB) reactor (D´Bastiani et al., 2021)*

Such a design requires that the reactors are very tall in order to provide enough residence time for the effluent and efficient separation of the three phases (sludge, water and biogas). Typically, the triphasic separator is responsible for over 30% of total reactor height. The main difficulties of this design lie in providing enough mixing energy to minimize mass-transfer limitations without causing solids dragging by the effluent water, and efficient recovery of methane, which solubility is not negligible in wastewater, especially at lower temperatures.These difficulties are tackled with the Pulsed Solids Hydrolyzer (PUSH®) design patented by FCC-Aqualia SA.

In this design, the feed is done with short high velocity pulses that provide mixing energy without destabilizing the sludge bed. The lower sludge bed has solids retention time (SRT in excess of 30 days, which will ensure hydrolysis of complex organic matter even at a temperature of about 15ºC. The effluent water is subjected to stripping to recover dissolved methane, thus helping further decrease residual BOD_5 in the effluent water. Pilot prototypes of this design were installed and operated in two WWTPs managed by Águas do Algarve SA within the framework of the IDIAqua (*Potenciación de la I+D+i de excelenciaenmateria de depuración de las* águasenpequeñasaglomeracionesurbanas) project.

The effluent of a UASB reactor requires polishing, specially to ensure the microbiologic quality of treated water (pathogen removal). Polishing solutions adapted to the reality of small urban settlements can be the use of polishing ponds, overland flow systems or artificial wetlands. Poor nutrient removal is not a disadvantage considering the treated wastewater may be reused for farm irrigation. This water reuse is crucial in dry climates such as that of south Iberia, and the nutrient content of treated water helps minimize the need for application of synthetic fertilizers, thus recovering another resource from wastewater.

ANMBRPROCESS

The AnMBR (Scheme in figure 2)is a compact equipment that carries out complete secondary wastewater treatment, including water clarification due to its characteristics:

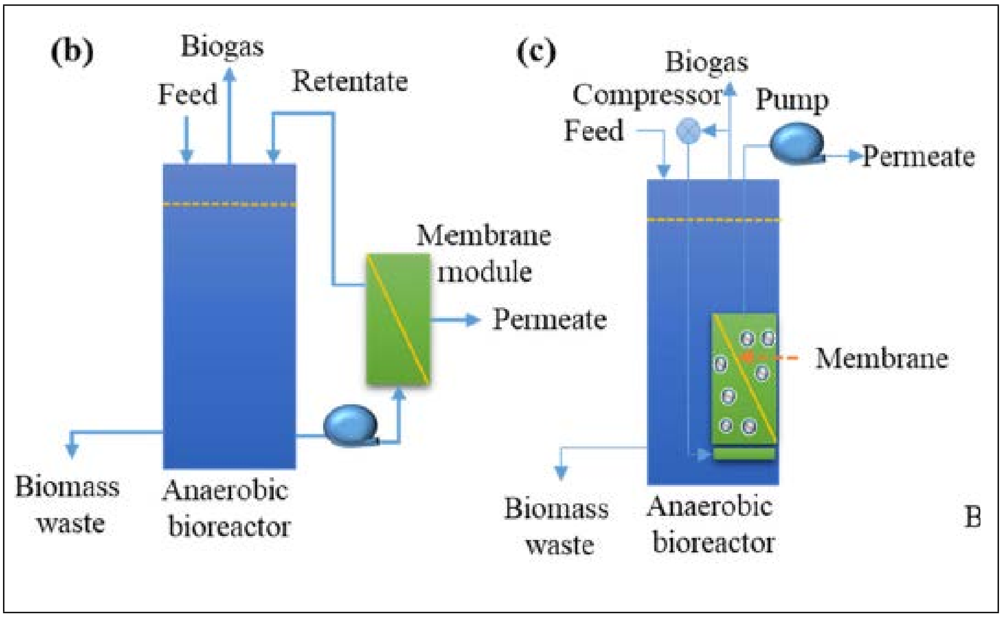

Figure 1. *Schematic of Anaerobic Membrane BioRactor (AnMBR) a) Side-stream AnMBR; b) Submerged AnMBR (Adapted from Maaz et al., 2019)*

– The water leaves the reactor via a semi-permeable membrane module. Ultrafiltration or Microfiltration membranes are usually employed.
– The membrane module is external to the bioreactor (figure 2a) or submerged (figure 2b).
– In 2a) a recirculation pump is needed to provide tangential flow and transmembrane pressure, while in 2b) a vacuum pump is used on the permeate side, limiting transmembrane pressure to a maximum of about 1.0 bar and requiring biogas sparging below the membrane to promote up-flow tangential water movement
– This system precludes the use of gravity solids separation, which often limitingin other anaerobic processes by needing large area for implementation and has issues with poor settling or floating biomass flocs.
– There is a near total solids retention by the membrane
  – Very high biomass concentrations are attainable within the reactor (above 20 g TSS/L is possible). Such high concentration compensates for sub-optimal temperature operation in domestic wastewater treatment. These are small, compact systems with no need for large secondary settlers. Very high SRT (up to 20 d) is obtained. Waste sludge is produced in small, well stabilized volumes, and slowly biodegradable organics are removed.
  – Effluent water has excellent quality, with low turbidity, SST, BOD_5 and pathogen contents. This is ideal for water reuse for farm irrigation in small communities.
– Low dissolved methane content on treated water. The use of specific membranes or vacuum operation allow for a higher energy recovery yield and lower greenhouse gas emissions.
– Reliable treatment process. As long as the operational conditions of the membrane modules are respected, the quality of treated water will be consistently high.

The AnMBR design implies also a number of significant disadvantages and limitations:

– Membranes are subjected to fouling. Membrane surfaces and pores are subjected to the accumulation of insoluble material (biofilms, precipitates, adsorbed biomolecules such as proteins and polysaccharides), which hamper liquid flow across the membrane. The flux across the membrane decreases with time as fouling accumulates, creating the need for counter measures.
– Investment cost significant. Though to a great extent compensated by system compactness Operation and maintenance costs significant. Energy for recirculation and/or transmembrane pressure providing pumps; maintenance of the membrane modules, especially to counter fouling; periodic replacement of the membranes

The main fouling control measures applied are:

– Gas scour. Biogas generated in the process is compressed and bubbled along the membrane surface to minimize particle attachment and biofilm growth.
– Granular activated carbon abrasion. Suspended GAC particles erode biofilms and deposited fouling. Porosity supports biofilm growth within the particles, further increasing SRT, which, together with adsorption, helps remove recalcitrant organics.

– Frequent periodic backwash. Helps detach fouling on membrane surface or within its pores.
– Frequent periodic chemical washing of the membrane. For short periods, using dilute solutions of chlorine, hypochlorite or citric acid, often used as backwash solution, does not involve removing the membrane module from the system.
– Maintenance chemical washing of the membrane. Requires removal of membrane modules and installation of spares. The modules are treated in more aggressive chemical baths: concentrated chlorine, caustic soda, detergents or enzymes may be used. Cleaning may last up to 24 h.

These investment and maintenance costs may preclude the use of AnMBR technology by small urban settlements. However, given that maintenance washing for fouling control is only periodic and that automation and remote sensing minimize the need for local supervision, a single team of skilled operators can reasonably handle several facilities in separate nearby urban settlements, thus reducing the overall operating costs.

CONCLUSIONS

Anaerobic wastewater treatment processes bear the promise to reduce operating costs by avoiding aeration energy costs and producing minimal amounts of waste sludge. However, special designs are needed to counter their intrinsic disadvantages, especially the kinetic effect of sub-optimal temperatures and the poor settling characteristics of anaerobic biomass. The best solutions for small urban settlements are the UASB / polishing post-treatment and the AnMBR alternatives. Both designs can be installed at small scales, making it possible to use the in small urban settlement contexts. Both designs counter the effects of low temperatures by keeping a large concentration of biomass inside the system, and while for the UASB the solids removal has to be complemented by a polishing post-treatment, for the AnMBR the use of membranes that totally reject solids tackles the problem, directly providing water with enough quality to be used in farm irrigation. The special maintenance and operation requirements of these systems can better be afforded by small communities with the use of automation, remote sensing and information technologies, so that a single team of operators can handle a small number of nearby facilities using similar technologies. Taking into account that these technologies also provide resources recovered from wastewater – irrigation water, fertilizing nutrients in both treated water and stabilized sludge, and energy in biogas – they may help small rural communities meet some of their special demands in a sustainable, circular economy concept.

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PERFORMANCE ANALYSIS OF INNOVATIVE UPFLOWANAEROBIC REACTOR AT THE LAGOS AND LOULÉ WWTP

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Abstract

The interest in the use of UASB (*Upflow Anaerobic Sludge Blanket*) reactors for the treatment of urban wastewater has been growing, especially in tropical and subtropical regions with wastewater temperatures above 20°C. However, in countries with slightly lower temperatures at certain times of the year (15°C) the technology has rarely been tested, which is why Águas do Algarve, S.A. has advanced with the performance of prototype-scale tests. For this purpose, anaerobic reactors, designed and patented by FCC AQUALIA SA, called *Pulsed Solids hydrolyser* (PUSH®), were used. These reactors were developed to overcome the performance limitations of UASB reactors in the treatment of urban wastewater with temperatures below 20°C, as in the case of winter in the Algarve region. Thetestswerecarriedoutundertheframework of theprojectsIDIAQUA – *Potenciación de la I+D+i de excelencia en materia de depuración de las* águas *en pe-*

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*queñas aglomeraciones urbanas*andPREDÁQUA - *Pré-tratamientos en Depuración de Aguas Residuales Industriales y Domésticas: Contribución al Desarrollo Económico,* respectively at the Lagos and Loulé WWTP. At the Lagos WWTP, whose effluent has high salinity (conductivity in the range 2 - 30 dS / m), the system performance was quite poor, fundamentally reflecting the low COD / SO_4^{-2} ratio (1.8). As for the Loulé WWTP, which has no saline intrusion (conductivity in the range 0.5 - 2.7 dS / m; COD / SO_4^{-2} ratio of 25.4) and in which the effluent generally has a high concentration in organic matter (BOD_5 of 630 mg / L) the results obtained were excellent, with removal of COD and BOD_5 above 80%.

INTRODUCTION

Anaerobic processes, due to their economic viability, have been applied in the treatment of wastewater with higher concentrations of organic matter (typical COD values in the range of 1500 - 2000 mg/L). However, technologies such as EGSB (*Expanded Granular Sludge Bed*) and UASB (Upflow Anaerobic Sludge Blanket) have been identified for some years as promising for treatment of wastewater with lower concentrations of COD (Brito and Melo, 1997). Anaerobic digestion in urban wastewater treatment depends heavily on the temperature of the wastewater, given the lower activity of anaerobic microorganisms at temperatures below 20°C, so, naturally, this technology is more attractive in tropical and subtropical countries (Chernicharo, 2007). The main advantages and disadvantages of anaerobic processes are summarized in Table 1.

Table 1. *Advantages and disadvantages of anaerobic processes (Chernicharo, 2007)*

Advantages	Disadvantages
Low sludge production, about 3 to 5 times lower than aerobic processes.	Anaerobic microorganisms are more susceptible to inhibition due to the presence of numerous compounds and to adverse temperature conditions.
Low energy consumption, usually associated with pumping the influent.	The start-up of the process may be slow, if anaerobic biomass is not inoculated.
Reduction of construction costs.	Possibility of emitting odors, which can be treated.
Smaller area for implementing biological treatment.	The biochemistry and microbiology of anaerobic digestion is complex, requiring further studies.
Produces renewable energy in the form of biogas.	Post-treatment needed for effluent polishing.
High tolerance to organic loads.	Low removal of nutrients and pathogenic microorganisms.
Small and large WWTP application.	
Low consumption of nutrients.	
Removal of COD and BOD_5 in the order of 65 to 75%.	

In anaerobic systems, most of the biodegradable organic matter is converted into biogas (70 to 90%), which can be used as a renewable energy source, and a small part of organic matter (5 to 15%) is converted into biomass, resulting in a reduced amount of excess sludge, generally more concentrated and easily to dewatered(Figure 1).

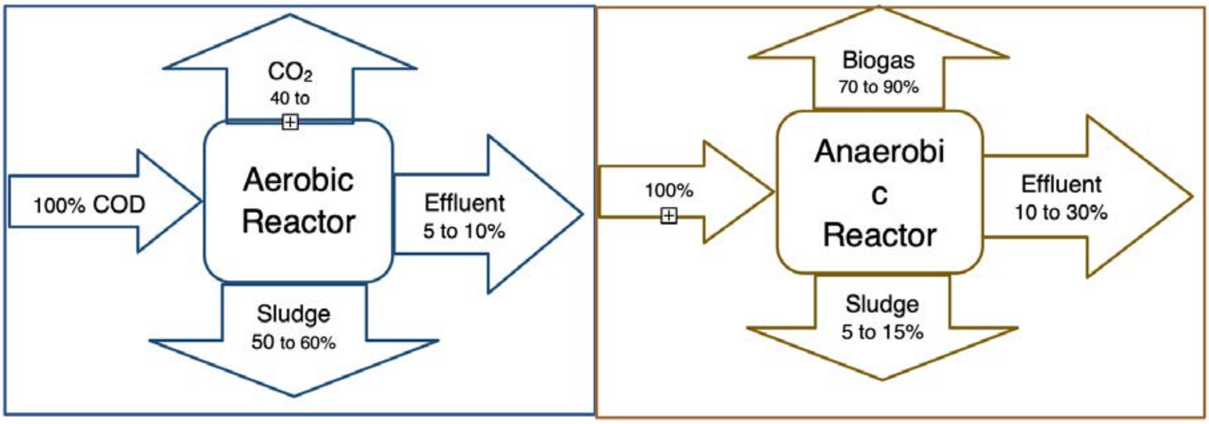

Figure 1. *Biological conversion of aerobic and anaerobic systems(Chernicharo, 2007)*

Most of the WWTPs offer the solution of biological treatment by activated sludge, so the application of anaerobic digestion in the liquid phase, upstream of the biological treatment, with COD removal efficiencies in the order of 65%, will allow not only a reduction in energy costs associated with the aeration stage, which represent about 40 to 60% of the total energy consumption at the WWTP, as well as a lower sludge production.

METHODOLOGY

Within the scope of IDIAQUA project activities (A4 - *Plan de ProyectosDemostrativosInnovadores*) e PREDAQUA (A3 - *Construcción e instalación de infraestructuras de pre-tratamiento: obra civil y maquinaria)*, Águas do Algarve SA installed, in the treatment of the liquid phase, prototypes of anaerobic digestion, with the objective of testing this technology in two WWTPs, which have different influents characteristics: the Lagos WWTP with influentsubject to saline intrusion and the Loulé WWTP, with relatively high concentrations of organic matter.

To this end, in January 2019 Águas do Algarve SA launched the Public Tender: “Design, Installation, Start-up and Monitoring of the Upflow Anaerobic Reactor Prototype UASB(WWTP in Lagos - Lot 1 and WWTP in Loulé - Lot 2), and the contracts were awarded at the end of May 2019 to the company FCC AQUALIA S.A, with a duration of 24 months. The anaerobic reactors, designed and patented by FCC AQUALIA SA, called Pulsed Solids hydrolyser (PUSH®), were developed to overcome the performance limitations of conventional UASB reactors in the treatment of wastewater with temperatures below 20°C, as is the case with winter in the Algarve region, and low concentrations of organic matter in the affluent wastewater. PUSH® anaerobic reactors have the following specific characteristics:

1. An upper tank used to feed the reactor by pulses, ensuring a high mixing energy in the sludge bed to get a good contact between the organic matter and the biomass. As a consequence, it promotesa higher organic matter removal efficiency and a higher biogas production. Also, this pulsed feedavoids the obstructions in the inlet pipes due to the high flow velocity reached during the pulse.
2. A cold digestion zone at the bottom to achieve solid retention times greater than 30 days, in order to reach the hydrolysis of the suspended solids, especially working with wastewater temperatures below 16°C.
3. A stripping system for recoveringthe dissolved methane from the reactor effluent.

The start-up of prototypes in the Lagos and Loulé WWTPs took place in January 2020 with the filling of anaerobic sludge in the reactor, and its continuous operation started in February 2020. During the operation period, monthly laboratory analyzes were performed on the physical-chemical parameters (organic matter, solids, sulfates) in the wastewater at the inlet and outlet of the prototype, quality of biogas and biomass concentration in the reactor. The prototypes were also equipped with continuous probes allowing the monitoring of the conductivity in the wastewater fed to the prototype, and pH, temperature and suspended solids in the treated wastewater.

Figure 2. *Photos of the UASB PUSH® prototypes installed at the Lagos (a) and Loulé (b) WWTP*

RESULTS AND DISCUSSION

The PUSH® prototypes operated in an initial phase, from Feb-20 to Jul-20, with hydraulic retention times of 36 hours, in order to promote better conditions for the formation of biomass in the anaerobic digester, and are currently operating with 12-hour hydraulic retention times. In terms of energy consumption, the installation had a specific consumption of only 0.13 kWh / m^3, which is in line with the low energy consumption expected in these systems (Chernicharo, 2007). Tables 2 and 3 show the results of the monitoring carried out during the operation phase of the prototypes at the Lagos and Loulé WWTP.

Table 2. *Removal efficiencies and physico-chemical characterization in the wastewater fed to the prototype and treated wastewater at the Loulé and Lagos WWTP*

Parameters	PUSH® Lagos with saline inflows			PUSH® Loulé with high organic loads		
	Feed effluent	Effluent Treated	Removal efficiencies	Feed effluent	Effluent treated	Removal efficiencies
pH	7.5	7.5	-	7.3	6.9	-
COD (mg/L)	594	357	40%	1143	218	81%
BOD_5 (mg/L)	217	208	4%	631	105	83%
TSS (mg/L)	214	42	80%	448	47	90%
VSS (mg/L)	208	27	87%	373	38	90%
SO_4^{-2} (mg/L)	332	116	65%	45	10	78%
COD/SO_4^{-2}	1.8	-	-	25.4	-	-

From the analysis of table 2, it can be seen that the Loulé WWTP prototype has high organic matter removal efficiencies, of 81% for COD and 83% for BOD_5, when compared with reference values that range between 65 and 75% (Chernicharo, 2007), which reveals a good performance of biological treatment. The Loulé WWTP has no saline intrusion, presenting a COD/SO_4^{-2} ratio of 25.4 (table 2) and high average COD values for a typically urban wastewater (1143 mg /L), data that are favorable for the anaerobic digestion process. For example, Hulshoff Pol *et al.* (2001) indicate that the COD/SO_4^{-2} ratio must be greater than 10 for the anaerobic wastewater treatment to be successful, which is in line with the results obtained at the Loulé WWTP.

At the Lagos WWTP, the prototype has lower organic matter removal efficiencies, only 40% for COD, justified by the fact that the WWTP has a significant contribution of undesiredsaline water inflows in the drainage network, with values conductivity in the influent to the prototype in the range of 2 to 30 dS/m (Figure 3), and average sulfate value of 332 mg/L. The COD/SO_4^{-2}

ratios (table 2) in the Lagos WWTP are low, mainly due to the significant presence of sulfates in the wastewater.

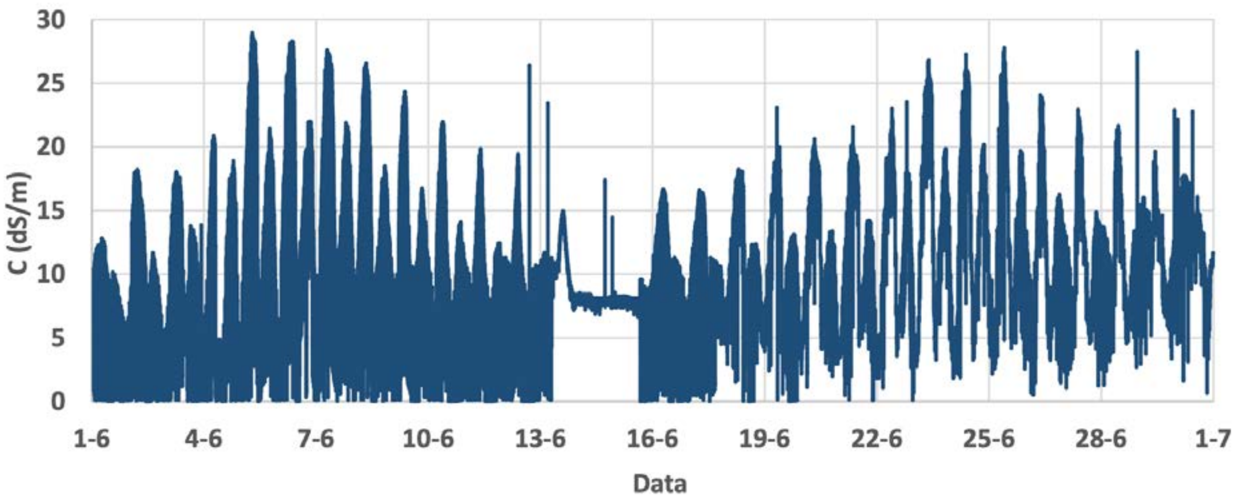

Figure 3. *Example of the monthly conductivity evolution of the PUSH® prototype in Lagos (June 2020)*

Table 3. *Average biogas composition in the reactor and after stripping the treated effluent in the Loulé and Lagos WWTP prototypes*

Average biogas composition	PUSH® Lagos with saline inflows		PUSH® Loulé with high organic loads	
	Reactor	Stripping	Reactor	Stripping
Methane CH_4 (%)	75	54	75	72
Carbon dioxide CO_2(%)	6	4	9	10
Oxygen O_2(%)	0	0	0	0
Nitrogen N_2 (%)	18	40	15	17
Hydrogen Sulfide H_2S (ppmv)	7428	7372	2251	2821

From the analysis of table 3, it is apparent that the WWTP in Lagos has high values of Hydrogen Sulfide in the biogas composition, which corroborates the fact that the WWTP has high concentrations of conductivity and sulphates in the influent wastewater. Operation data from the Lagos WWTP shows a lower biogas production when compared to the Loulé WWTP, conditioned by the operating mode of the prototype,given the high conductivity values. In the period from March to August, the wastewater supply to the prototype was programmed to be interruptedwhen the conductivity values are above 10 dS/m on the wastewater. In order to promote the functioning of the prototype in a more continuous way, this threshold was increased to 20 dS/m even in August, having been maintained until November. However, there was a need to adjust again to the maximum value of 10 dS/m, in view of the lower performance that the prototype presented with these conditions.

In terms of the yield of methane production, the PUSH® in Loulé has an average value of 0.21 m^3 CH_4/kg $COD_{eliminated}$, which is quite good considering that the theoretical value is 0.35 m^3 CH_4/kg $COD_{eliminated}$. In figure 4, it can be seen how the decrease in water temperature, by the onset of autumn, negatively affects yield. The Lagos PUSH® prototype showed much lower yields (in the order of 0.10 m^3 CH_4/kg $COD_{eliminated}$) due to the previously mentioned issues arising from saline intrusion. Furthermore, in the presence of sulphate there will be microbiologic activity of sulphate-reducing bacteria, which consume COD more efficiently than methanogenic bacteria, without forming methane and forming hydrogen sulphide (H_2S), which inhibits methanogenesis (Tchobanoglous et al., 2003).

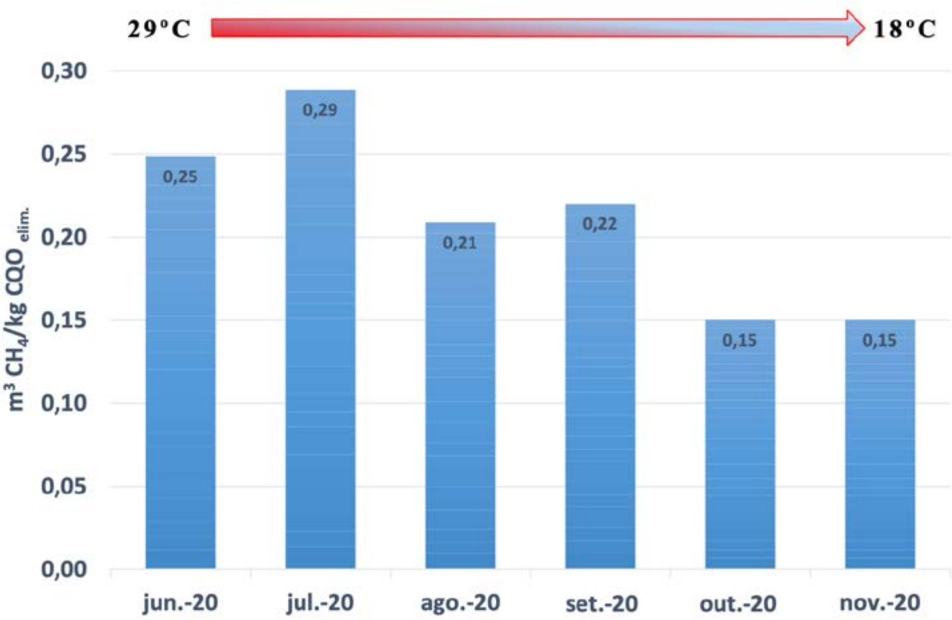

Figure 4. *Yield of methane production of the PUSH® prototype of Loulé*

CONCLUSIONS

The PUSH® prototype at the Lagos WWTP showed a lowerperformance thanthe Loulé WWTP, with organic matter removal efficiencies of 40% for COD and 4% for BOD_5,and a methane production yield of 0.10 m^3 CH_4/kg COD $_{eliminated}$. The poor results obtained in the Lagos WWTPis due to the high influentsalinity (conductivity in the range 2 - 30 dS/m), which affects the performance of the system, fundamentally reflecting the low COD/SO_4^{-2} ratio (1.8). Higher conductivity values cause instability in the anaerobic digestion process, namely inhibition of microbiological activity, difficulty in settling biomass and consequently the biomass washout. In order to minimize the impacts of the conductivity peaks, the feeding to the prototype was programmed to be interrupted in the presence of conductivity peaks.However, this measure alone was not enough to promote an improvement in its performance, taking into account long periods without feed.

The PUSH® prototype of Loulé showed high organic matter removal efficiencies, in the order of 81% for COD and 83% for BOD_5, with an average methane production yield of 0.21 m^3 CH_4/kg $COD_{eliminated}$. It is important to mention that, in addition to not having affluence of saline waters (conductivity in the range 0.5 – 2.7 dS/m and CQO/SO_4^{-2}ratio of 25.4), the Loulé WWTP has an influent generally with a high concentration of organic matter ($BOD_5$630 mg/L), conditions that favor the process of anaerobic digestion. This technology implemented in the liquid phase at the Loulé WWTP, after a screening system, proved to be quite attractive, considering that the installation has high organic loads.

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EVALUATION AND MATHEMATICAL MODELING OF A NEW FILTERING DEVICE TO IMPROVE THE EFFICIENCY OF REMOVAL OF CONTAMINANTS IN UASB REACTORS

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Abstract
Water scarcity is a problem that has been present for many years and water treatment plants do their best to remove their pollutants. Even so, these processes, mainly biological ones, should be improved by implementing alternatives that improve the removal of solids present in wastewater. In the search to improve the conventional treatment and its economy, this study presented: the physical and mathematical implementation of an Upflow Anaerobic Reactor (AR) coupled to two ceramic gravity filters (FGC) of different thickness (2 and 3cm), low real operating conditions, consequently, a bibliographic review of mathematical models of UASB reactors and ceramic filters simulated and calibrated in the WEST software was elaborated and finally, the statistical analysis was elaborated to compare the performance between the proposed FGC vs. The common alternative, settlers.

INTRODUCTION

Population growth, industrialization, agriculture, and urbanization have increased water consumption in recent years, thus generating a deficit of fresh water and indirectly an increase in the discharge of wastewater (Hasan M, 2011), causing wastewater treatment plants (WWTP) to be forced to improve their purification processes. Due to the characteristics of the existing treatment processes in Mexico, the implementation of an up-flow anaerobic reactor provided with two ceramic gravity filters was proposed, to improve the removal efficiency of organic pollutants and reduce the operating costs of conventional systems. In addition, modelling has been implemented on many occasions to know the dynamic behavior of process and the same way to be able to optimize and/or improve that processes, which would be an important application in wastewater treatment studies (WEF, 2013). However, there are not many studies that statistically represent the capacity that different types of filters must perform a similar operation of secondary settlers, which would provide a new avenue for the investigation of this type of study. Based on the aforementioned, this project implemented a AR coupled to a set of FGC seeking to improve the efficiency of solids removal along with the development of a mathematical model of its process and determining the suitability of the filters as an alternative to replace the secondary settlers.

METHODS

For the preparation of this study, 3 phases were defined, the first one was the developed startup of the RAFA-FGC in the Wastewater Treatment Plant in the North of Monterrey Metropolitan Area under real conditions. This system has the following physical characteristics:

Table 1. *Physical conditions of RAFA-FCG*

Unit	Height (cm)	Internal diameter (cm)	Diámeter external (cm)	Volume (L)
UASB	600	42.5	44.7	7.00
FCG 1	30.0	16.0	18.0	6.00
FCG 2	30.0	15.0	18.0	5.00

The filters (figure 1) were made of different materials such as by-products of the Copper smelting, annealed clay partition by-products, and Kentucky old mine clay # 4.

Figure 1. *Ceramic Filter*

On the other hand, within the operation of the RAFA-FGC, the wastewater enters the lower zone of the UASB-type anaerobic reactor, performing an ascending behavior (Hydrolysis) until it reaches its discharge point. The effluent from this reactor is sent and distributed through a series of pipes to the filtration system. At this moment, the water makes a downward movement because of gravity, until it reaches the ceramic filters. Subsequently, the treated water encounters each of the filters, which with the help of the hydraulic load will be filtered, thus obtaining an additional post-treatment to that carried out by the UASB reactor. The particles retained by the filters that little by little will obstruct its internal pores will be washed and returned to the pumping system by the lower discharges.

In the second phase, an exhaustive bibliographic review was carried out regarding the modeling studies of filters and UASB reactors to find the equations that could be used to determine the dynamic behavior of the proposed RAFA-FGC system. Concerning the filters, a system was applied that helps to improve their washing conditions, thus improving the operating process. The filters were washed by a reverse flow gravity system through two valves using a double outlet technique that regulates the flow, as necessary. It should be noted that both valves at the outlet of each filter never would be open.

After obtaining the analysis and results of the removal of pollutants, the performance was evaluated in comparison with the reports of average separation efficiency of secondary settlers installed downstream of a RAFA. For this study, clarifiers operating under similar working conditions to those of the filters understudy at the North Plant were taken as comparison examples; Based on the above, a statistical analysis was developed to compare the performance of the filtration system vs. sedimentation system, to determine if the filtration efficiency is statistically comparable to conventional sedimentation, which corresponded to the third phase of this investigation.

RESULTS AND DISCUSSION

According to the first phase of this project, the UASB Reactor coupled to the ceramic filters was put into operation. Next, Figure 2 shows the AR assembled in the North plant of the Monterrey metropolitan area, connecting to the distribution duct of the tributary to the treatment trains of the North WWTP, installed downstream of the settling.

Figure 2. *RAFA-FCG*

For the mathematical modelling of this process, an exhaustive bibliographic review of 37 articles was implemented. Of these, we tried to find those equations that were applicable in our project. Next, in Tables 2 and 3 some of the models found are presented.

Table 2. *Equations AR*

AR	
Author	**Equatión**
Andrews (1971)	$\frac{\partial S1}{\partial t} = \frac{F}{V} - \frac{F}{V}S1 + \frac{\mu}{\frac{YK}{S}}X1$
Tamayo (2005)	$\frac{\partial x}{\partial t} = \frac{q}{V}(Xo - Xe) + \frac{\mu maxS}{K+S}X - Kdx$ $\frac{\partial S}{\partial t} = \frac{q}{V}(So - Se) - \frac{\mu maxS}{Y(K+S)}X$
Enitan (2015)	$V\frac{\partial s}{\partial t} = QS_i - (\mu - K_d)V\frac{X_r}{Y} - QS_e$ $V\frac{\partial s}{\partial t} = QS_i - (\mu - K_d)V\frac{X_r}{Y} - QS_e$

Wardhani (2017)	$\frac{\partial S}{\partial t} = \frac{1}{Y}\frac{U_{max}xs}{K_s + S}$ $\frac{\partial P}{\partial t} = YP\frac{U_{max}xs}{K_s + S}$

Table 3. *Equations FGC*

FCG	
Author	**Equatión**
Mininni (1984)	$\frac{\partial v}{\partial dt} = \frac{P}{\mu\left(R\,C\,\frac{V}{A} + R_m\right)}$
Barre & Conway (2004)	$\frac{\partial P}{\partial L} = \frac{\mu v}{k}$
Varela (2014)	$\frac{\partial x}{\partial t} = \frac{Q^t}{A\varnothing}\frac{df^w}{ds^w}$

The above tables summarize the bibliographic review of some models developed for the anaerobic digestion of wastewater and filters. It should be noted that were considered only those investigations that show possible contributions to the present project and those most related to it were included. Now, considering the bibliographic reviews, it was possible to establish that the differential equations most closely related to the present study were Enitan et al. (2015) for the anaerobic reactor and Barre and Conway (2004) for the filter, due to the variables used since their equations manage to adequately describe the dynamic behavior of the biomass, the substrate, and the methane production for the reactor, as well as the filtration in the FGC. Besides, the input data to both models are easy to obtain, therefore they will be taken as the base for modelling this research.

CONCLUSIONS

From this study, it deduced that the assembly of the AR-FGC was carried out in the North plant of the metropolitan area of Monterrey together with the operation of this novel process.

Monod kinetics is the most appropriate to use due to the great affinity with the behavior of wastewater treatment including anaerobic processes.

Existing mathematical models were selected, and their equations were adapted to the conditions of the proposed RAFA-FGC system.

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DESCRIPTION AND OPERATION OF THE ANAEROBIC DIGESTION UNIT (ENERGY UNIT) IN THE WATER2RETURN PROTOTYPE

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Abstracts

The Water2REturn project involves the large-scale implementation of a waste treatment system for Matadero del Sur S.L. based on the Circular Economy model. In this context, the anaerobic digestion of the sludge generated in a previous aerobic stage and enzymatic pre-treatment with *Bacillus Liqueniformis* allows the generation of clean renewable energy as biogas as well as a final digest free of pathogens and susceptible to be used as a substrate for the growth of microalgae.

INTRODUCTION

In recent years, the population and industry have grown notably, causing a general decline in water quality. The meat industry is a growing subsector in our country that feeds mainly on short-cycle animals and intensive feeding. Andalusia occupies the third position of meat production by Spanish autonomous communities.

Wastewater from slaughterhouses, dairy, brewery, pharmaceutical industries, among others, can cause harmful effects on humans, flora and fauna (Aziz et al., 2019). Specifically, wastewater originating from slaughterhouses has characteristics that depend on various factors such as the size of the slaughter facility, the type of animal slaughtered, or the amount of water consumed in washing the equipment. In general, they have a high concentration of organic matter composed mostly of blood, skin and manure; suspended solids, oils and fats, mainly, in addition to the presence of heavy metals and antibiotics (Agabo-García et al., 2020). The direct discharge of these wastewater produces negative effects for the environment and human health, making it necessary to look for sustainable management alternatives to avoid these risks, mainly based on reuse and recycling. In this context, the Water2Return project involves the treatment of slaughterhouse water in an industrial prototype with the following objectives:

· to recover nutrients from wastewater from slaughterhouses.
· To obtain a final product suitable as a pathogen-free biofertilizer.
· To generate energy self-sufficiency of the plant and the slaughterhouse factory.
· To eliminate the waste destined for landfills.
· To decrease environmental pollution and the production of adverse effects in water bodies.

The place chosen to carry out this project is located in Seville, specifically in Salteras, in an industrial slaughterhouse called Matadero del Sur S.L.

The proposed treatment process involves the different public and private institutions participations that work together to optimize each operation involved in the biomass and energy revaluation process. The coordination of the project is carried out by the company BIOAZUL, which acts as an intermediary between the partners and the European Commission. This will also manage the innovation management group.

Within the entire process there are different stages carried out by the different groups:

· Wastewater treatment, carried out by BIOAZUL S.L. and CENTA (Center of New Technologies of Water).
· Fermentation of the sludge, whose managers are the research group of Technologies and application of enzymes of the University of Seville
· Anaerobic Digestion, which is in charge of the Department of Environmental Technologies of the University of Cádiz and ADVENTECH.
· Treatment with microalgae, managed by ALGEN.
· And finally, KIMITEC Group, the University of Seville and the University of Ljubljana, which are in charge of the physical-chemical and biological characterization of all the products.

Among the different treatments that make up the industrial prototype, this work focuses on the anaerobic digestion process. This process consists of the degradation of organic compounds with the help of various groups of microorganisms (archaea and bacteria) in the absence of oxygen and consists of 4 consecutive stages: hydrolysis, acidogenesis, acetogenesis and methanogenesis. Anaerobic digestion can present drawbacks such as a prolonged start-up period or the sensitivity of the process to temperature, but at the same time it has numerous advantages such as excellent removal of organic matter, lower production of sludge, low energy requirements and renewable energy production..

Another important advantage is the reduction of space that is achieved with an anaerobic digester to treat large quantities of wastewater compared to aerobic ponds that need a lot of surface, a treatment that is traditionally widely used in the treatment of wastewater in these industries. Thus, it is possible to value a by-product and obtain biogas that can self-supply the plant and the slaughterhouse in addition to the recovery of nutrients in the form of digest with agronomic properties. In addition, the CO_2 generated in the biogas will be used to feed the microalgae pool, reducing the emission of greenhouse gases.

This work describes the stage corresponding to the start-up and stabilization of the anaerobic process as it is the one with the greatest involvement for the University of Cádiz.

Materials and methods Substrates and inoculum

The substrates used in the feeding of the AnSBR are the following:

· Slaughterhouse waste water generated in the activity carried out in the Matadero del Sur S.L., that is, slaughter, cutting, washing of equipment ...

· Pre-treated sludge generated in the aerobic treatment of slaughterhouse wastewater.

· Fats removed in the pre-treatment of slaughterhouse wastewater.

Initially, the AnSBR digester was filled with a mixture of sludge and water from one of the treatment ponds from the slaughterhouse itself, with a content of 2.3 g / L of volatile solids.

Description of the treatment process

Within the Water2Return prototype, the University of Cádiz is working on optimizing the anaerobic digestion process. The central unit of the process is made up of AnSBR (Anaerobic Sequential Batch Reactor) technology. The system is also equipped with homogenization tank that allows the storage of: raw water from the slaughterhouse, aerobic sludge, effluent from the enzymatic aerobic sludge treatment process, as well as fats removed during degreasing. The biogas generated in the process goes through a cogeneration cell in which the CH_4 is transformed into H_2, while the CO_2 will serve to feed the microalgae pool. Finally, the clarified, nutrient- rich effluent will serve as a culture medium for the microalgae.

Description of the homogenization tank and AnSBR

The anaerobic system has a homogenization tank and an AnSBR digester. The homogenization tank has a 15 m^3 capacity and 3 meters in height, whose utility is to store the waste generated in the slaughterhouse. It has a stirring system, pH control, level probe and sampling. Connected to this tank there is an anaerobic digester with a capacity of 30 m^3 and 6 meters high, equipped with a stirring system, level probe, temperature control, pressure probe, sludge outlet and sampling. It has a gas outlet connected to an energy cogeneration system to obtain clean electricity to supply the plant. The effluent outlet is connected to an algae plantation in order to obtain biomass. A flow meter is located between the two tanks to control the feed flow.

Operating conditions

The AnSBR process involves 4 operational stages. The established agitation was 30 minutes / hour for 20 hours, leaving 4 hours without agitation for settling and evacuation of the effluent. In the connection from the homogenization tank to the anaerobic digester there is a flow meter to control the amount of feed transferred. And each tank has a level probe that gives us a reading of the volume of waste inside. It also has a boiler that is responsible for keeping the digester in mesophilic conditions, around 35 ° C, as well as a pH control system.

Initially, a hydraulic retention time (HRT) of 20 days was set, which implies a daily feed equivalent to $1m^3$ and, therefore, a discharge of 1 m^3 of effluent.

The entire anaerobic process is programmed to operate automatically through a SCADA program. Through the computer program we have access to control the operating conditions of our unit remotely, as well as all the control parameters that are included in the plant, allowing it to operate automatically.

Analytical methods

First, an initial characterization of the substrate used was made, measuring parameters such as total chemical oxygen demand (TCOD), soluble oxygen chemical demand (TCOD), total solids (TS), volatile solids (VS), organic matter (O.M.) , carbon / nitrogen ratio (C/N ratio) and volatile fatty acids (VFA) content, according to standard methods (Standard methods, 2012).

RESULTS AND DISCUSSION

The experimental results of operation in the initial stage of feeding to the AnSBR reactor are shown below. As can be seen, the initial characteristics of the substrates vary in each sampling since they depend on the daily activity carried out in the slaughterhouse, as shown in Table 1.

Table 1. *Initial characterization of the AnSBR.*

Feed	pH	TS(g/L)	VS(g/L)	TCOD(mg/L)	SCOD(mg/L)
23-sep	7,98	2,19	0,47	1012,93	752,46
01-oct	6,83	5,15	2,33	3762,33	2556,45
22-oct	6,74	2,28	0,93	9087,47	4876,56
28-oct	6,95	2,64	1,10	2749,39	1939,04

The experimental data of characterization of the feed and effluent of the AnSBR digester during these first weeks show an efficiency of purification of organic matter, measured as COD, of 57% TCOD and 44% of SCOD, with expectations that it increases rapidly. Total solids (TS) and volatile solids (VS) are progressively decreasing, obtaining an increasing purification efficiency as can be seen in figure 1.

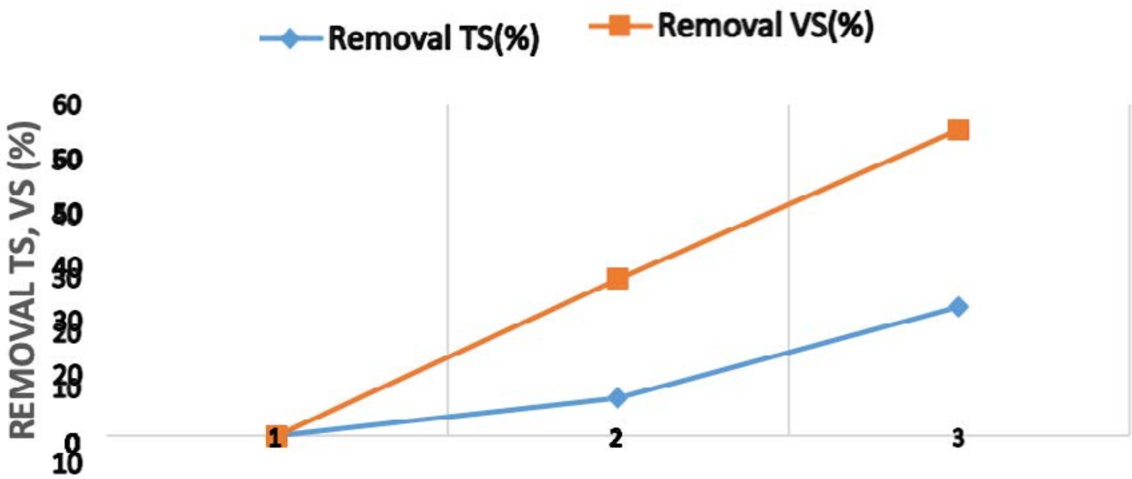

Figure 1. *Purifying efficiency in the elimination of TS.*

The pH measured in the feed varies according to the discharge received with values between 6.7 and 8, however, the pH measured in the effluent has values very close to 7.

In relation to the VFA, the experimental data show an increase in the concentration of acetic acid in the effluent. Propionic acid is removed by approximately 50% while butyric acid disappears completely in the effluent. The other VFAs levels decrease in the effluent or even disappear completely.

CONCLUSIONS

After the first stages of optimization of the anaerobic digestion process of the WATER 2RETURN Project, the industrial AnSBR digester shows a stable behaviour with high purification yields and a stable pH value. Percentages of removal of TS, TCOD and total VFA are reached above 50%. As a result of these results, it is concluded that the effluent generated in the anaerobic process is suitable as a culture medium for the microalgae reactor.

ACKNOWLEDGEMENTS

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OBTAINING FERTILIZERS FROM ANAEROBIC CODIGESTION OF SEWAGE SLUDGE AND WINE VINASSE

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Abstract
The pathogen content in the final digestatesof anaerobic codigestion of sewage sludge and wine vinasse was studied in three different conditions: mesophilic temperature, thermophilic temperature and temperature-phase system (thermophilic-mesophilic). All three digestates obtainedhave lower limits than those required by European legislation to be used as organic fertilizer. In addition, the digestate of the temperature phase system and the thermophilic reactor could also be classified as class A biosolid, as it complies with American regulations.

INTRODUCTION

Excess sludge is the main problemfor wastewater treatment plants (WWTPs), since its management implies more than 50% of operating costs, being this problem more serious in small urban agglomerations. Anaerobic digestion (AD) is the most widely used technology to reduce the volume of sludge and, in addition, to generate energy as methane gas.

On the other hand, the large wine production that takes place in Spain is related to a high amount of waste in the wine sector. The vinasse generated in the wine distillation process is one of these wastes. Although AD is a process that has been studied on numerous times for the production of methane from wine vinasse(Jiménez et al., 2006; Pérez-García et al., 2005)performance of two high rate technologies, upflow anaerobic fixed-film reactor and fluidized bed laboratory-scale, treating distillery wastewater (wine vinasses, the implementation of a specific treatment for the purification of wine vinasse supposes high economic costs for wineries. Moreover, wine vinasse is a seasonal waste, so the codigestion with sludge in WWTPs is an alternative to its management. (Tena et al., 2019).

Mesophilic AD is the most commonly applied operating regime at the European and Spanish level. Although, currently,the interest in thermophilic AD has increased due to several advantages such as the improvement biogas productivity, the enhancement of solids removal efficiency and inactivation of pathogens(Chuenchart et al., 2020)the process stability was affected at higher loading. Whereas, the reduction in FOS/TAC value (the ratio of volatile fatty acids to total alkalinity. However, higher energy input as well as poor stability and reliability suppose some of the reasons why thermophilic AD is not widely used(Labatut et al., 2014). Temperature Phased Anaerobic Digestion (TPAD) emerged in order to take advantages of both process and reduce their drawbacks.TPAD process combines both thermophilic and mesophilic processes in a sequential process, allowing the advantages of both process to be used(Han et al., 1997)volatile solids (VS.The use of this technology for sewage sludge treatment has been commonly used due to thermophilic phase accelerates the limiting stage of AD and also achieves the sterilization of the waste of pathogenic organisms. This make possible the use of the obtained digestate for agronomic purposes(Riau et al., 2010)such as temperature-phased anaerobic digestion (TPAD. In addition, TPAD also allows to improvethe sludge dewaterability characteristics(Bivins and Novak, 2011).

Digestate is the result of the degraded biomass after biogas production, being stable organic matter rich in various nutrients (N, P, K). The composition and quality of the digestate is highly dependent on the composition and quality of the feedstock used(Al Seadi et al., 2013)not only rich in plant nutrients, but also unpolluted by undesirable matter and compounds of physical, chemical or biological nature. The most important premise of producing high-quality digestate is utilization of high-quality feedstock for the digestion process. Measures for quality management of digestate are implemented in a number of countries as part of national environmental, waste or agricultural legislations. More recently, digestate quality assurance systems are also increasingly used. The overall aim is to secure the production of high-quality digestate and to enhance its subsequent use for agricultural purposes. Digestate can be utilized as it is produced, or it can be further refined through various treatments and technologies commonly known as digestate processing. By digestate processing, marketable biofertilizers can be produced or the nutrient load of the remaining effluent can be decisively reduced, up to discharge quality. A common technology is solid-liquid separation, using screw press separators and decanter centrifuges. Further treatments and technologies can be applied for stabilization of the solid fraction or further refining of the liquid. This chapter emphasizes the main issues related to the quality management of digestate use as a fertilizer, with references to the regulatory framework. The possibilities of improving digestate quality, transportability and marketability through digestate processing are also emphasized. A brief overview of other possible utilizations of digestate and digestate fractions (side streams in digestate processing. The term "biosolids" is used when the digestate comes from wastewater treatment("Boletín agrario," n.d.). Biosolids are divided into two classes according to their pathogen content: Class A and Class B(Gerba and Pepper, 2009). Class A biosolids are treated to reduce the level of pathogens below detectable levels and can be used without pathogen-related restrictions at the application site. The United State Environmental Protection Agency (USEPA) sets a density for Class A biosolids quality of less than 1000 faecal/g total solids and for *Salmonella*below 3 Most Probable Number/4 g total solids. Furthermore, current European legislation (Regulation (UE) 2019/1009 of the European Parliament and the Council of 5 June 2019) stipulates that *Escherichia coli* (E.coli) density must not exceed the limit of 1000 colony forming units (CFU)/g total solids and that *Salmonella*must be absent in 25 mL of samples in an organic fertilizer.

In this study, final effluents from a temperature-phase system (thermophilic-mesophilic) and both single-stage reactors operating under mesophilic and thermophilic conditions were analysed to evaluate their pathogen content.

METHODOLOGY

The sewage sludge was collected at the Guadalete WWTP and the wine vinasse was supplied by Bodegas González Byass, both located inJerez de la Frontera, Cadiz.

The reactors used have a working volume of 2 liters. The reactor head has several openings that are used for feed inlet, effluent outlet, biogas outlet and temperature sensor. The reactors are placed on top of a heating plate that maintains the desired temperature. The system has a stirring device.TPAD system consists of two reactors in series, the first operating under thermophilic conditions and the second under mesophilic conditions.

The determination of total coliforms, E. coli and*Salmonella*in the digestate of the three systems studied was carried out according to Standard Methods("Standard methods" 2012).

RESULTS AND DISCUSSION

The pathogen content of the mixture of sewage sludge and wine vinasse fed to the reactors is shown in Table 1. As can be seen, it does not comply with the legal requirements imposedby European legislation or by the USEPA to be classified as class A biosolids.

Table 1. *Pathogen content of the mixture sewage sludge and wine vinasse fed to the reactors.*

	Total Coliforms	E. coli	*Salmonella*
CFU/g TS	14442	6983	10000

After anaerobic codigestion process, neither in the TPAD system nor in both the single-stage reactors were *Salmonella* colonies detected.

Regarding coliforms, the results obtained are shown in Figure 1. In the digestate of the thermophilic single-stage reactor, no coliforms were detected, so they are not represented in Figure 1. In relation to the other two systems, the density of E. coli in the digestate decreased 89 % and 96 % with respect to the initial concentration for the mesophilic single-stage reactor and for the TPAD system, respectively, being below the limit imposed by European legislation (1000 CFU/g TS).Although the density of fecal coliforms has not been determined, in the TPAD system it can be assured that this density is below the limit imposed by USEPA (1000 CFU/g TS)because the density of total coliforms in the digestate of this process is 1014 CFU/g TS.In the case of the digestate obtained from the mesophilic single-stage reactor this fact cannot be assured because the density of total coliforms is higher than1000 CFU/g TS, as can be seen in Figure 1. Several authors(Lloret et al., 2013; Riau et al., 2010; Rubio-Loza and Noyola, 2010)such as temperature--phased anaerobic digestion (TPADconcluded that TPAD system can be used for the production of Class A biosolids because the density of *Salmonella* and E.coli can reach values below the legal limits.

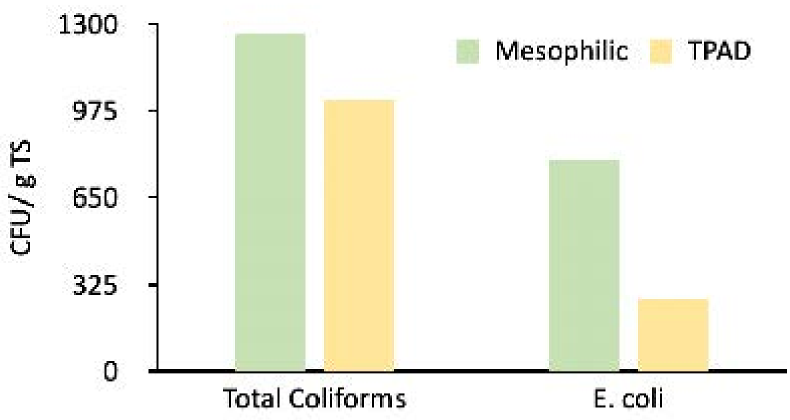

Figure 1. *Coliform densities in the digestate of each system tested.*

CONCLUSIONS

From the results obtained, it can be concluded that the final product obtained in the thermophilic single-stage reactor and in the TPAD system have safe levels of pathogens and could be used as organic fertilizer according to European legislation, replacing chemical fertilizers that cause a great environmental impact. These effluents could also be classified as Class A biosolids according to USEPA.However, the digestate obtained from the single-stage mesophilic reactor, due its microbiological characteristics, it can only be used as organic fertilizer according to European legislation.

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03

EMERGING POLLUTANTS

PRESENCE OF PHARMACEUTICAL RESIDUES IN A NATURAL WASTEWATER TREATMENT SYSTEM AND EVALUATION OF THEIR REMOVALEFFICIENCY

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Abstract
The use of reclaimed waters is a great strategy of adaptation to climate change but it is necessary to ensure the quality of this resource. In this regard, reclaimed waters with a good quality does not produce harmful effects when reach the environment after being reused in different activities. However, emerging pollutants such as pharmaceuticals could affect reclaimed water quality because these compounds have noticeable toxic effects over ecosystems, especially aquatic ones. It is known that conventional wastewater treatments are moderately effective to eliminate this type of pollutants from sewage, but the researches about the efficiency of natural treatments are scarcer. In this study, the presence of 11 pharmaceutical compounds has been assessed in a natural wastewater treatment plant of Gran Canaria (Spain). The sampling was performed monthly for two years.The concentrations of the detected compounds ranged from ng·L^{-1} to mg·L^{-1} and the removal efficiencies were moderately high for most compounds such as stimulants or some anti-inflammatories.

INTRODUCTION

The use of reclaimed waters in agricultural or recreational activitiesis an important alternative and a strategy to adapt to climate change in arid and semi-arid zones. Nevertheless, the use of reclaimed waters in these activities could be controversial because the possibility that emerging pollutants as pharmaceutical residues could enter in the food chain (Martinez-Alcalá *et al*, 2018). In this regard, it is important to highlight that wastewaters are the main input of pharmaceuticals into the environment and for this reason it is necessary to ensure that reclaimed waters present low concentrations of these compounds that do not produce deleterious effects on the environment and farmlands.The main problem about pharmaceutical residues is that current legislation does not establish concentration thresholds for these pollutants in water and, subsequently, the treatments of wastewater treatment facilities are not designed to eliminate them. This is especially important in natural wastewater treatment facilities, because the wastewater purifying treatments are based on natural processes and pharmaceutical residues could affect them. The elimination of pharmaceutical compounds from wastewaters is significantly complex due to the variety of these analytes, their properties and the possible conjugated compounds formed during metabolization (Gurke *et al*, 2015).

In the last 15 years, many scientific studies have been focused in the determination of pharmaceuticals in conventional wastewater treatment systems (Wang *et al*, 2015), but the evaluation of natural treatment systems (NWTS) is scarcer because these systems are not as common as conventional ones. Nevertheless, the effects of pharmaceutical residues of wastewaters from NWTS could produce a greater environmental impact because these facilities are usually located in small, isolated or protected areas where conventional systems could not be installed.

In this study, eleven different compoundsbelonging to different pharmaceutical families (anti-inflammatories, stimulants, lipid regulators, antihypertensives, anticonvulsants, and antibiotics) have been determined in the different purifying stages of a NWTS of Gran Canaria island (Spain) in order to know the removal efficiency of the different natural wastewater treatments. Because of the expected low concentrations of the compounds, solid phase extraction (SPE) has been used as extraction and preconcentration technique. Then, ultra-high performance liquid chromatography tandem mass spectrometry (UHPLC–MS/MS) has been used as detection and determination technique.

METHODS

The samples were taken monthly during two yearsafter the different wastewater treatmentsof a natural wastewater treatment plant from a rural zone of Gran Canaria Island (Spain).This facility is based in two constructed wetlands (CWs), the first one is a vertical flow wetland and the second one a planted sub-superficial horizontal flow wetland. This WWTP was designed to treat the wastewater of 500 inhabitant equivalents but nowadays it treats a higher volume of wastewater with great results. Samples were taken in the influent (point C1) and after each process of the treatment: Imhoff tank (point C2), vertical flow wetland (point C3) and horizontal subsuperficial flow wetland (point C4). (Figure 1)

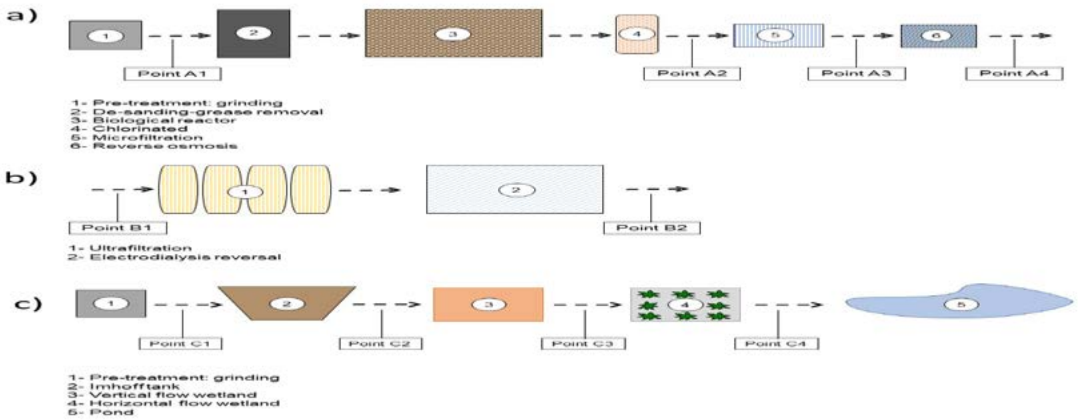


Figure 1. *Layout of the surveyed NWTS. Figure adapted from (Afonso-Olivares et al, 2017)*

To extract and preconcentrate the target pharmaceuticals in wastewater samples, a methodology based on SPE previously optimized (Afonso-Olivares *et al*, 2017)was used.After the extraction, ultra-high performance liquid chromatography tandem mass spectrometry (UHPLC–MS/MS) was used as separation and detection technique.This analytical methodology present great recoveries, between 52.9 and 123.6% and appropriate detection limits, between 15.3 $ng \cdot L^{-1}$ and 13.3 $\mu g \cdot L^{-1}$. The method showed also a great linearity, with correlation coefficients (r^2) over 0.99 in all cases and a good intra–day and inter–day repeatability, with relative standard deviation (RSD) values below 22%

Results and discussion

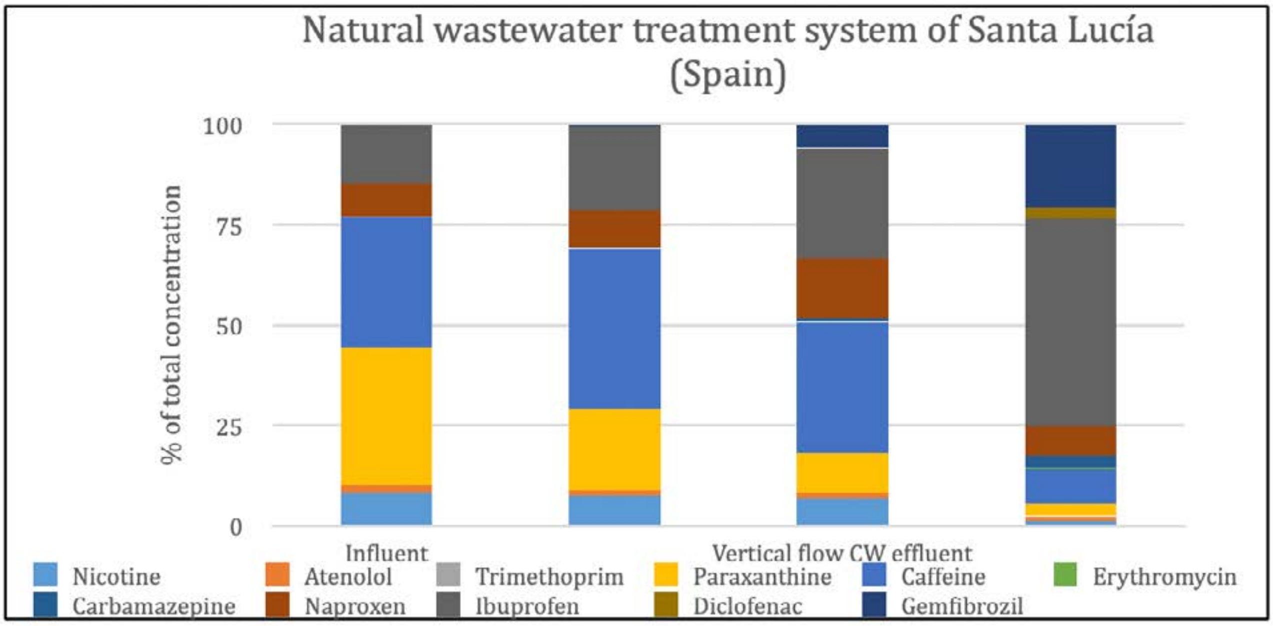
The eleven pharmaceuticals were detected in all sampling points during the monitoring campaign. Antibiotic erythromycin was the compound less detected (in less than 33% of the samples) while stimulants such as caffeine and nicotine or anti-inflammatories such as ibuprofen or naproxen were detected in more than 80% of the studied samples

Table 1. *Median concentrations* (μg·L^{-1}) *and percentage of positive analysis (in brackets) for target pharmaceuticals*

	Influent	Imhoff tank	Vertical flow wetland	Horizontal flow wetland
Nicotine	9.606(89.5)	7.651 (89.5)	3.515 (88.9)	0.245 (81.8)
Atenolol	2.195(84.2)	1.402 (89.5)	0.742 (83.3)	0.198 (81.8)
Trimethoprim	0.034(63.2)	0.031 (68.4)	0.020 (50.0)	0.018 (36.4)
Paraxanthine	40.050(89.5)	21.243 (89.5)	5.084 (88.9)	0.596 (81.8)
Caffeine	37.888(89.5)	41.497 (89.5)	16.771 (88.9)	1.633 (81.8)
Erythromycin	0.101(10.5)	0.066 (10.5)	0.081 (33.3)	0.064 (18.2)
Carbamazepine	0.299(89.5)	0.306 (89.5)	0.432 (88.9)	0.604 (81.8)
Naproxen	9.653(84.2)	9.383 (84.2)	7.713 (83.3)	1.397 (81.8)
Ibuprofen	16.942(84.2)	21.886 (84.2)	14.209 (83.3)	9.803 (81.8)
Diclofenac	0.035(21.1)	0.028 (47.4)	0.139 (61.1)	0.525 (81.8)
Gemfibrozil	0.222(73.7)	0.647 (78.9)	2.904 (83.3)	3.972 (81.8)

As can be seen in table 1, the highest concentrations of target pharmaceuticals were from caffeine, paraxanthine, nicotine, ibuprofen and naproxen. These compounds presented median concentrations in the influent point that ranged from 9.60 to 40.05 μg·L^{-1} while the rest of target compounds were in the range of ng·L^{-1}. The highest concentrations in a sampling were for naproxen and caffeine, both in an influent sample. The concentrations of the target pharmaceuticals decreased as the purification process was performed. This can be observed in the total median concentrations of the NWTS that were of 117.02 μg·L^{-1} in the influent, 104.14 μg·L^{-1} after Imhoff treatment and 51.61 and 19.06 μg·L^{-1} after vertical–flow and horizontal–flowwetlands, respectively.Figure 2 shows the distribution of pharmaceuticals in the sampling points and it can be observed thatthe majority of detected compounds are stimulants and anti-inflammatories. Also, it can be observed that the contribution of stimulants is decreasing as the natural treatment processes are performed, while the contribution of some anti-inflammatories, such as ibuprofen, increases.

Figure 2. *Distribution of pharmaceutical residues in the different points of the studied NWTS*



Regarding the removal efficiency of the different treatments, the Imhoff process showed a limited removal efficiency for pharmaceutical residues. For constructed wetlands (vertical and horizontal flows), they showed similar behaviors. In the vertical flow system, removals over 60% were achieved for the three stimulants, nicotine, caffeine and paraxanthine. Atenolol showed a medium removal of 51.4% and poor elimination efficiencies (between 10.6 and 33.1 %) were obtained for trimethoprim, and the three anti-inflammatory compounds. Carbamazepine showed a recalcitrant behavior and subsequently, low or even negative removal efficiencies, which coincide with similar studies (Hai *et al*, 2018). The horizontal flow treatment provided slightly higher removal efficiencies than vertical flow treatment. In this sense, five compounds presented median eliminations over 75% (nicotine, caffeine, paraxanthine, atenolol and naproxen). For trimethoprim and ibuprofen, the removals were low (30.0 and 26.4% respectively), but higher than those obtained in the vertical flow systems. Overall, the natural treatment processes performed in this NWTS showed great eliminations for stimulants (over 97.5%), atenolol (90.9%), naproxen (79.4%) and trimethoprim (64.0%). Three compounds (carbamazepine, diclofenac and gemfibrozil) showed negative removals after the whole purification system and this trend coincide with a previous study performed in this NWTS(Afonso-Olivares *et al*, 2017).

CONCLUSIONS

The 2-year monitoring performed in this study have permitted to evaluate the presence of pharmaceuticals in a NWTS as well as the removal efficiency of pharmaceutical residues using natural treatments. Due to the singular characteristics of each family of target pharmaceutical compounds, their removal rates were very variable but satisfactory removal efficiencies were obtained for different compounds such as stimulants as nicotine or caffeine and other drugs as atenolol and naproxen. For other compounds as ibuprofen, diclofenac or gemfibrozil, natural treatments were not so effective as conventional ones.Since all the target pharmaceutical compounds are still present in the studied treated water after using natural systems (with concentration levels from $ng \cdot L^{-1}$ to $\mu g \cdot L^{-1}$), further studies are demanded in order to improve the purification systems. Special interest must be paid in some recalcitrant compounds as carbamazepine, for which very low removal rates were achieved.In light of the above, it could be concluded that the studiednatural wastewater treatment system is effective in the elimination of some pharmaceutical residues and the purified waters present a lower environmental risk linked to these compounds than raw wastewaters.

ACKNOWLEDGMENTS

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CHARACTERIZATION OF THE MAIN EMERGING POLLUTANTS IN WWTP AFFLUENTS OF SMALL RURAL VILLAGESAND MONITORING OF THEIR REMOVAL PERFORMANCE

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Abstract

In this work, emerging pollutant removalperformances of Wastewater Treatment Plants of medium and small villages in the south of Extremadura have been analyzed. All of them are geographically located in areas with significant agricultural and livestock activity. A total of 13 pollutants belonging to the groups of analgesics, antibiotics, herbicides and antiparasitics were analyzed. The results showed high elimination rates (greater than 90%) in analgesics in all the WWTPs studied. In all other substances analyzed (herbicides, antiparasitics and antibiotics) removal efficiencies around 70% were observed.

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INTRODUCTION

Thanks to the advance of analytical detection techniques, it has been possible nowadays to detect and alert of the presence of new pollutants present in water masses at trace concentrations. These compounds are called Emerging Contaminants, which comprise a wide range of substances of different origins and chemical nature, pharmaceutical products, personal care products, surfactants, plasticizers and industrial additives, which are present in water in trace concentrations.

The increase in the world population and,along with it, the current overexploitation of water resources has led, in recent years, to an increase in the demand for water for domestic, agricultural and industrial use, giving rise to an increase in the presence of Emerging Contaminants (EC) in the aquatic environment (Argarwal, 2005). However, their low concentration and diversity make their detection and elimination difficult (Gorito et al, 2017; Coquery et al., 2005). With the aim of improving the quality of the aquatic environment, Executive Decision 2455/2001/CE was published in 2001,in which the first list of 33substances dangerous for the aquatic environment, even in trace concentrations, appears. All of them were identified as Emerging Contaminants. Therefore, new technologies need be developed and implemented in Wastewater Treatment Plants (WWTP) to deal with those new pollutants in order to improve the quality of the water bodies. Thus, it can be said that research on detection and removal of EC in waste waters are crucial for a good management of water resources. In this work a study of the EC removal efficiency in small and medium towns located in the south of Extremadura is carried out.

METHODS

The main source of EC in the aquatic environment is wastewater, which carries man-made pollution along with that from agriculture and livestock as a result of runoff or infiltration processes that end up in the treatment system or into surface and underground waterbodies, thus entering into the water cycle. This study focuses on the CE removal efficiency in WWTPs in small and medium rural towns. Their economic activitiesare based on agriculture and livestock, which generates a strong environmental impact ofEC. Specifically, seven villages located at the south of Extremadura were chosen. They were selected with different population densities, in order to analyze the presence of ECrelated to their population density. The villages chosen were: Monesterio and Fuente de Cantos with 4500 Equivalent Inhabitants (EI), Talarrubiaswith 3500 EI, Zahinos with 2700 EI, La Albuera with 2000 EI, Cheleswith 1200 EI and Valencia de las Torres with 500 EI.

The WWTPsstudied were monitorizedthroughout six months, in autumn and winter seasons. A total of 13 emerging pollutants belonging to the groups of analgesics, antibiotics, herbicides and antiparasitics were analyzed. All these pollutants were chosen taking into account the economic activities present in these villages. The analytical technique used to detectthe EC was liquid chromatography for separationalong with mass spectrometry for detection, identification and quantification of organic compounds (Hoffmann et al., 1999; Di Corcia et al., 2002).

In order to analyze the efficiency of the treatment plants in removingEC, the pollutant removing rates during the entire study period were calculated according to the following equation:

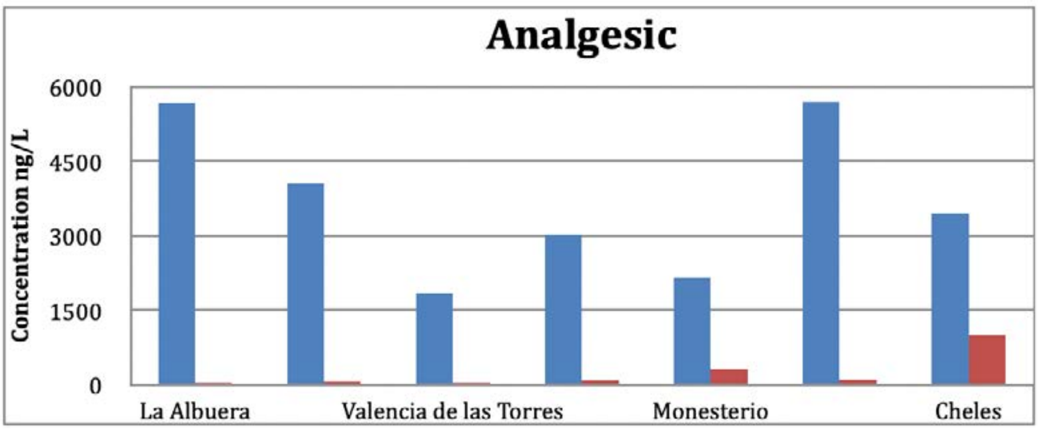
$$R(\%) = \frac{C_e - C_s}{C_e} \cdot 100$$

where C_e is the concentration of each EC at the plant input and C_s that at the corresponding output.

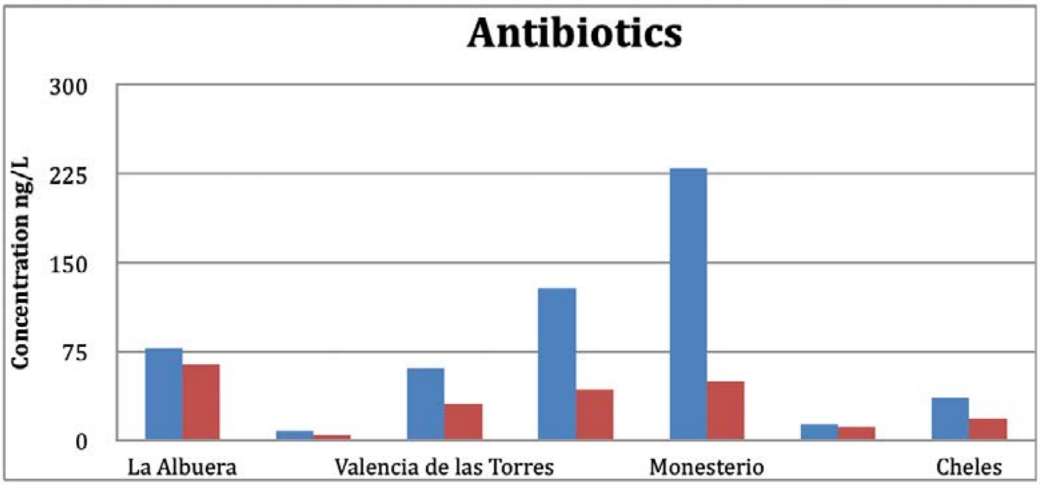
RESULTS AND DISCUSSION

Figure 1 shows mean concentrations at the inputs and outputsof all WWTPs studied for each group of EC analyzed (herbicides, antibiotics, analgesics and antiparasitics) throughout the entire study period. As can be seen, the highest concentrations at the inputs appear in analgesics (Figure 1a). It is noteworthy that villages with lower population density have higheranalgesicconcentrations. High elimination rates are observed in every village, reaching rates above 90% in La Albuera, Zahinos, Valencia de las Torres, Fuente de Cantos and Talarrubias (see Table 1). However, Chelesbarely exceeds 70%. The villages with the highest antibiotic removing rates are Monesterio (79%) and Fuente de Cantos (67%). However, no of the villages studied exceeds 79% inantibiotic removing rates. Talarrubias is the town with the lowest rate, which is lower than 15%. It is worth noting that no of the villages studied show high concentrations of herbicides at heir WWTP input. This could be because this study was carried out during autumn and winter months, and during these seasons the use of herbicides is very low. Talarrubias is the village that shows the highest efficiency in herbicides removal (68%). Finally, regardingantiparasites, no village showremoval rateshigher than 57%, being Valencia de las Torres the one with the highest rate.

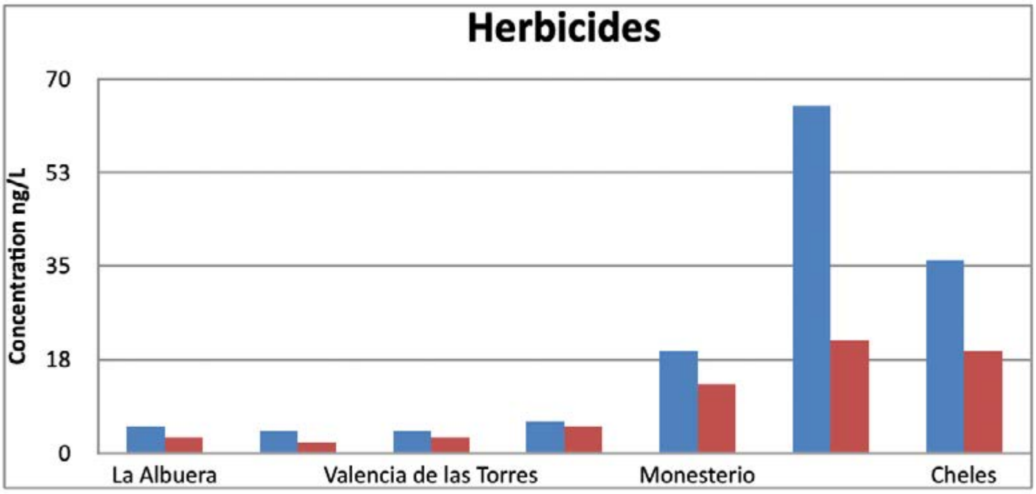
a)



b)


Antibiotics
Concentration ng/L
300
225
150
75
0
La Albuera
Valencia de las Torres
Monesterio
Cheles

c)


Herbicides
Concentration ng/L
70
53
35
18
0
La Albuera
Valencia de las Torres
Monesterio
Cheles

d)

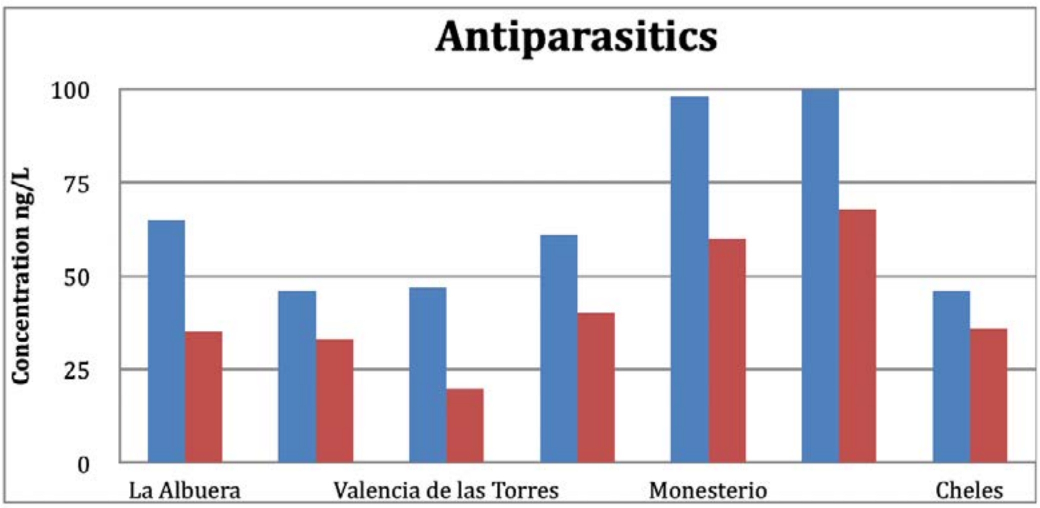

Antiparasitics
Concentration ng/L
100
75
50
25
0
La Albuera
Valencia de las Torres
Monesterio
Cheles

Figure 1. *Emerging Contaminant concentrations at plant inputs and outputs.a) Analgesics, b) Antibiotics, c)Herbicides, d)Antiparasitics.*

Tabla 1. *Removing rates for each group of analyzed substance in the villages studied.*

	La Albuera	Zahinos	Valencia de las Torres	Fuente de Cantos	Monesterio	Talarrubias	Cheles
Analgesics	99%	98%	98%	97%	86%	98%	71%
Antibiotics	18%	50%	49%	67%	78%	14%	47%
Herbicides	40%	50%	25%	17%	32%	68%	47%
Antiparasi-tics	46%	28%	57%	34%	39%	32%	21%

CONCLUSIONS

In order to address the problem of Emerging Contaminants in southern Extremadura and study their adverse effects on the aquatic environment, this work has analyzed the removing rates of several EC in the Waste Water Treatment Plants of seven villages located atthe south of Extremadura, whose main economic activitiesare agriculture and livestock. 13 ECs belonging to the groups of analgesics, analgesics, herbicides and antiparasitics were analyzed. The results showed that the highest concentrations at the WWTPinputs were observed in analgesics, which in turn were the most efficiently removed in every village studied, so that rates higher than 95 % were measured in most of them. It is worth noting that villages with lower population density have high concentrations of analgesics. This may be due to the fact that all of them haveageing populations. On the other hand, it can be seen that for the rest of the EC studied (herbicides, antiparasitics and antibiotics) some villages show removal rates around 60%, with Monesterio being the one with the highest antibiotic removal rates (78%). Finally, Talarrubias and Valencia de las Torres presented the highest herbicides and antiparasiticsremoval rates.

It is worth noting thatherbicide concentration is not high in all the WWTP inputs studied. This could be due to the fact that this study was carried out during autumn and winter months, when the use of herbicides is very limited. On the other hand, it is also worth noting that those villages in which there are industrial slaughterhouses as the main economic activity, high concentrations of antiparasitics and antibiotics were detected at WWTPinputs. This fact could be due to their presence in animals that are processed in those slaughterhouses.

Finally, it can be said that, currently, WWTPs located in the south of Extremaduraprovide high performances in the elimination of certain EC, especially analgesics, although the removal ratesof antibiotics, herbicides and antiparasitics should be further improved. Therefore, it could be necessary to implement new technologies in conventional WWTPs to improve the removal performance ofEC.

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MICROPLASTICS IN SMALL WASTEWATER TREATMENT PLANTS: A CASE OF STUDY IN SIERRA DE CÁDIZ (SPAIN)

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Abstract

The presence of microplastics in WWTPs with less than 10,000 population equivalents is also a problem, as these WWTPs also have less conventional technologies that do not remove these pollutants. This study presents an industrial WWTP and an urban WWTP for small populations where microplastics are observed in n/L amounts as well as different types of microplastics.

INTRODUCTION

Eliminating microplastics in the environment is the subject of current research. The pollution of microplastics to the environment is an increasingly serious problem, and it is expected that the problem will continue for hundreds of years (Wu et al., 2016). Larger plastics in turn degrade into particles. Microplastics (MPs) are plastic particles smaller than 5 mm. These microplastics are creating a global environmental problem that not only affects the environment itself, but

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especially the food chain. They represent risks that all living things should consider. The serious consequences for food safety have been proven and must be considered (Ajith et al., 2020).

One of the most important inputs of these compounds into the environment are waste water treatment plants (WWTP) (Franco et al., 2021). Current technologies are not capable of removing microplastics. Recent studies show that they can remove 50 to 90% of the water line, discharging microplastics in the sludge line. Considering that small populations of waste water treatment plants tend to have conventional technologies that are even less effective for these pollutants, these populations should be taken into account in the studies.

METHODS

Wastewater Treatment Plants

El Bosque

This treatment plant belongs to the *Mancomunidad de Municipios de La Sierra de Cádiz*, is located in the municipality of El Bosque and the water it treats comes from the industrial estate in the area, including industries such as: furniture manufacture, manufacture and marketing of all types of meat and dairy products, industrial laundry, processing and marketing of potatoes, aluminium and glass, leather goods, carpentry, manufacture of composites, cutting and moulding of marble, among others. The WWTP has been designed to treat industrial wastewater of 500 inhab-eq, generated in the *Huerto Blanquillo* industrial estate. Taking into account the discharge authorisation of the *Junta de Andalucía* (2010), a brief description of the water treatment line is given below:

– Pre-treatment: it has a roughing through an automatic grate for the retention of fine solids, with a passage of 12 mm.
– Biological treatment: this is an activated sludge process with aeration, the biological reactor has an automatically operating aeration turbine. In addition, the coagulation-flocculation process is carried out at the reactor outlet (coagulant: aluminium sulphate and polyelectrolyte flocculation). The secondary decantation is of the pyramidal trunk type, and there is also a sludge recirculation line from the decanter to the biological reactor.
– Disinfection: it has a contact chlorination chamber, which is only used in the event that exceedances are detected.

Prado del Rey

This treatment plant belongs to the *Mancomunidad de Municipios de La Sierra de Cádiz* and is located in the municipality of Prado del Rey. This facility treats water of urban origin corresponding to 6002 inhab-eq, generated in the urban nucleus of the municipality. Taking into account the discharge authorisation of the *Junta de Andalucía* (2010b), the water treatment line is described below:

– Pre-treatment: roughing with inclined auger screen (3 mm light) for separation of solids and subsequent compaction and dewatering. The water line continues to the desanding and degreasing unit.

– Biological treatment: an activated sludge system with prolonged aeration is used. The reactor is rectangular in shape and is separated into two lines; oxygen is supplied through fine bubble diffusers.

– Secondary decantation: it has two circular decanters with a diameter of 10 m, gravity type.

Experimental procedure

Sample collection and pre-treatment

Samples were taken at two points in the two WWTPs: in the influent, after coarse roughing to avoid larger waste, and in the effluent, before the treated wastewater is released into the river or stream.

Table 1. *Volume of samples*

WWTP	Inhab-eq	V annual (m^3)	Samples	
			Influent (L)	Effluent (L)
El Bosque	500	73,000	5	30
Prado del Rey	6,002	470,000	5	30

The samples were filtered through three stainless sieves of 1000, 355 and 100 µm mesh size. The solid fractions were then collected with ultrapure water into beakers and were left to dry in the oven at 70 °C.

Once the samples were dry, the process of oxidation of organic matter with wet peroxide oxidation (WPO) in the presence of an Fe (II) catalyst was carried out. After the WPO, density separation was carried out, which allowed the density of the solution to change, causing the polymers to float. Finally, the supernatants were filtered using a cellulose acetate filter with a diameter of 45 mm and a pore size of 0.8 µm. The filters were dried for approximately 4 hours at 40°C.

Identification and quantification of microparticles and microplastics

With the filters dry, we proceeded to the quantification and characterisation of microparticles and, finally, the identification of the type of polymers by FTIR spectroscopy.

RESULTS AND DISCUSSION

A comparison of the wastewater treatment plants was made and their microplastic removal efficiency was determined. Taking into account that different sieve sizes were used, ranging from 1mm to 100 µm, the size of the microparticles collected on each sieve is distributed. Figure 1 shows the amount of microparticles (n/L), according to the size dispersion in each of the wastewater treatment plants (influent and effluent). It was found that the highest number of microparticles in the influent was collected in the 100 μm sieve, reaching values of 426 n/L for El Bosque. As can be seen, the El Bosque WWTP has a high input of microparticles, but a higher yield, as fewer are discharged into the effluent.

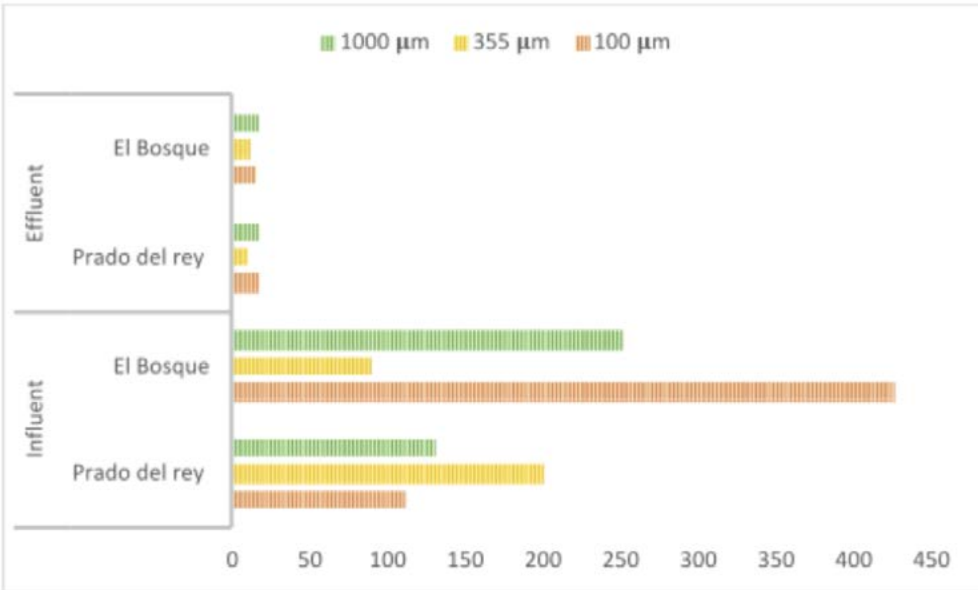

Figure 1. *Amount of microparticles (n/L), according to the size dispersion in each of the wastewater treatment plants (influent and effluent)*

Taking into account the visual PM identification technique mentioned in the methodology, the microparticles (n/L) identified in the influent and effluent are presented below in Table 2, according to their form: flake, sphere, filament, fibre and fragment.

Table 2. *Microparticles (n/L) identified in the influent and effluent.*

WWTP	El Bosque		Prado del rey	
Form	**Influent (n/L)**	**Effluent (n/L)**	**Influent (n/L)**	**Effluent (n/L)**
Flake	120	6.4	105.6	6.93
Sphere	3.2	0.4	2.4	0.27
Filament	125.6	7.74	71.2	4.27
Fibre	264.8	12.4	129.6	16.13
Fragment	248	15.73	120.8	15.2

One of the main forms that can be easily sorted in wastewater are fibres, which are significantly longer than wide (Sun et al., 2019). Fibres together with fragments are the forms that were found in the highest proportion in this study. The abundance of the fibres can be attributed mainly to the release of the fibres through the washing of synthetic garments (Browne et al., 2011; Napper et al., 2015; Thompson et al., 2009). On the other hand, it is inferred that the fragments correspond to secondary microplastic, due to the fragmentation of larger particles.

There is a large variability in the proportion of the type of polymers found in the WWTPs in this study, both in the influent and in the effluent. Sun et al. (2019), mentions that the most common polymers in wastewater are PE (4% - 51%) and PET (4% - 35%). However, PET was not detected in either of the two WWTs studied as shown in Table 3.

Table 3. *Type of polymers found in the WWTPs.*

WWTP	El Bosque		Prado del rey	
Polymers	**Influent (MP/L)**	**Effluent (MP/L)**	**Influent (MP/L)**	**Effluent (MP/L)**
PET	--	--	--	--
PE (HD and LD)	281.60	14.75	33.98	7.21
PVC	17.77	1.33	132.70	0.79
PP	--	--	8.35	--
PMMA	171.12	--	88.31	0.36
Others	142.15	7,66	14.19	--

Within the "Others" classification, a large variety of polymers was found, in particular up to eight different types were identified. Three of these polymers were identified in the Prado del Rey WWTP in low concentrations, corresponding to: styrene butadiene rubber (SBR), which is mainly used for tyres and the footwear industry; chlorinated polyethylene (PCL) and butyl styrate, which are used by the cosmetics industry for personal care products. The other types of polymers were identified in the industrial wastewater treatment plant, both in influent and effluent.

CONCLUSIONS

Microparticles were identified in the inlet water of the treatment plants at 429 n/L (Prado del Rey) and 761 n/L (El Bosque); while the range of microparticles in the outlet water is between 42 n/L. A great variability of polymers was observed; for the urban wastewater treatment plant where the predominant types were PVC, PE (HD-LD) and PMMA, attributed to the daily use of this type of plastic by human activities; while in the industrial wastewater treatment plant other types of polymers also appeared, among which SAN, PEVA and PMP stand out, related to the industrial activities that take place in the industrial area. The annual volume of microplastic discharge from the wastewater treatment plants studied has been estimated to be between 1.2-109 MP/year (Prado del Rey) and an exceptional case is the industrial wastewater treatment plant El Bosque, which contributes a significant amount of MP/inhab-eq, despite being the smallest wastewater treatment plant.

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HORMONAL RESIDUES IN A NATURAL WASTEWATER TREATMENT SYSTEM FROM GRAN CANARIA (SPAIN): PRESENCE, REMOVAL AND RISK ASSESSMENT

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Abstract
The presence of emerging pollutants in wastewaters has become a concerning issue for both scientific community and legislators becauseof the environmental problems associated to these contaminants when wastewaters are released into the environment. Some of those emerging pollutants are steroid hormones which can result in harmful effects over exposed biota, even at trace concentrations. Waste-stabilization ponds and constructed wetlands (CWs) are particularly suited wastewater treatment systems for small communities because of their low cost and easy maintenance. Nevertheless, it is necessary to ensure that these systems are effective in the elimination of emerging pollutants such as steroid hormones to guarantee that produced reclaimed waters are not a risk when re-used or discharged into rural ecosystems. In this work, a combined macrophyte pond-CW system was evaluated for the presence of fifteen steroid hormones. Eight different steroid hormone compounds were detected at concentrations of $ng{\cdot}L^{-1}$ in influent and effluent samples. The pond-CW system showed high elimination rates of steroid hormone

residues with average removal efficiencies of over 76%. This efficacy was confirmed by the ecological risk assessment evaluation performed.

INTRODUCTION

Among emerging pollutants, steroid hormones are a group of organic compounds of growing concern because of their potential capacity to disrupt the systems of aquatic organisms, even at concentrations in the range of $ng \cdot L^{-1}$ (Adeel *et al*, 2017). The concern about the deleterious effects of steroid hormones has provoked that some legislators such as the European Commission have added three oestrogens (oestrone, 17β-oestradiol and 17α-ethynylestradiol) to the first *Watch List* of the Water Framework Directive. The goal of the Watch List is to obtain "high-quality Union-wide monitoring data on potential water pollutants for the purpose of determining the risk they pose and thus whether Environmental Quality Standards (EQS) should be set for them at EU level" (Loos *et al*, 2018). Also, in the subsequent revisions of this *Watch List*these three steroid hormones have been remained to obtain more information about them and also some other steroids have been considered as future candidates (Gomez-Cortés *et al*, 2020).

The efficiency of some natural wastewater treatment systems such as ponds or constructed wetlands in the removal of emerging pollutants has been found to be very variable. This is because these systems are conditioned by many variables such as operational parameters, wastewater physico-chemical properties or ambient conditions. In this regard, ponds have shown removal efficiencies from 50 to 75% for estrogenic hormones, but lower than those obtained using activated sludge systems (Pessoa *et al*, 2014). On contrast, some studies have stated that CWs are very effective in the removal of steroid hormones, with some exceptions such as ethynylestradiol (Chen *et al*, 2019; Ávila *et al*, 2014). The removal of oestrogens in activated sludge systems has been well studied, but less is known in the case of CWs and to our knowledge, the efficiency of a combined macrophyte pond-CW system in the removal of steroid hormone residues has not previously been studied.

In this research, a monitoring study was performed to determine the presence and concentration of fifteen steroid hormones and their removal efficiency of a macrophyte pond-CW system treating raw wastewater from a university campus. Samples were collected for four months every ten days to evaluate the distribution of steroid hormones and their associated ecological risk

METHODS

The studied natural wastewater treatment system (NWTS)consists of a macrophyte pond and a horizontal flow CW. The systemhas been working for almost 20 years and treats the wastewaters produced in a part of the Campus of University of Las Palmas de Gran Canaria (Spain) which includes cafeterias, laboratories, sport facilities and toilets from different buildings. Hazardous laboratory residues are selectively treated and are not discharged in this system. The NWTS was designed to treat the wastewaters produced by 150 population equivalent (p.e.), i.e. 7.5 m^3 d^{-1} considering 50 L $p.e.^{-1}$ (the third of the wastewater produced by a person in one day in Canary Islands). The macrophyte pond is 1.8 m deep and has a surface area of 157 m^2 and a volume of 235 m^3, approximately. Water flows horizontally from the pond to the CW and, following the water direction, it is comprised of a stone filter, a free water channel, a second stone filter, a second free water channel and a final subsurface flow channel. Stone filters can be regarded as short horizontal subsurface flow CWs. The mean depth of the CW is 0.8 m. Basaltic stones (Ø ~10–15 cm) were used to construct the filters. Specimens of *Phragmites*, *Cyperus*, *Pontederia*, *Canna* and *Typha* were planted around the edges of the system. Grab samples were taken in three sampling points (Figure 1): 1) influent, 2) pond effluent or CW influent, which is located just after the first stone filter, and 3) CW effluent.

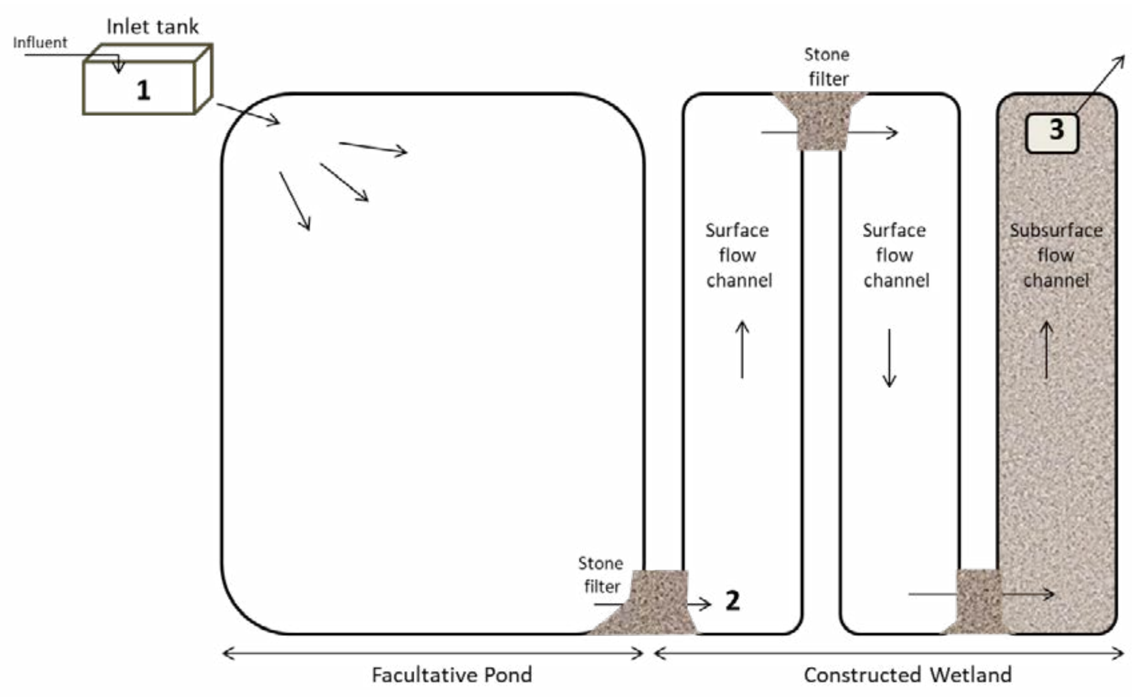

Figure 1. *Macrophyte pond – constructed wetland layout and sampling points*

For the determination of steroid hormone concentrations, an on-line solid phase extraction system coupled to an ultra-high-performance liquid chromatography system equipped with a triple quadrupole detector (SPE-UHPLC-MS/MS) was used (Guedes-Alonso *et al*, 2015).

RESULTS AND DISCUSSION

Eight different hormonal compounds were detected during the sampling campaign. Five of them were naturally occurring hormones (oestrone, oestriol, testosterone, cortisone and progesterone) and the other three were synthetic (norgestrel, boldenone and norethisterone). Influent samples showed the highest detection rates since the eight steroid hormones were detected at least in a 66% of the samples (except oestriol, 41.7%). After the pond treatment, the frequency of hormone detection decreased in all cases. Only three of the hormones under study (oestrone, norgestrel and norethisterone) were detected with a greater frequency after the CW treatment which could be explained by deconjugation processes.

Table 1 shows the average concentrations of hormones in the three sampling points. Natural hormones presented the same decreasing trend in concentration levels as in detection frequency and remarkable concentration reductions were observed. For testosterone or cortisone, average concentrations decreased from hundreds of $ng \cdot L^{-1}$ in the influent to few $ng \cdot L^{-1}$ in the final effluent. As for the synthetic hormones detected, boldenone presented the same trend as natural hormones. However, norgestrel and norethisterone presented a different trend. For these two compounds, the concentrations in the final effluent were slightly higher than at the intermediate point. Regarding seasonal variations, it can be said that in April there was an increase in the average concentrations of steroid hormones, which could be explained by the higher student attendance after the Easter break.

Table 1. Average concentration (minimum and maximum concentrations in brackets) in $ng \cdot L^{-1}$ of steroid hormones under study in the different sampling points.

	Influent	Pond effluent	CW effluent
Estrone	17.26 (12.46 – 19.99)	15.45	12.80 (12.33 – 13.26)
Estriol	<LOQ[a]	<LOQ[a]	n.d[b]
Norgestrel	247.65 (43.06 – 801.45)	11.40	22.07 (8.01 – 36.12)
Testosterone	183.23 (37.12 – 382.54)	41.35 (14.16 – 134.66)	15.12 (9.67 – 20.57)
Cortisone	127.00 (27.12 – 262.25)	28.28 (4.52 – 52.78)	5.38
Boldenone	53.97 (2.73 – 105.38)	36.14 (7.17 – 64.93)	8.27 (1.86 – 14.68)
Norethisterone	84.39 (14.49 – 183.72)	n.d[b]	8.10 (5.32 – 10.88)
Progesterone	53.05 (2.91 – 163.17)	46.66 (2.47 – 111.57)	12.19 (0.99 – 48.98)

[a] Concentration below quantification limit

[b] Not detected

With respect to the removal efficiency of the different treatments, in the macrophyte pond four compounds (oestrone, norgestrel, testosterone and norethisterone) presented average efficiencies of over 87%. For cortisone, boldenone and progesterone, the removal efficiencies were lower and, in some samples, cortisone and progesterone showed negative removal rates as the concentrations after the treatment were higher. This increase in concentration could be explained by hormone deconjugation. With the exception of estrone and testosterone, the rest of compounds showed higher removal rates in the CW. Nevertheless, it is important to highlight that the removal rates of both treatments cannot be compared because the treatments are set in series.

Overall, the pond-CW system provided high average removal efficiencies (over 89%) for six hormones and for the other two compounds detected, the average removals were over 76%.

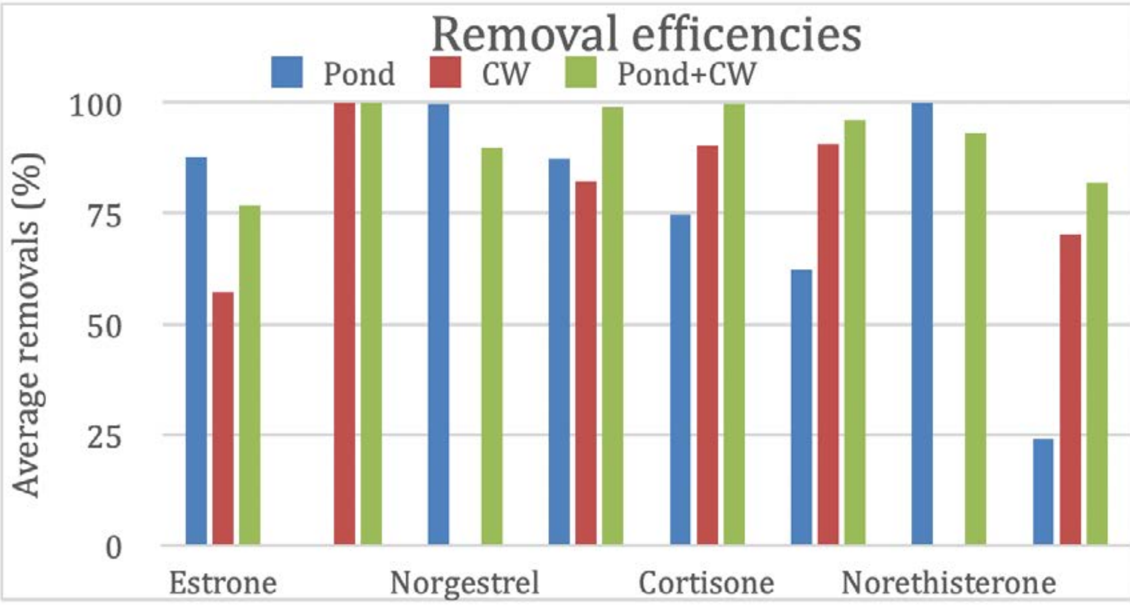

Figure 2. *Average removals in the different treatments and in the whole NWTS.*

Finally, regarding the ecological risk associated to steroid hormones in the different treatments, the effluent of the pond-CW system presented a low ecological risk associated with steroid hormones because of the high removal efficiencies obtained. In this regard, it is important to state that the compound with the highest risk quotient (RQ) value was testosterone, although this was not the compound with the highest concentrations. Norethisterone and progesterone had RQ values in the medium risk range, while the other detected hormones showed low ecological risk.

CONCLUSIONS

The macrophyte pond-CW system studied was found to be highly effective at reducing hormone residues and the associated ecological risk from wastewaters of small communities or rural areas. In the study, eight steroid hormones were detected in concentrations that ranged from 8.1 to 247.7 $ng \cdot L^{-1}$. Regarding the ecological risk, the medium to high risk that were found in the influent samples gradually diminished as the water passed through the system. The natural macrophyte pond-CW combination was found to be highly efficient in the removal of steroid hormone residues, with average removal efficiencies over 76%.

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04

PHYTO-TREATMENT USING MICROALGAE

EVALUATION OF SEWAGE TREATMENT THROUGH MICROALGAE-BACTERIA CONSORTIA IN A RACEWAY PHOTO MEMBRANE BIOREACTOR

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Abstract

Currently about 80% of wastewater worldwide is discharged without prior treatment and it reaches up to 95% in developing countries. The microalgae-bacteria consortia (MBC) have gaining attention as a result of its capacity of nutrients removal without mechanical aeration. However, the main limitation of this system is the low settleability of microalgae that makes effluent clarification a challenge. Therefore, this work aims to evaluate the potential of MBC cultivated in a in a raceway photobioreactor with an external membrane separation for the treatment of a real sewage study (no primary settling). Two reactors R1 (without membrane) and R2(M) (with membrane) were operated at 7 days of HRT. Both systems were operated at 26 ± 3 °C with 12 h light - 12 h dark cycles. The BOD removal in R2(M) was higher than in R1, mainly attributed to the suspended solids that in R1 remained after simple settling. Furthermore, the fecal coliforms in R2(M) were undetectable and in both systems the

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NH3-N and PO43- removal were high, close to 90% and 75%, respectively. The main removal mechanisms of nutrients were by algae assimilation. The contaminants removal and fecal coliforms were addressed without mechanical aeration and disinfection in R2(M). The membrane filtration allows to obtain a clarified affluent that could later be reuse. Then, MBC systems may be an interesting alternative for the treatment of sewage (before primary settling) in small communities, by reducing the number of required operations.

INTRODUCTION

Currently about 80% of wastewater worldwide is discharged without prior treatment and it reaches up to 95% in developing countries, promoting social, environmental and public health problems, mainly in rural and peri-urban communities. The microalgae-bacteria consortia (MBC) has emerged as an alternative for wastewater treatment, and it has gained attention as a result of its capacity to remove pollutants, without the need of mechanical aeration. This is the result of the oxygenation effect produced by photosynthesis (Oswald et al., 1953). In MBC, microalgae grow in presence of light and produce O_2 (photo-oxygenation), which is supplied to the hetero-trophic and autotrophic bacteria responsible for COD and ammonia conversion. Meanwhile, the heterotrophic bacteria produce CO_2 which is used as a carbon source by microalgae. Moreover, the NO_3 produced by nitrification can be assimilated as a nitrogen source by microalgae (Li et al., 2019). Then, MBC systems have the potential to provide an efficient wastewater treatment, at a fraction of the energy requirements of traditional alternatives, such as activated sludge. However, when applying MBC systems to wastewater treatment, effluent clarification can be a challenge, considering that normally microalgae cannot be harvested easily and efficiently by simple settling. Therefore, this work aims to evaluate the potential of a microalgae-bacteria consortium cultivated in a in a raceway photobioreactor with an external membrane separation for the treatment of a real sewage study (no primary settling).

METHODS

Two raceway photobioreactors with an effective volume of 50 L were used to conduct this study. One of the reactors was supplied with a membrane separation unit, that provided phys-ical biomass retention. Mixing was carried out by a paddlewheel, operated at around 10 rpm. In each system, light was provided by 6 white LED lights located over each reactor, which supplied an average light intensity of 450-500 mmol m^{-2} s^{-1} (on the surface of the liquid). Both systems were operated at 26 ± 3 °C with 12 h light - 12 h dark cycles. The photobioreactors were started-up by applying discontinuous feed for a week. Then were subsequently operated continuously for

100 days. An HRT of 7 days was applied, photobioreactors were fed with real sewage taken from Mapocho-Trebal treatment plant located in Metropolitan region, Chile. Sewage was collected after screening, and before primary settling. Average total COD during operation period was 807 mg L^{-1}. Reactors were inoculated with a mixture of activated sludge from the Mapocho-Trebal plant and microalgae obtained from a lab scale batch culture of *Chlorella Vulgaris*. Membrane system installed in one of the reactors consisted of a module of outside-on tubular microfiltration membrane. The membrane area was 0.042 m^2, aeration was provided in the module to control the fouling. The membrane was operated with filtration-backflush cycles of 10 and 1 min respectively. The complete biomass retention was provided by the membrane, then biomass waste was performed manually.

RESULTS AND DISCUSSION

During the 100 days of operation, we realize a monitoring campaign of some parameters such COD, nitrogen, phosphate, and volatile suspended solids (VSS) as is shown in ***Figure 1A.*** Due the effect of membrane module in the reactor the direct parameter affected was the VSS, in R1 (reactor without membrane retention) the VSS reached 1 g VSS L^{-1}. On the other hand, R2(M) (the reactor with membrane) reached higher biomass concentrations (≈ 3 g VSS L^{-1}) in the day 94 of operation. During the operation in R2 the biomass waste was performed manually several time to prevent excessive biomass development that could prevent excessive light obscuration. In ***Figure 2 (B)*** (***C)***, is possible to appreciate the difference of biomass concentration in the mixed liquor of both bioreactors.

In ***Figure 1(B)*** is shown the soluble COD of the effluent in both reactors (R1 and R2). In both cases, values were similar, in order of 100 mg/L that represents a removal of COD of 78.8 % for R1 and 79.8% for R2. However, such values represent effluent COD of R2(M), but not that of R1, since biomass was present in the effluent of latter system. In order to provide a more realistic scenario to compare the performance of both systems, composite samples were taken by the end of operation for a period of 48 h. Effluent of R1 was subjected to a process of 3 h settling in order to simulate the effect of a subsequent settler. Biochemical Oxygen Demand (BOD) analysis in the effluents of R1 (after settling) and R2(M) were performed. Concentrations of BOD were 108 and 32 for R1 and R2(M), respectively. Even though settling was effective in removing big part of the biomass (R1), an important part remained in the clarified, providing a higher BOD concertation, when compared to R2(M), where membrane removed all presented biomass. Enhanced clarification of R2(M) provided by membrane filtration resulted in undetectable fecal coliforms presence in reactor effluent. On the contrary, R1 effluent (after settling) presented a value of 1300 MPN per 100 mL. Those concentrations of BOD and fecal coliforms in the effluent R2 (M) meets with the parameters established in local normative for wastewater discharge (Standard DS90, section 4.2) in Chile. In ***Figure 2 (D)*** is shown the quality of R2(M) effluent and in ***Figure 2 (E)*** the quality of R1effluent after 30 minutes of settling.

On the other hand, ***Figure 1 (C)*** presents ammonia nitrogen concentration on the effluent of each reactor. In both cases removal was elevated. After day 25, NH_3-N remained below 5 mg L^{-1} in both reactors, meaning over 90% of removal. The removal of NH_3-N was probably mainly attributed to microalgal assimilation, since low concentrations of NO_3-N were found in the system, less than 5 mg L^{-1} NO_3-N in R1 and 15 mg L^{-1} NO_3-N in R_2. Other investigations have found similar nitrogen removal percentages ranging from 83% to 95% (Jia & Yuan, 2018; Khaldi et al., 2017; Su et al., 2012). Such performance demonstrates the high efficiency of the system, without need to supply an external oxygen source.

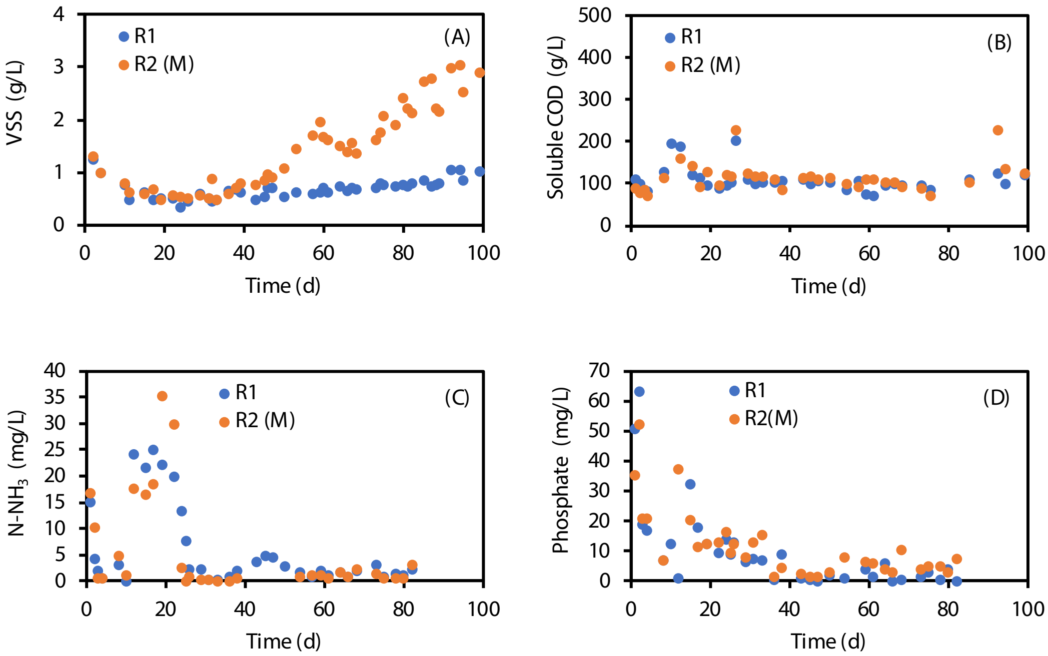

Figure 1. *Reactors performance during operation:* ***(A)*** *Biomass concentration in the reactor,* ***(B)*** *soluble COD concentration in the effluent,* ***(C)*** *ammonia nitrogen concentration in the effluent* ***(D),*** *phosphate concentration in the effluent. R1 represent the reactor without membrane, and R2(M) the one with biomass retention by membrane filtration*

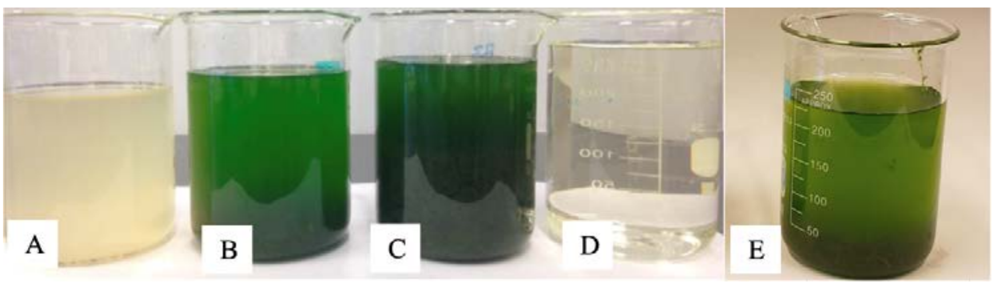

Figure 2. *Solids apparency in* ***A****. wastewater from Mapocho-Trebal treatment plant.* ***B****. mixed liquor of R1.* ***C.*** *Mixed liquor of R2.* ***D.*** *Effluent of R2.* ***E.*** *Effluent of R1 after 30 minutes of settling*

Regarding phosphate removal, in ***Figure 1 (D)*** shows the phosphate concentration in the effluent during reactors operation. Again, removal was high, most likely as a result of biomass assimilation. Phosphate removal was in the range of 75% for both systems, most likely as a result of biomass assimilation. In their research Khaldi et al. (Khaldi et al., 2017) obtained similar results, with a 76% of phosphate removal.

Finally, during R2(M) operation, membrane modules were operated at a flux in the range 10-15 L m^{-2} h^{-1}. Critical flux measurements revealed that those flux levels were compatible with an operation with reduced membrane fouling. Similar results were obtained in the research conducted by Jixiang et al. (Yang et al., 2018)which has not been attempted before. A low light intensity, 200 μmol/(m 2 ·s, using a suitable flux of 15 L m^{-2} h^{-1} enabled operating the filter of the PMBR at a transmembrane pressure as low as 4 kPa.

CONCLUSIONS

Results showed the capacity of MBC to provide efficient treatment of sewage. A relevant aspect of this research is that sewage used during this research was sampled before primary settling, so it contained a high fraction of suspended solids. Most of the reported research available have indeed used pre-settled sewage. Despite that fact, the systems showed quite good levels of organic matter and nutrients removal. Membrane filtration showed to be useful in order to provide a completely clarified affluent. It moreover provided an effluent almost free of fecal coliforms aspect that would certainly facilitate potential reuse of treated water, limiting or avoiding requirements of post disinfection. Then, MBC systems may be an interesting alternative for the treatment of sewage in small communities, by reducing the number of required operations.

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EXPERIMENTAL OPERATION OF AN ALGAE-BACTERIA MIXOTROPHIC REACTOR FED WITH URBAN WASTEWATER

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ABSTRACT

A raceway reactor with a mixotrophic culture of microalgae and bacteria was operated in a batch mode under different environmental conditions to determine the kinetic parameters governing the system and to translate it into controllable operating parameters. The kinetics of nutrient (N and P) removal and biomass growth were calculated, and the most unfavourable Hydraulic Retention Time (HRT) was adopted to operate with. An experimental methodology is thereby established with which to proceed each time the environmental conditions change in a similar system and to know the best strategy of operation. This work is based on a particular case, a real experiment, but it is extrapolated to a generic case to be applied in future plants with a phycotreatment system.

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INTRODUCTION

Currently, there are wastewater treatment technologies that can effectively remove nutrients (N and P); however, the cost of these processes and operational difficulties make their application in small communities impossible, and the eutrophication generated by the discharge of these pollutants is still a major problem. For this reason, effective, economical, and simple wastewater treatment technologies for the removal of nutrients and organic matter for these peculiar conditions are being investigated. The phycotreatment of wastewater, due to its simple operation, has become a viable alternative to the treatment options for small communities. This paper develops the experience of operating a raceway photobioreactor and provides recommendations to determine the variables that control this type of systems.

METHODS

In the studied system, the microorganisms are derived from the inoculum of bacteria and microalgae naturally developed in the reactor located at the CENTA (Experimental Centre of New Water Technologies) facilities (37°21'38.7 "N 6°20'01.3 "W). This 116.23 m^3 raceway, occupying an area of about 387 m^2, is 60 meters long, 3 meters wide per channel and has a depth of 0.3 meters.

The urban wastewater used to feed the reactor comes from the Sevillian town where CENTA is located, Carrión de los Céspedes (2500 inhabitants), the flow rate can be regulated according to the experimental needs and it passed through a roughing pre-treatment (up to 3 mm) and a desanding-degreasing process. Subsequently, to reduce the solids load in the reactor, a pre-treatment system of a 25 m^3 Imhoff tank was included. On the other hand, the reactor effluent flowed through a 13.5 m^3 secondary settler, with a diameter of 2.4 m. The settled biomass fed an anaerobic lagoon, where it was sent for digestion.

The medium speed inside the reactor was adjusted to a speed of about 0.28 $m \cdot s^{-1}$, which could be regulated with a mechanical gearbox and a manual variator installed between the six-bladed wheel and a 1.5 kW motor.

Continuous measurements of pH, dissolved oxygen (DO), temperature and PAR (Photosynthetically Active Radiation) were taken every 10 min. At later stages, a SOLITAX SC sensor was added to measure turbidity. In addition, periodic analyses of suspended solids (SS) by gravimetric methods and total phosphorus (TP), total nitrogen (TP) and COD by colorimetric methods were performed according to standard methods (APHA, 2012) on 0.45 µm filtered samples. Both in feed wastewater samples and reactor samples.

The modes of operation tested were batch and continuous. For the batch tests, the inoculum of the reactor consisted of one third of the reactor volume with the concentrated medium of microalgae and bacteria and quickly filled to its maximum capacity, at that time, the filling was stopped and the batch experiment began. This experiment consists on monitoring the evolution of biomass growth and nutrient removal (N and P), together with the rest of the studied parameters. Two "0 times" are distinguished in this experiment, the time of the beginning of the filling and the end of the filling (beginning of the batch). From this experience it is determined how to operate in such conditions the continuous test. In this case, the continuous experiment consists of operating at a flow rate determined by the optimum HRT estimated in the batch.

The kinetic models used to estimate biomass growth and nutrient removal were the logistic kinetic model of Verhulst (1838) (Eq.1) and a first order chemical reaction approximation but with unassimilable substrate (Eq.2), respectively.

$$\frac{dX}{dt} = \mu \cdot X \cdot \left(1 - \frac{X}{X_m}\right) \qquad \text{Eq. 1}$$

Where: X is the biomass concentration at time t, (mgSS·L^{-1}), X_m is the maximum concentration that the system can achieve in batch, (mg SS·L^{-1}), X_0 is the initial biomass concentration (t=0), (mgSS·L^{-1}) and μ is the maximum specific growth rate [d^{-1}].

$$S = \left(S_0 - S_{na}\right) \cdot e^{-k \cdot t} + S_{na} \qquad \text{Eq.2}$$

Where: S is the substrate concentration, which is, Total Dissolved Nitrogen (TDN) or Total Dissolved Phosphorous (TDP) in mg·L^{-1}, k is the kinetic rate constant of substrate removal, h^{-1}, S_0 is the initial substrate concentration at t=0 (mg·L^{-1}) and S_{na} is the non-assimilable substrate concentration (mg·L^{-1}).

The experiments took place from June 2018 to October 2019, with a distinction between batch and continuous experiments with different mixing regimes, which its comments are out the scope of this study.

RESULTS AND DISCUSSIONS

In the batch experiments, it is important to measure the irradiation, the temperature, and the composition of the feed water and to relate them to the kinetics obtained. For instance, for the composition of the water, we have characterized the feeding of the pre-treated water and then the Imhoff tank water. The results were compared, and the removal capacity of the system was calculated for each pollutant (Table 1).

Table 1. *Comparison of pre-treated water versus the same water after Imhoff tank treatment (Mean values and confidence interval)*

	Pre-treated	Imhoff Tank	%removal
SS [mg/L]	224,33 ± 24,99 (n = 9)	116,67 ± 44,25 (n = 9)	48%
COD_{tot} [mg/L]	984,54 ± 102,00 (n = 9)	704,64 ± 209,05 (n = 6)	28%
COD_{dis} [mg/L]	537,10 ± 35,24 (n = 9)	515,44 ± 31,29 (n = 9)	4%
TN_{dis} [mg/L]	71,11 ± 8,35 (n = 9)	65,37 ± 4,60 (n = 9)	8%
TP_{dis} [mg/L]	8,31 ± 0,37 (n = 9)	7,69 ± 0,19 (n = 9)	7%

The solids loading of the pre-treatment water influences the biomass analyses as well as the average irradiance intensity (I_{av}) inside the reactor. In terms of biomass productivity (calculated as the difference between the mass flow rates of suspended solids at the reactor outlet and inlet divided by the reactor volume, in continuous experiments) the results were 123,24 ± 53,05 $mg·L^{-1}·d^{-1}$ y 77,71 ± 16,86 $mg·L^{-1}·d^{-1}$, which means that, with pre-treated water, the culture produces more mixotrophic biomass.

One of the parameters that most influence microalgae growth and, therefore, nutrient removal are solar irradiance and temperature, both dependent on the climatic conditions of the location where the system is located (Lundquist et al., 2010). These parameters are not constant throughout the day or throughout the year, and there are circadian cycles that influence the metabolism of the culture and its pH and DO (Figure 1).

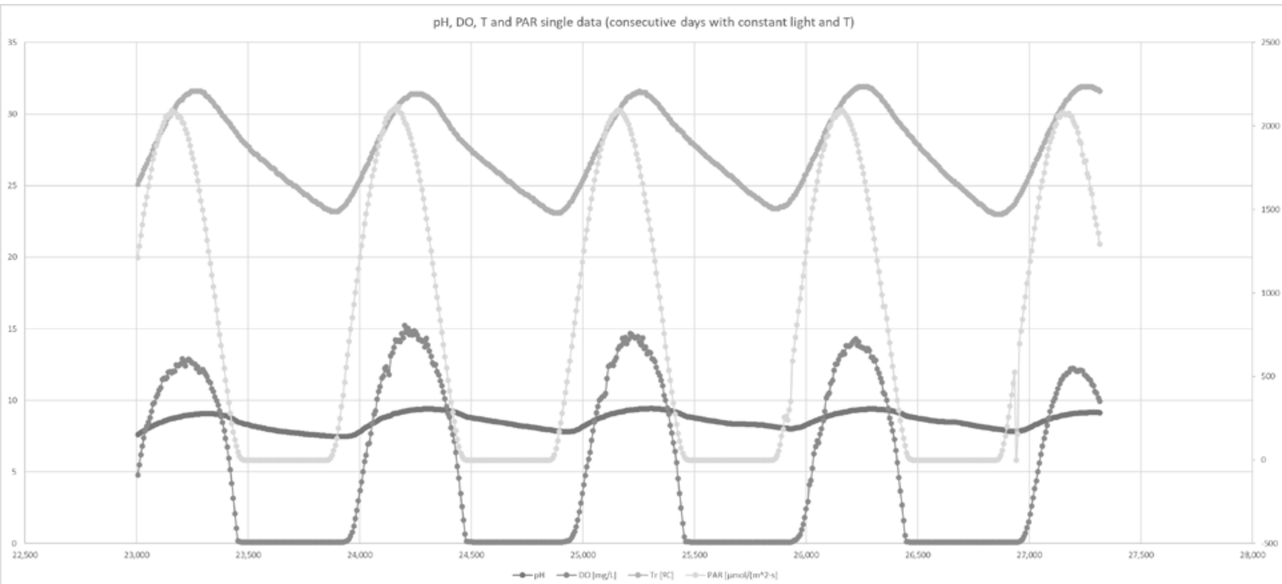

Figure 1. *Data for pH, DO, reactor temperature and PAR (secondary axis) on consecutive days (under constant daily T and PAR).*

For calculating biomass growth, the turbidity units were calibrated with suspended solids analysis, obtaining a calibration line that related this parameter with reasonable precision (R^2 = 0,9498). Using the Verhulst model and the least squares adjustment it is possible to determine the kinetic constants, which for the case of Figure 2 were: X_0 = 529,56, μ = 0,74 d^{-1} y X_m= 660,26 $mg{\cdot}L^{-1}$ (R^2= 0,8768). The same figure also shows the filling time, the end of filling time and the beginning of the exponential phase.

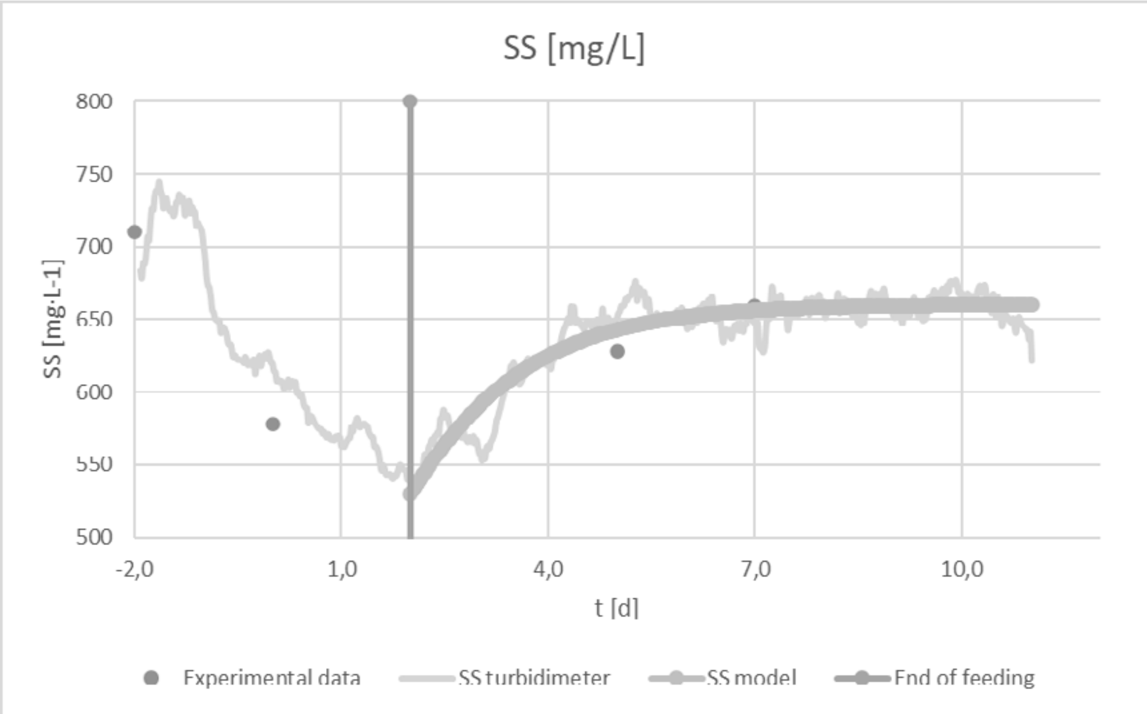

Figure 2. *Example of modelling the evolution of biomass during a batch experiment*

With this model, the optimal hydraulic retention time can be obtained according to the method used by Ruiz et al. (2013)biomass concentrations and productivities in continuous operation can be successfully predicted as a function of the specific hydraulic retention time (HRT, which, given the volume of the reactor, would be equivalent to operating with a flow rate of 43 $m^3{\cdot}d^{-1}$ (HRT =2,7 days).

Similarly, it is necessary to calculate the minimum hydraulic retention time required to reduce the concentration of N and P to levels permitted by Directive 91/271/CEE for populations discharging into sensitive areas (15 $mgN{\cdot}L^{-1}$ y 2 $mgP{\cdot}L^{-1}$). Two methods can be used to calculate this, one by using Equation 2 (first order reaction model) and another determined from the result of the experiments that calculates the kinetic parameters from the non-stationary phase of the batch experiment, in other words, during filling. With this method, we obtained Equation 3 and it was possible to model the nutrient concentration during filling as well (Figure 3).

$$S_2 = \left(S_{na} + \frac{Q \cdot S_F}{V \cdot k}\right) \cdot \left(1 - e^{-k\cdot(t_1 - t_2)}\right) \ \cdot + S_1 \cdot e^{-k\cdot(t_1 - t_2)} \qquad \text{Eq.3}$$

Where: S_2 and S_1 are substrate concentrations, mg·L^{-1}, at time t_2 and t_1, respectively, S_F the substrate concentration in the feed [mg·L^{-1}], V the reactor volume [m^3] and Q the flow rate [m^3·d^{-1}].

Once the experiment has been carried out and the kinetic constants have been determined, the targeted hydraulic residence times are calculated and, for this case, 3.3 days are required to eliminate nitrogen and 1.1 days to eliminate phosphorus up to admissible concentrations.

Knowing all the reaction rates involved, it is necessary to choose to operate at the limiting rate, that is, at the highest HRT, in this case 3.3 days, at a flow rate of 35.2 m^3.

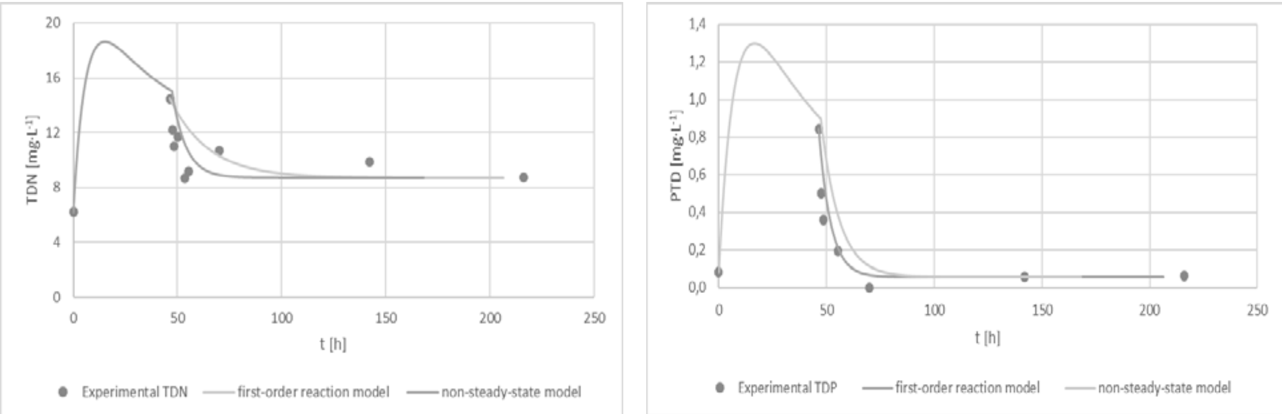

Figure 3. *Evolution of TDN (left) and TDP (right) concentration during the batch experiment. Dots: Experimental data; lines: first-order reaction model and non-steady-state model.*

CONCLUSIONS

To operate a wastewater treatment plant with a phycotreatment system, environmental parameters must be considered. Light and temperature are fundamental and their variation throughout the year implies changes in the operation of the reactor seasonally. Feed composition is not a controllable parameter either, although with an adequate pre-treatment design we can limit the concentration of certain pollutants in the reactor influent. The method described can be used to determine the optimum operating mode for different environmental conditions throughout the year by means of batch experiments. In the CENTA the controllable parameter is the flow rate (higher flow rate lower HRT), since there are other systems that support the flow rate variation. In the case of a new plant, several raceway reactors would be needed to support the periods of batch experiments and to allow modification of the HRT by operating with more or fewer reactors. Volume would be the control parameter in this case (higher volume, higher

HRT), so it is advisable to design the plant considering periods in which kinetics are more unfavourable, generally winter.

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PHOTOSYNTHETIC BACTERIA-BASED MEMBRANE BIOREACTOR AS AN ADVANCED TREATMENT OF A SECONDARY EFFLUENT

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Abstract:
The design of more sustainable water regeneration processes, which promote the recovery of organic matter and nutrients present in wastewater, will allow progress in the concept of regeneration and integral reuse of water. In this work, the viability of a membrane photobioreactor as an advanced treatment of a conventional secondary effluent is assessed in order to improve the final quality of the treated water and evaluate the growth rate of the combined biomass made up of bacteria and microalgae. Likewise, the influence of the development of membrane fouling and the loss of permeability experienced by it has been evaluated. Irreversible fouling that can compromise the full-scale operation of the process could be linked to the presence of biopolymers.

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INTRODUCTION

The potential of microalgae-based technologies for the removal of nutrients from municipal wastewater has been widely demonstrated (Gonçalves et al., 2017). However, as a consequence of the low degree of flocculation of the microalgae, the separation of the biomass is usually identified as one of the main limitations of this type of process. In conventional photobioreactors, the operation is limited to high hydraulic residence times to avoid washing of the microalgae. In this context, the use of membrane bioreactors has aroused great interest in recent years, because it allows decoupling the hydraulic residence time (HRT) and the cell retention time, thus increasing the concentration of microalgae and, thereforethe treatment capacity (Luo et al., 2017).

The present work focuses on the study of a laboratory-scale membrane photobioreactor applied to the treatment of a secondary effluent, analyzing biomass growth, treatment efficiency and membrane fouling.

MATERIALS AND METHODS

The photobioreactor consisted of a transparent high-density polyethylene tank with an effective volume of 2.8 L. The system was continuously illuminated at an intensity of 490 μmol/(m^2s) and the temperature of the reactor was kept constant at 20 ± 2 °C. The photobioreactor contained a ZeeWeed® ZW1 hollow fiber module (SUEZ Water Technologies and Solutions). Fibers of polyvinylidene fluoride (PVDF) with a hydrophilic and non-ionic surface were arranged longitudinally around a central axis that led air towards the base of the module. The membrane had a nominal pore diameter of 0.04 μm and the fibers an external diameter of 1.9 mm. The total filter surface of the module was 0.047 m^2. Filtration was carried out at constant flow (J = 10 L/(h m^2)), recording the transmembrane pressure over time. The fouling of the membrane was controlled by backwashes applied every 7.5 min at constant conditions (J_B = 30 L/(h m^2); t_B = 30 s). Fouling was quantified by monitoring the initial transmembrane pressure $TMP_i$of each filtration cycle, which evaluates irreversible fouling that cannot be removed by backwashing and TMP_fthat reports the final fouling that reaches the membrane at the end of each filtration cycle.

The system operated at an HRT of 0.75 days and without biomass purge. The feed, suspension and permeate were characterized by the usual parameters: chemical oxygen demand (COD), total organic carbon (TOC), nutrient concentration ($N\text{-}NH_4^+$, $N\text{-}NO_2^-$, $N\text{-}NO_3^-$ and $P\text{-}PO_4^{3-}$) and suspended solids (SST) according to standardized methods (APHA, 2005). A preliminary identification of the microorganisms developed was carried out by light microscopy (DM 750, Leica).

RESULTSAND DISCUSSION

The system operated for 650 h without previous inoculation of biomass, allowing the free growth of the species found in the diet. After 500 h, a stable biomass concentration was obtained (1889 ± 132 mg/L). Under these conditions, a moderate elimination of organic matter (22% as COD) and phosphorus (21%) was achieved, as well as high nitrification (88%) (Table 1). The analysis of the biological suspension showed the presence of different types of microalgae, diatoms and cyanobacteria (*Scenedesmus* sp., *Nitzschia* sp. And *Oscillatoria* sp.).

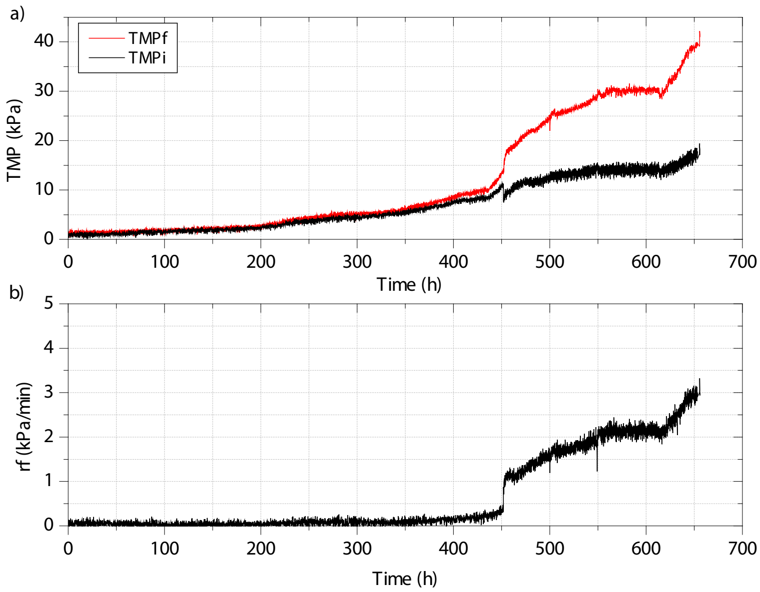

Figure 1. *Evolution of the initial and final transmembrane pressure (a) and the fouling rate (b) during the experimental period.*

Figure 1 shows the evolution of the transmembrane pressure at the beginning of the filtration (TMP_i) and the transmembrane pressure at the end of the filtration cycle (TMP_f). Two types of fouling are seen, identifiable by their effects on both transmembrane pressures after backwashing: reversible and irreversible fouling. The increase in TMP_i, continuous during the experimental period, is associated with the notable presence of groups of biopolymers (23 mg/L, measured as TOC) and is considered the cause of irreversible fouling. After 450 h, a significant increase in TMP_f was observed, associated with reversible fouling, possibly due to the increase in biomass concentration (from 466 mg/L at 220 h to 1889 mg/L under stable conditions). These results are consistent with the evolution of the fouling rate, r_f, defined as the slope of the TMP during the filtration cycle, which remained stable and at a practically zero value during the first 450 experimental hours, to undergo an appreciable increase until reaching values of approximately 3 kPa/min.

	TOC (mg/L)	COD (mg/L)	N-NH_4^+(mg/L)	N-NO_2^- (mg/L)	N-NO_3^-(mg/L)	P-PO_4^{3-}(mg/L)
Feedwater	18±9.8	73±46	20±14	1.3±2.2	2.3±3.5	1.9±1.6
Permeate	17±8.2	57±33	2.4±4.8	3.2±3.9	18±13	1.5±1.3

Table 1. *Characteristics of feedwater and permeate*

CONCLUSIONS

The membrane photobioreactor showed significant nitrification and an appreciable elimination of organic matter and phosphorus under the conditions studied. The biological suspension included different types of microorganisms among which *Scenedesmus* sp., *Nitzschia* sp. and *Oscillatoria* sp. The results showed the adequate efficiency of the process as an advanced treatment of treated water, obtaining a significant growth of biomass with a high potential for energy use.

ACKNOWLEDGMENTS

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NEW AGITATION AND SEPARATION TECHNOLOGIES FOR SEMI-INTENSIVE MICROALGAE WASTEWATER TREATMENT

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Abstract:
This work studies the effect of promoting vertical mixing on the efficiency of urban wastewater treatment through the use of microalgae biotechnology in raceway reactors. The use of bacteria-microalgae mixotrophic biological reactors is a technology with a high potential in the treatment of wastewater from small populations, highlighting as main advantages: (a) the lower energy consumption (around 75% less than by activated sludge treatment); (b) the elimination of CO_2 emissions associated with bacterial activity; (c) high nutrient removal; (d) a valuable sludge production; and (d) lower maintenance needs due to its simplicity of operation and because it requires less equipment with moving parts.

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The study is aimed at trying to improve the most unfavorable aspects of purification with microalgae, such as the productivity of the culture, which in large tanks is fundamentally limited by the availability of light from the microalgae, and the decantability of the generated flocs.

The stirring system that produces a greater intensity of vertical mixing of those tested is that of the paddle-wheel, although this system has the highest energy consumption. The SBTech blade-based stirring system is shown to be a promising technique due to its practically negligible energy consumption (derived from the low friction they introduce) and its very low cost, while achieving greater biomass growth and the formation of algae-bacteria flocs is favored, which decant in a much shorter time than when only a paddle wheel is used as a stirring technique for the entire reactor. Furthermore, the final effluents present a higher quality from the point of view of discharge requirements.

INTRODUCTION

The treatment of wastewater generated in small urban agglomerations presents a series of conditioning factors, both technical (strong daily and seasonal flow fluctuations and loads to be treated) and economic (the advantages of the economy scale are not applicable), which require the implementation of specific treatment solutions adapted to these conditions.

In this regard, the EPA in 1977 (EPA, 1977) recommended the following criteria when selecting the type of treatment to apply in small agglomerations:

– Technologies that require a minimum maintenance and that this maintenance is not excessively complex.
– Minimum energy consumption.
– Effective operation over a wide range of flow and load.
– Treatment facilities where possible equipment and process failures cause the least quality deterioration in the effluent.
– Maximumenvironmentalintegration.

At present, to this set of requirements has been added the need to adapt these technologies in order to achieve the reductions in nutrient concentrations (N and P), which are required by the increasingly demanding discharge parameters.

The phytoremediation of wastewater through the use of microalgae-bacteria mixotrophic systems is a very promising technological solution for the simultaneous removal of nutrients and organic matter from wastewater from small urban centers, with a much lower energy cost per cubic meter of treated water than conventional biological treatments. Furthermore, another advantage of this technology is its simplicity of operation, resulting in less need for maintenance and, therefore, reduced personnel costs.

Phytoremediation takes advantage of the nutrient consumption kinetics of these microorganisms together with the elimination of other pollutants such as drugs (Villar-Navarro et al. 2018), heavy metals (Yu and Wang 2004) and even, in general, to achieve greater wastewater detoxification (Díaz-Garduño et al. 2017). Likewise, phytoremediation allows promoting the joint growth of bacteria and microalgae in the same photobioreactor, establishing a synergy between both populations. In this way, and without the need to consume energy in aeration, it is possible to reduce the pollutant load in organic matter and nutrients simultaneously (Muñoz and Guieysse 2006).

Finally, another aspect to highlight is that purification with microalgae greatly reduces the carbon footprint in relation to traditional aerobic systems, not only because of the reduction in energy consumption that has already been mentioned, but also because the release to the atmosphere of CO2 produced by aerobic bacteria, which is assimilated by microalgae in their natural growth.

The photobioreactors that are most used on an industrial scale for water treatment are raceway or High Rate Algal Pond (HRAP) systems. These reactors consist of ponds with a high length / width ratio, where the water normally has a depth of less than 30 cm and is driven by a paddle-wheel that rotates around a horizontal axis (figure 1).

Figure 1. *HRAP of 116 m^3 located in the R&D Center of the CENTA Foundation, Carrión de losCéspedes (Seville)*

One of the main drawbacks of this technology is the high surface requirements, since the growth kinetics of photosynthetic microorganisms is limited by the availability of light in the reactor, which is heterogeneously distributed among the microbial population, given the low turbulent intensity of the flow in the major part of these reactors. For this reason, one of the main objectives of the IDIAqua project, in which this work is framed, has been the study of different agitation techniques of HRAP and their influence on the characteristics of the crop and the quality of the treated water.

The Reynolds number in HRAPs is typically on the order of 10^5, so the flow is in a fully developed turbulence regime. However, the intensity of the turbulence is very low in most of the reactor, except in the area near the paddle-wheel, in the bottom and wall boundary layers, and on the surface of the water on days with intense wind. This makes mixing in a large tank very limited. For the growth of microalgae this is critical, since only those microalgae that are a few centimeters below the surface receive enough light to carry out photosynthesis. Under these conditions, the period of the light-dark cycles is well above the optimal values for the growth of most microalgae species, which is around 1-10 s.

MATERIAL AND METHODS

Culture medium: urban wastewater

This study of wastewater treatment with microalgae has been carried out with urban wastewater from the Sevillian population of Carrión de losCéspedes (around 2,500 hab.eq.), mainly with agricultural activity and with some agri-food industry (N 37°22'10.88, O 6°19 '45, 313 "). This wastewater is treated in the R&D Center managed by the CENTA Foundation.

The wastewater receives a pretreatment before feeding to the reactor, which consists of a roughing with a coarse screen (3 cm), followed by a fine grid (3 mm), and in an aerated grit-degreaser, equipped with a sand classifier and fat concentrator. Next, the water undergoes primary treatment by means of a 25 m3 Imhoff tank, which operates in the settling zone with a retention time of 90 minutes at maximum flow rate. After this treatment, an electromagnetic flow meter / recorder is available, which allows determining and regulating the daily feeding flow to the bioreactor.

Microorganisms

During all the experiments we have worked with a natural bloom of native species of bacteria and microalgae, which have colonized and grown naturally in the raceway reactor, which has been operating for years. The dominant population is variable, highlighting the Scenedesmus species as the dominant one, although there are also manychlorophyceae.

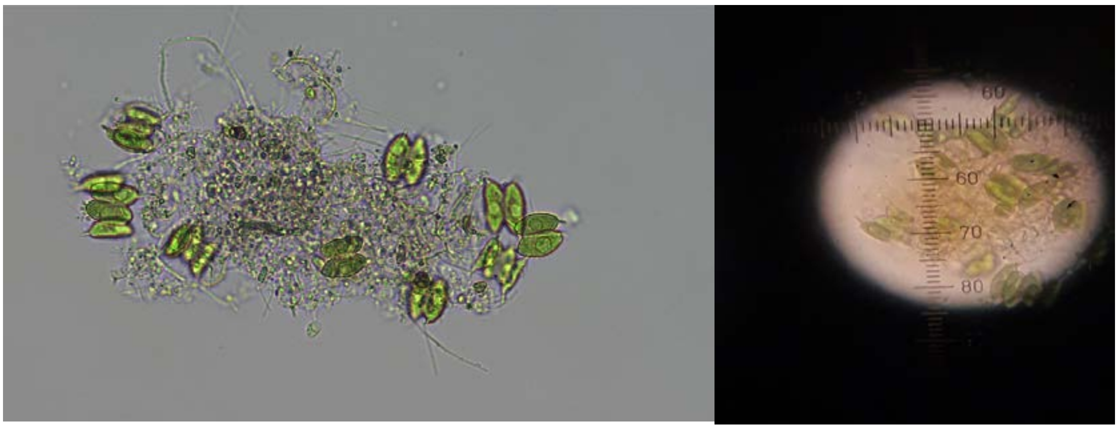

Figure 2. *HRAP microalgae and bacteria flocs (highlighting the presence of Scenedesmus- The image on the right shows a typical floc from the bioreactor used in the present work.*

Experimental set-up

The main equipment consisted of a single 400 m^2 open raceway photobioreactor built in reinforced concrete (figure 1), consisting of two parallel channels 3 m wide and 70 m long, separated by a central wall. To reduce the presence of dead zones, the ends of the channels end in curved shapes and have guide baffles to uniform the flow in these zones.

The movement of the water is achieved with a six-bladed paddle wheel, rotating by the action of a 1.5 kW motor. Thanks to a mechanical gearbox with manual variator, the revolutions of the wheel can be modified. During all the experiments, they were maintained at 4.8 rpm, which, as determined with the current meter, corresponds to a horizontal speed of the culture of 0.21 m/s, suitable for this type of systems, since it must be greater than 0.1 m/s to avoid sedimentation of algae in the channel (Weissman, et al., 1988) and less than 0.6 m/s to avoid cell damage (Hadiyanto et al., 2013).

A height-adjustable weir, located in the evacuation area of the photobioreactor, allows the height of the water height to be changed between 0.2 and 0.3 m. In the course of the works presented, this height has been kept at 0.3 m, which corresponds to a useful volume of the photobioreactor of 116 m^3.

To harvest the algal biomass, the effluents from the reactor are led, by gravity, to a decanter built in GRP and with cylindrical-conical shape, with an effective volume of 13 m^3. The effluents from the reactor enter the decanter through a soothing hood, leaving the clarified waters through a circular Thompson weir, which has a baffle to minimize the escape of floats.

At the bottom of the decanter there is a submersible pump, with timed operation, which allows the periodic purging of the settled bacterial/algal biomass. By means of a set of valves, it is possible to regulate which part of the decanted biomass is recirculated to the reactor and which part is extracted from the system and sent to a storage pond.

To improve the mix provided by the paddle-wheel, 3 transverse lines of PE pipes were arranged inside each channel, with 3 mm perforations every 45 cm, 15 m apart, the first of the lines being 6.45 m from the paddle wheel. These lines were connected to a 0.9 kW blower, with an air flow of around 1,800 l/min.

Subsequently, a series of SBTech® type fixed blades, designed by D&B Tecnología S.L., were installed inside the raceway channels, with the aim of producing a vertical mix without energy input, beyond a reduced pressure drop. less than 3% of the energy consumed by the paddle wheel. These blades, patented by the University of Seville (PCT ES2019070842), generate wing-tip vortices similar to those that appear on aircraft wings or turbo-machine blades. Numerical simulations CFD (Computational Fluid Dynamics) have been used as a fundamental tool for its design. Figure 3 shows a cylindrical vortex parallel to the flow direction that arises from the marginal edge of one of these blades.

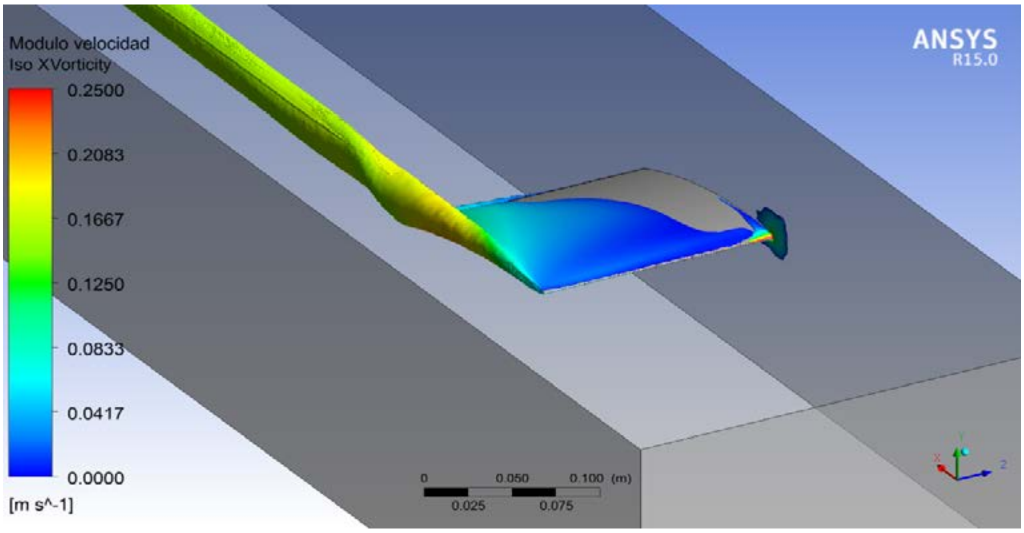

Figure 3. *CFD numerical simulation of the cylindrical vortex arising from the marginal edge of aSBTech® blade.*

3 lines of SBTech® blades were installed, one 28 meters from the paddle-wheel in one of the channels and two lines in the other, 20.5 m apart, having as a central reference the line of blades of the opposite channel. This blade system was optimized for the dimensions and depth of the CENTA Foundation raceway in terms of chord, curvature, span and angle of attack, as well as the relative position between the blades.

To monitor pH and DO in the reactor, the system has an Endress + Hausser Liquid line Probe with Orbipac CPF81D and Oxymax COS61D sensors from which data is extracted every 10 min. A HOBO U12 4-External probe has also been installed with a PAR radiation sensor (photosynthetically active radiation, with a wavelength range between 400 and 700 nm) and temperature. On the other hand, the turbidity was measured using a SOLITAX SC turbidity sensor, coupled to an SC200 controller, both from the HACH brand. The installation has an electric meter, which allows determining the energy consumption of the entire treatment system, to determine the kWh/m^3 ratio of treated wastewater.

Experimentaldesign

Three stirring and mixing systems operating continuously have been compared:

– Reactor only with the paddle wheel (PW).
– Reactor with six lines of perforated pipes that give a total air flow of 1,800 l/min, through daytime pulses of 100 s every 15 min (BB).
– Reactor with three lines of SBTech® blades distributed along the channels (SB).

Each system was operated in the same way: the reactor was fed from the Imhoff tank at a nearly constant flow rate and after a few days of operation and after having reached steady state conditions, the samples were taken.

The steady state condition was determined by the turbidity collected by the SOLITAX SC probe: if no large changes were observed over several days it was assumed that a steady state had been reached.

Data collection consisted of taking a sample from the feed and from the interior of the reactor for three non-consecutive days, within the steady state condition. These samples were analyzed: Suspended Solids (SS), Total Dissolved Nitrogen (NT), Total Dissolved Phosphorus (PT), and Total and Dissolved Chemical Oxygen Demand (CODtot and CODdis).

Finally, settling tests were also carried out in situ using the turbidimeter with a recording interval of one minute and an Imhoff cone. In this way it was possible to know and record the speed with which the culture decants.

Analysis procedure

Suspended solids analyzes (SS) were performed by gravimetric methods collected by standardized methods (American Public Health Association et al., 2012).

Nutrient analyzes (total nitrogen, NT, and total phosphorus, PT) were determined by a modification of the method proposed by Köthe and Bitsch (1992), in which 10 ml of sample are mixed with 1.5 micro spoons of OXISOLV® (Merck KGaA), and digested at 105°C for 90 minutes. Once the samples had been oxidized to nitrate and phosphate, and cooled to room temperature, they were analyzed using colorimetric methods of the Spectroquant® Cod. 1.14773.001 (Merck) kits and the standardized method with Ref. 4500-PE from the American Public Health Association et al. (2012), respectively.

On the other hand, the COD analyzes of samples, both total and dissolved, were carried out following the colorimetric method with Ref. 5220 D and D1252-06 (American Public Health Association et al., 2012), and chlorophyll by method 446.0 collected in Arar (1997).

Tests for determination of vertical mixing

To determine the degree of vertical mixing, an acoustic velocimetry Doppler probe, or Nortek Vector currentmeter, was used. 64 data per second were taken from the velocities in the 3 components of space.

The points analyzed are shown in figure 4 for the reactor operating only with the paddle-wheel (upper diagram), in intermittent bubbling regime (central diagram) and with the SBTech® blade system installed (lower diagram).

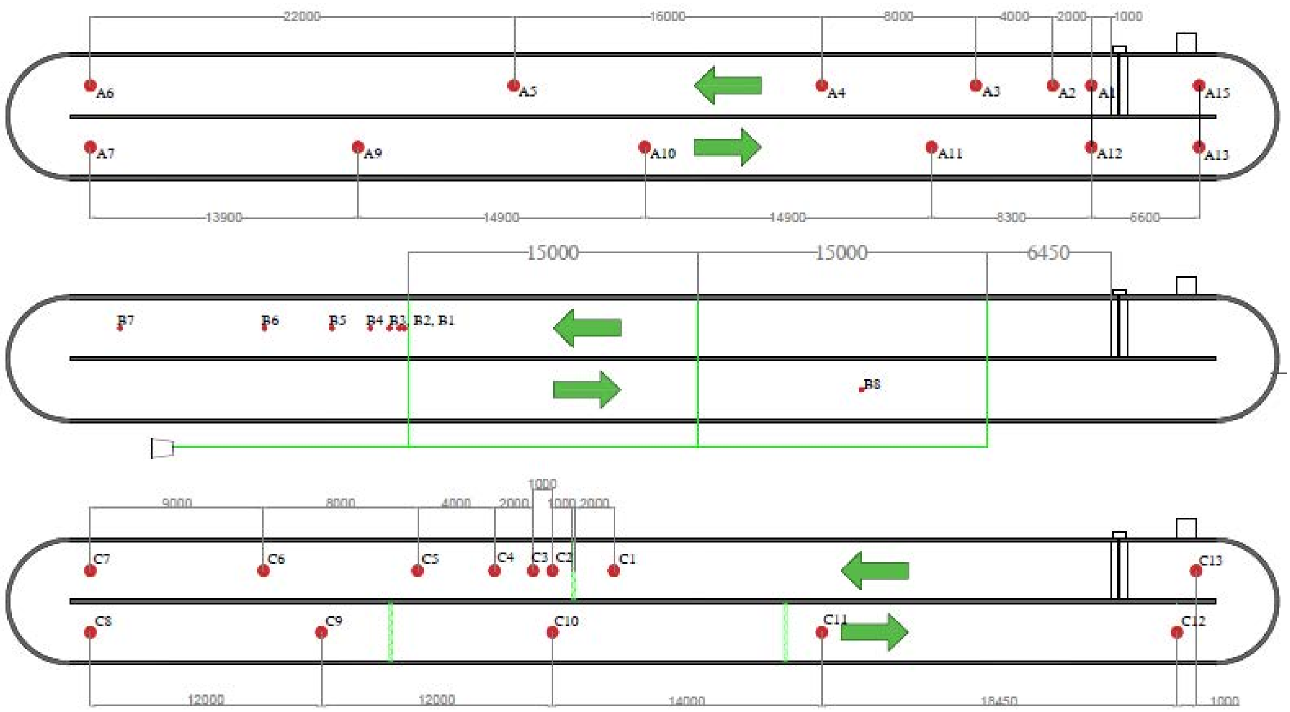

Figura 4. *Esquemas de los tres casos analizados (PW, BB y SB) con la ubicación de las tuberías de aireación, las líneas de álabes y los puntos de muestra (arriba PW, centro BB y abajo SB).*

RESULTS AND DISCUSSION

Climatic parameters and input composition

Table 1 shows a summary of the radiation, temperature and flow data from the tests carried out. We can consider that the light and the temperature are equivalent in the three cases, since the difference does not exceed 10%. However, as one of the flows differs so much, it is convenient that the composition of the inlet wastewater is treated in mass flow units when comparing. The mean mass flows of pollutants are shown in Table 2.

Table 1. *Mean values of the mean daily radiation (PAR), the mean of the mean daily temperatures (T) and the mean of the daily flows (Q), together with their confidence intervals (α = 0.05) and the number of samples.*

	PW	BB	SB
PAR [μmol·m^{-2}·s^{-1}]	713,61 ±11,4 (n=24)	664,11 ±22,4 (n=42)	658,49 ±35,5 (n=21)
T [°C]	24,23 ±0,7 (n=24)	26,45 ±0,5 (n=42)	23,71 ±0,7 (n=14)
Q [m^3/día]	26,36 ±1,8 (n=25)	21,05 ±1,2 (n=42)	22,76 ±1,3 (n=21)

Table 2. *Mean mass flow rates of the pollutants (mean flow rate x mean concentration) and, in parentheses, the difference with respect to the maximum of the three cases.*

	PW	BB	SB
NT [kg/día]	1,66 (%0)	1,08 (%34,7)	1,49 (%10,5)
PT [kg/día]	0,17 (%4,8)	0,13 (%24)	0,18 (%0)
SS [kg/día]	5,17 (%0)	3,03 (%41,4)	2,66 (%48,7)
DQO_t [kg/día]	18,33 (%12,6)	8,22 (%60,8)	20,96 (%0)
DQO_d [kg/día]	12,61 (%0)	4,3 (%65,9)	11,73 (%6,9)

The percentages presented in Table 2 reflect what deviates from the maximum value among the three experiments. Except for the mass flow rate of suspended solids, we can say that the amount of nutrients that entered during the experiment PW and SB are practically the same, while the amount that entered BB is lower.

Verticalmixing

The data collected by the Doppler probe make it possible to obtain the vertical speeds produced by each of the installed elements, which makes it possible to determine a value of the vertical mixing intensity as the mean square speed. Figure 5 shows the values of the vertical mixing intensity as a function of the downstream distance of the different mixing elements that have been analyzed.

It is observed that the intensity of vertical mixing produced by the paddle-wheel is clearly higher than that of the other systems, also having a persistence greater than 5 meters. The bubbling system produces a very weak agitation, although it maintains the agitation throughout about 15 meters above 0.01 m / s. The agitation produced by the SBTech blades is not as intense as that produced by the paddle-wheel, but it is much more intense than that produced by the bubbling. Its persistence along the raceway channels is less than 3 meters.

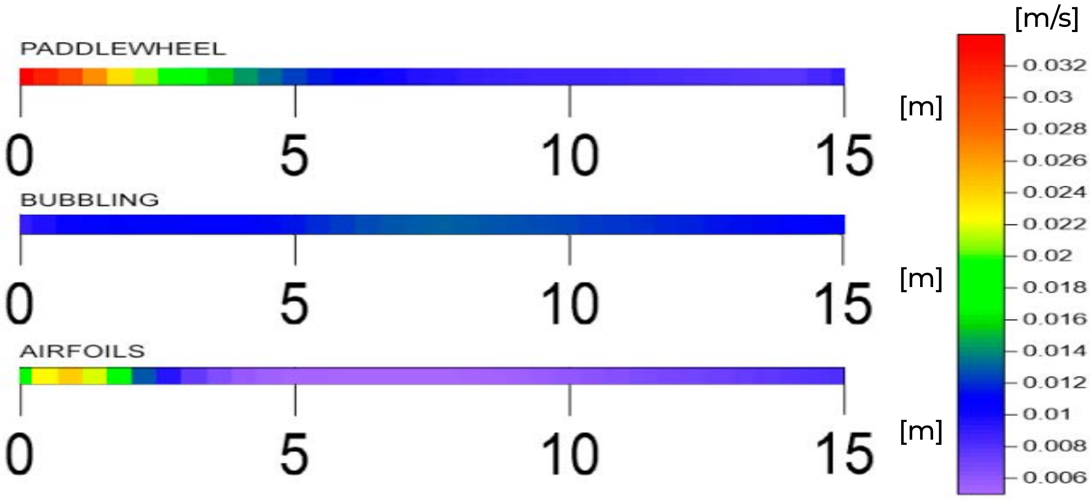

Figure 5. *Intensity of the vertical mixture (m/s) throughout the first 15 meters starting from the stirring element used in each case.*

Figure 6 shows the vertical mixing intensity distribution in the reactor for the three systems. The area near the paddle wheel (upper right part of the reactor) stands out. The lower image also clearly shows the increase in vertical mixing in the area close to each of the three lines of blades installed.

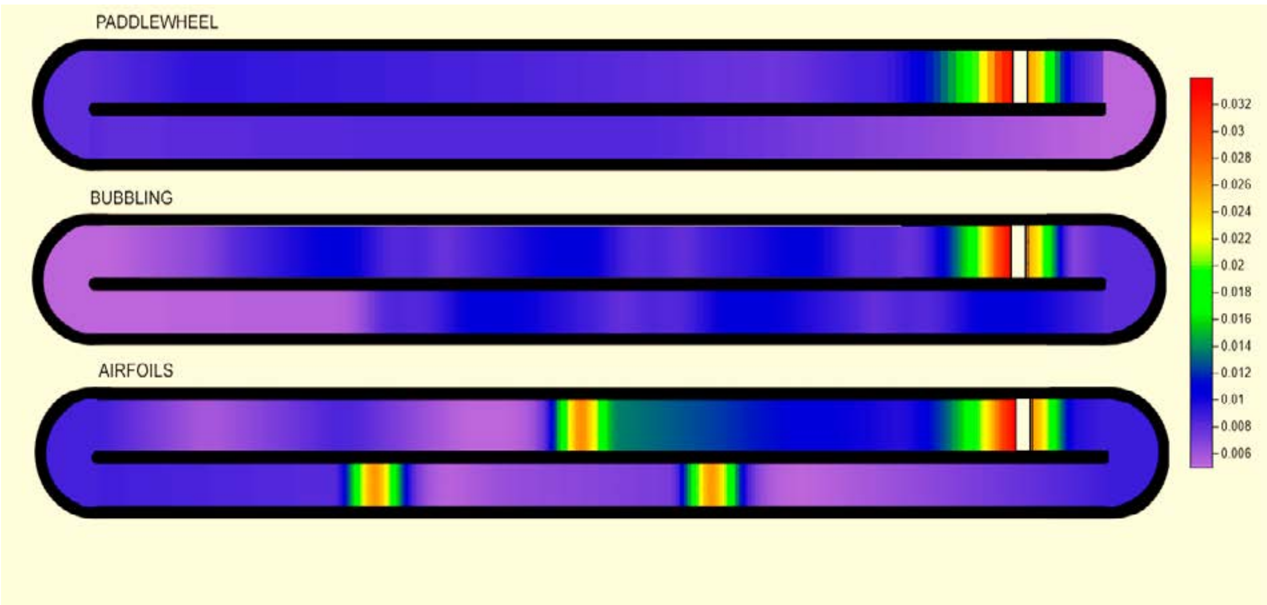

Figure 6. *Vertical mixing distribution in the three cases studied: above the reactor agitated only by the paddle-wheel; in the center with the intermittent bubbling system and in the lower image with three lines of blades.*

Using the results of the distribution of the mixing intensity produced by the different mixing elements analyzed, the mixture in the entire reactor can be determined for each case. Figure 7 shows the points at which the speed in the reactor was measured when only the paddle wheel (PW) was present, with intermittent aeration (BB) and with the SBTech blades installed (SB).

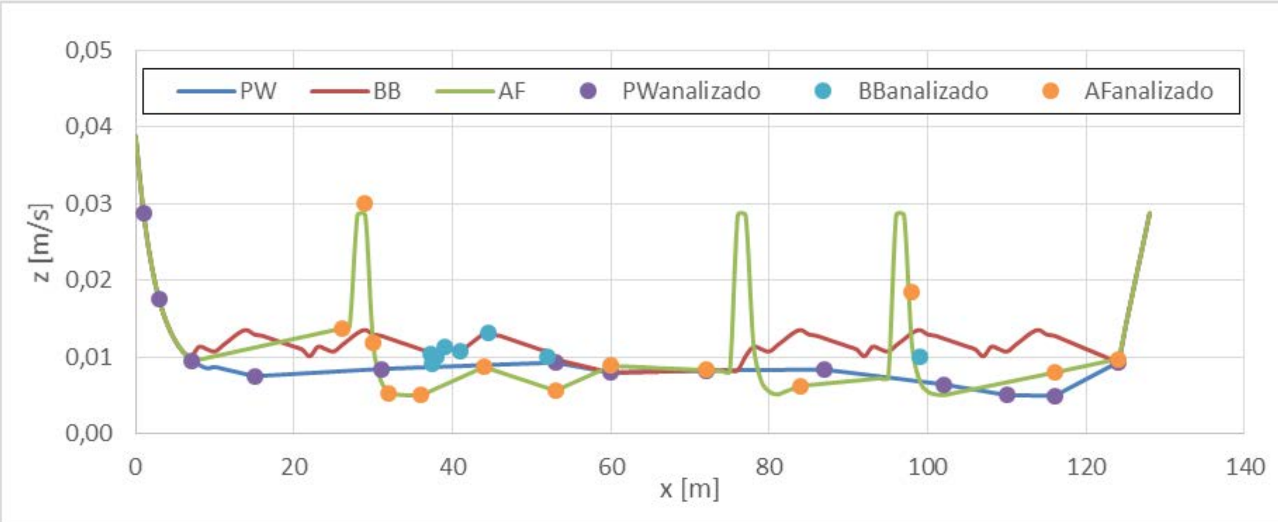

Figure 7. *Distribution of the mixing intensity throughout the reactor in the three cases analyzed. The symbols show the points where the sampling was performed.*

With the bubbling system there appears to be a global agitation effect, while the other two systems generate a much more localized vertical agitation. The paddle-wheel and the bubbling system require a functioning electromechanical equipment for their application, with the consequent increase in maintenance. Of these two, the energy consumption of the paddle-wheel is greater, although it provides an intense vertical mix that is maintained in a wider area of the raceway. Lastly, the SBTech blades hardly represent an energy consumption, which would only be due to the increase in friction caused by the blades on the flow. This increase was not appreciable with the paddle-wheel motor energy consumption measurement equipment that has been used. On the other hand, the SBTech blades have a much lower cost than the other two systems, although they produce a lower agitation peak than the paddle-wheel and of course require equipment that generates the main flow in the reactor.

Biomassgrowth

Another important aspect related to the growth of microalgae is the frequency with which intense vertical mixing occurs. The growth of most microalgae is maximum if the period of the light-dark cycles is less than 10 s. For this reason, it was to be expected that a stirring technique of the type that is achieved with several lines of blades, which produces intermittent areas of intense agitation for cultivation, would be more effective.

Figure 8 shows two images obtained in the Hydrodynamic Channel of the Fluid Mechanics Laboratory of the University of Seville, in which the trajectories of particles only dispersed by the turbulence generated by the bottom boundary layer are compared (in the absence of blades) with particle trajectories modified by a tubular vortex generated by an SBTech blade. It is observed how these blades are capable of moving flocs and microalgae from areas near the bottom towards the surface of the reactor, substantially reducing the period of the light-dark cycles.

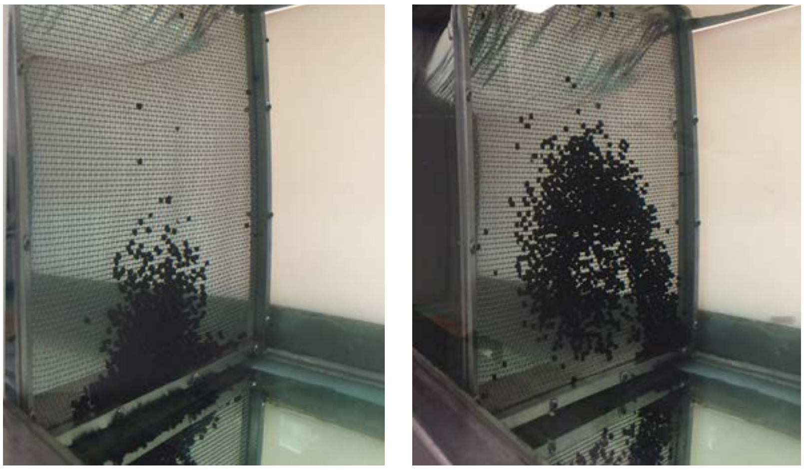

Figure 8. *Comparison of particle dispersion in a hydrodynamic channel without blades (left) and with SBTech blades (right).*

In Figure 9 it is observed that the system with the highest productivity is that of the SBTech blades, presumably motivated by a better distribution of the solar light in the reactor.

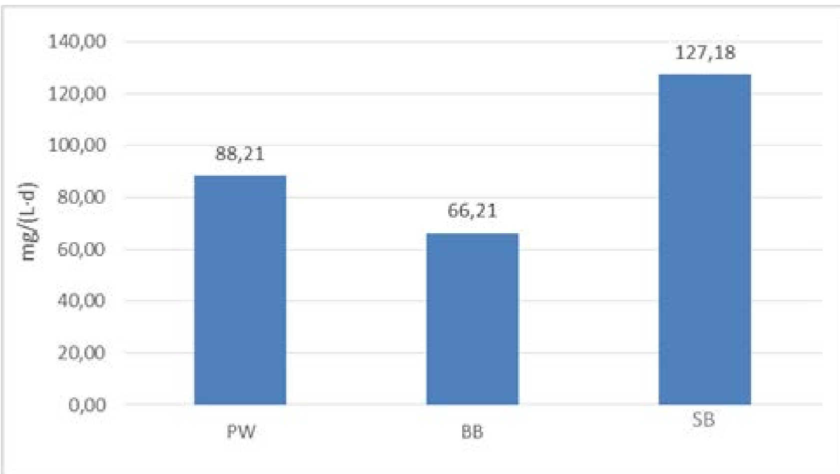

Figure 9. *Biomass productivity.*

pH

Table 3 shows the mean of the mean pH in each experiment and the mean of the difference between the daily maximum and minimum pH, a value that gives an idea of how active is the culture.

Table 3. *Mean daily mean pH and mean difference between maximum and minimum daily pH together with their confidence intervals (α = 0.05) and number of samples.*

	PW	BB	SB
pH_{med}	8,22 +/-0,11 (n=24)	8,75 +/-0,11 (n=42)	7,35 +/-0,08 (n=21)
pH_{max}-pH_{min}	1,07 +/-0,14 (n=24)	1,28 +/-0,09 (n=42)	0,46 +/-0,15 (n=21)

As can be seen, the difference between maximum and minimum with SBTech blades is 64% less than with the bubbling system, this being the one with the most activity of the three cases. The PW system is only 16.4% lower. In turn, the average pH value is distributed in the same way. That the pH is lower in the experiment with blades, it seems to indicate that the biomass has a composition of bacteria greater than in the other two cases.

Decantation

In the three cases, the decantation of the culture was studied and although in all cases very similar turbidity units were reached (around 50 NTU), not all of them started from the same starting concentration. Figure 10 shows how the culture decanted with the different agitation systems.

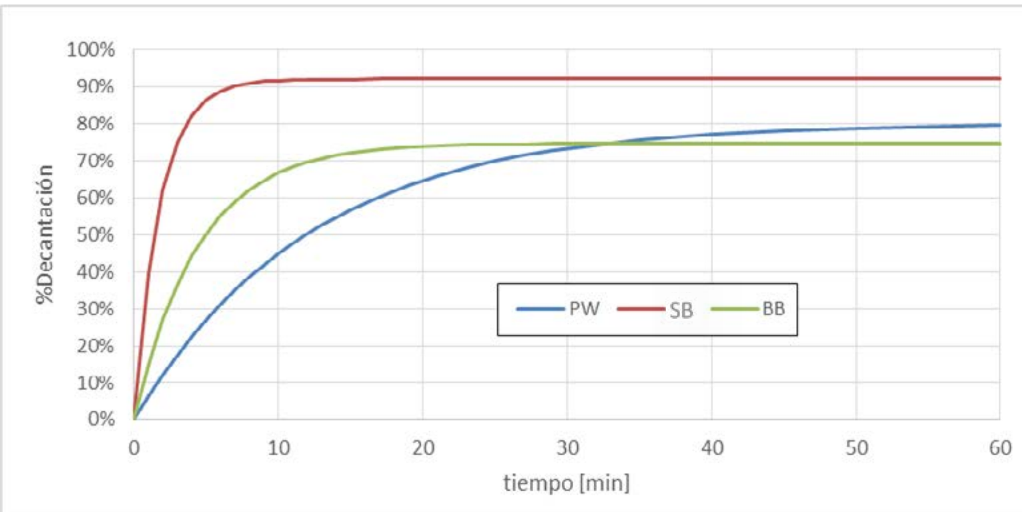

Figure 10. *Percentage of reduced turbidity of the reactor samples as a function of time, decanting in an Imhoff cone.*

The best result in terms of settling is obtained with the blades (SB), while the worst settling occurs in the case of the paddle-wheel (PW).

Figure 11. *Settling of the reactor content before and after the implantation of the SBTech blades.*

From the above data we can obtain the time it takes to reach an asymptotic value, the value of this asymptotic value and the corresponding turbidity value.

– PW: It takes 60 min to reach 80% settling, which corresponds to 55 NTU, starting from 270 NTU (or 575 mg/l).
– BB: It takes 27 min to reach 75% settling, which corresponds to 53.8 NTU, starting from 212 NTU (or 509 mg/l).
– SB: It takes 10 min to reach 92% settling, which corresponds to 46 NTU, starting from 517.84 NTU (or 823 mg/l).

CONCLUSSIONS

The tests carried out show that purification with microalgae is a very valid technique for the treatment of wastewater in small towns, highlighting as main advantages the lower energy consumption (around 75% lower than by purification with activated sludge) and emissions of CO_2 derived from energy consumption and bacterial activity, and a greater elimination of nutrients, at the same time that maintenance needs are substantially reduced due to the simplicity of the system's operation and by requiring fewer equipment with moving parts .

This study has been aimed at trying to improve the most unfavorable aspects of purification with microalgae, such as the productivity of the culture, which in large tanks is limited mainly by the availability of light from the microalgae and the decantability of the flocs. generated.

Three different stirring systems have been compared, with the aim of increasing vertical mixing in shallow tanks: a standard paddle wheel of this type of bioreactor (paddle-wheel), a bubbling system using perforated hoses and the system of SBTech® blades expressly designed for this application.

The stirring system that produces a greater increase in the intensity of vertical mixing than those that have been tested is the paddle-wheel, although this system is also the one with the highest energy consumption. The bubbling system produces agitation distributed throughout practically the entire reactor, although the intensity of the vertical mixing that is achieved is very limited. The SBTech blade-based stirring system is shown to be a promising technique due to its practically negligible energy consumption, solely derived from the friction that these blades introduce into the flow. In addition, its cost is very low and with this system it has been shown that a clearly greater biomass growth is achieved and the formation of algae-bacteria flocs is favored.

The sedimentation rate of the flocs produced with each of the systems has also been compared, showing that with the installation of the SBTech blades, the biomass generated decants in a much shorter time, with a reduction in settling time of more than 80%. compared to the system only mixed by a paddle wheel as a stirring technique for the entire reactor. In addition, with the SBTech system the final effluents present a higher quality from the point of view of the discharge requirements. The average yield of COD reduction has been 80.6%, the ammonium concentration in the effluent is practically zero and the elimination of total nitrogen and total phosphorus has been higher than 80% during most of the study period, reaching often 90% values. These data are very promising and most likely can be improved with better operation and control of the process.

ACKNOWLEDGEMENTS

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URBAN WASTEWATER FROM PRIMARY TREATMENT FOLLOWED BY MICROALGA CULTIVATION FOR CHLORELLAVULGARISBIOMASS PRODUCTION. PH INFLUENCE (SMALLWAT21)

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Abstract

The microalgae have many applications on aquaculture, biomedicine, human and animal feeding, wastewater treatment, etc. In this work, urban wastewater from primary treatment (UW-1) was used as culture media for *Chlorella vulgaris* growth. The pH value of the cultures was varied between 5 and 9, and the common operating conditions were agitation speed = 200 rpm, temperature = 25°C,aeration supply 0.5 L/min, continuous artificial light with illumination intensity equal to 359 μE m^{-2} s^{-1}.The experimental results from the culture at pH = 9 showed the following values of the maximum specific growth rate (0.0324 h^{-1}), volumetric biomass productivity (7.51 mg/(L h)), %$BOD_{5,removal}$ (92.1%), %$COD_{removal}$ (63.5%), %total $nitrogen_{removal}$ (94%) and total organic carbon removed (48.3%). The biochemical composition of the harvested algal biomass in the same culture determined the following percentages: %total pigments (chlorophylls and carotenoids) = 0.37%, %carbohydrates = 14.1%, %proteins = 79.8% and %lipids = 6.16%.

INTRODUCTION

Global population is estimated to be increased to nine billion people by 2050. The continuous population growth, urbanization, industrial development, and diversification of human activities has resulted in a water crisis that affects nowadays 4000 million people around the world which undergo water scarcity (Buonocore et al., 2016).

The generated wastewater has heterogeneous characteristicsdepending on its origin. Within the different types, urban wastewaters (UW) are generated by industrialised countries as a combination of liquid and solid residues from domestic, industrial, and commercial activities (Hodaifa et al., 2013). The main physicochemical characteristics of untreated UW are chemical oxygen demand (500–1200 mg/L), total nitrogen (30–100 mg/L), ammonium (20–75 mg/L), total phosphate (6 –25 mg/L) and high levels of suspended and volatile solids (between 250 – 600 mg/L and 200 – 480 mg/L, respectively), (Henze and Comeau, 2008). In addition, numerous pathogens microorganisms are commonly found in untreated UW, representing a major health risk for natural environments (Cai and Zhang, 2013).

In this work, primary urban wastewater at different pH-values (5, 6, 7, 8, 9, 10 and 11) were evaluated as culture medium of the green microalgae *Chlorella vulgaris* as new bioprocess for wastewater treatment.

METHODS

Microorganism used

The experiments were carried out with the microalga *Chlorella vulgaris* available in the research laboratories of the Chemical Engineering Area of the Pablo de Olavide University. The inoculum used was taken from a previous culture of *C. vulgaris* in wastewater from secondary treatment, specifically, from a photobioreactor with a working volume of 1L, where continuous artificial illumination was provided by means of white fluorescent light with a luminous intensity on the surface of the photoreactors of 359 µE/(m^2 s). The illumination is supplied from one side of the reactors (Hodaifa et al., 2012) at room temperature, with continuous air bubbling.

Experimental set-up and culture conditions

The photobioreactors used were made of glass with dimensions of 10cm inner diameter and 16cm high with a total volumetric capacity of 1 L (Fig. 1).

In the experiments, effluent from primary treatment of the urban wastewater treatment plant located in the province of Seville was used as a culture media. The crude urban wastewater was used directly without pre-treatment or non-sterilization conditions (Fig. 1).

(A) (B)

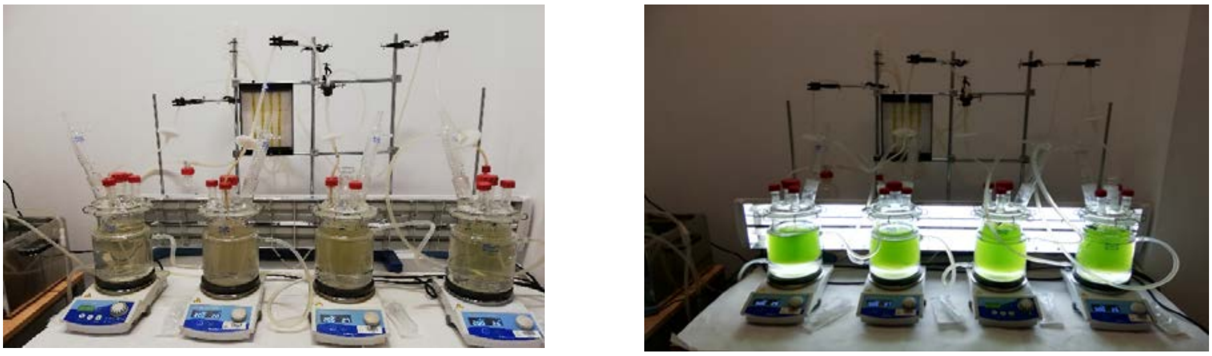

Figure 1. *C. vulgaris cultures inprimary urban wastewater:A) at time = 0 h and B) at the end of culture. Common operating conditions: pH = 9, aeration rate = 0.5 L/min, agitation speed = 200 rpm and T = 25 °C.*

During the experiments,the pH value was varied between 5 and 11 and the common operating conditions maintained were aeration rate = 0.5 L/min, agitation speed = 200 rpm and T = 25 °C. During the experiments, the pH of the culture media was adjusted by the addition of a 0.1 M NaOH and 0.1 M HCl solutions.

Microalgal growth measurement

Biomass concentration (x, g L^{-1}) was monitored over the course of the experiments. To that end, a volume of 5 ml of *C. vulgaris* suspension was centrifuged at 3000 rpm during 5 min to allow the separation of biomass and culture medium. Then, biomass was washed with ultrapure water three times. The measurement of the absorbance of the cell suspension was performed at 600 nm by using a UV-visible Spectrophotometer.

Biochemical composition of microalgal biomass

The harvested biomass obtained at the end of each experiment was characterized in terms of lipids, proteins, and carbohydrates. For this purpose, obtained biomass was separated by centrifugation at 3000 rpm for 5 min and washed three times with distilled water. After drying at 105 °C, total lipids was determined. Total lipids were extracted during 24 h in a micro-Soxhlet extractor with 50 ml of n-hexane as solvent. Then, n-hexane was removed, and samples were dried and weighed.

The crude protein content was calculated from the total nitrogen percentage according to the following equation: %Crude proteins = %TN x 6.25 (Becker, 2007). Total nitrogen was determined with a Total Carbon and Nitrogen Analyser provided by Skalar Company, mod. Formacs^{HT} and Formacs^{TN}.

The percentage of carbohydrates was determined in dry weight by the DNS method (dinitro-salicylic acid method) with previously sample sonication for half an hour to break the cell wall of the microalgae and dispose of most of the sugars inside (Miller, 1959).

The percentage of genetic material was considered equal to 1% as indicated by (Maaitah et al., 2020).

Analytical Methods

The following parameters were determined for urban wastewater: electric conductivity (EC), turbidity, chemical oxygen demand (COD), total phenolic compounds (TPCs), total carbon (TC), total organic carbon (TOC), total nitrogen (TN), inorganic carbon (IC), total iron, sulphate, and ammonium.

EC and turbidity were measured with a pH-meter Crison (mod. GLP 22C), a Conductimeter type Crison, mod. GLP31 and a Hanna Turbidimeter, mod. HI93703, respectively.

COD was determined photometrically at 620 nm according to German Standard methods (DIN 38409 H41, 1980).

TPCs were measured according to standard methods through their reaction with a derivative thiazol, resulting in a purple azo dye which was determined photometrically at 475 nm (DIN 38402 A51, 1986).

TC, TOC, IC, and TN concentrations (mg/L) were measured with a Total Carbon and Nitrogen Analyser provided by Skalar Company, mod. FormacsHT and FormacsTN.

RESULTS AND DISCUSSION

Table 1 shows the characterization of urban wastewater from primary treatment (UW-1) before and after microalga cultures for different pH values and control cultures (tap water and culture with UW-1 without microalgae).

The UW-1 characterized by high values of conductivity (1505 mS/cm), turbidity (69 FTU), BOD_5 (81.5 mg O_2/L), and COD (284 mg O_2/L) in comparison to tap water (Table 1).

Table 1. *Physicochemical composition of urban wastewater from primary treatment before and after algal cultures.*

Parameter	Control experiments		UW-1	UW treated by Chlorella vulgaris at different pH values					
	UW-1 aeration[1]	Tap water		5	6	7	9	10	11
pH	9.3	6.85	6.86	4.63	6.03	7.42	9.3	9.99	10.49
Conductivity, µS/cm	2010	0.00256	1505	1600	1780	1956	2000	1998	1990
Turbidity, FTU	9.71	1.19	69	7.7	5.59	15.48	5.57	6.07	5.93
COD, mgO_2/L	-[2]	0.00	283.8	192	297	204	104	125	3.57
BOD_5 (mgO_2/L)	0.01	ND[3]	81.5	5.34	6.84	6	6.42	6.94	0.67
Disolved O_2, mg O_2/L	7.8	8.2	3.07	8.01	8.11	7.315	8.28	8.11	7.65
Total solid, %	-	0.020	0.101	-	-	-	-	-	-
Organic matter, %	-	0.006	0.0368	-	-	-	-	-	-
Ash (%)	-	0.013	0.0639	-	-	-	-	-	-
TC (mg/L)	70.7	24.0	256	64.6	282	80.1	162.4	217	690
TOC (mg/L)	25.4	1.85	135	64.2	278	79.5	69.6	54.9	66.6
IC (mg/L)	45.2	22.1	121	0.47	4.39	0.59	92.8	162	624
NT (mg/L)	3.5	0.51	101	9.48	35.3	6.53	6.41	5.96	4.43
NN (mg/L)	0.0	0.15	0.00	0.00	0.00	0.00	0.00	0.00	0.00

[1]*UW-1 aeration: This culture simulated the culture with highest microalga growth at pH = 9 but without microalga inoculation and applied the same operating conditions: aeration rate = 0.5 mL/min during 469 h, agitation rate 200 rpm, artificial continuous illumination at intensity = at 395 mE cm^{-2} s^{-1} and temperature = 25 °C.*

[2]*-: Data not determined.*

[3]*ND: not detected.*

Fig. 2 shows the variation of the kinetic parameters of *C. vulgaris*versuspH values of the culture medium, which varied from 5to11.Both parameters, maximum specific growth rate (m_m, h^{-1}) and volumetric biomass productivity (P_b, g/(L h)), were increased with the augment of the pH values up to 9, then the values were decreased. The highest values of the maximum specific growth rate (0,0324 h^{-1}) and the volumetric biomass productivities (7.51 mg/ (L h)) were determined at pH = 9.

In Table 2, shows the biochemical composition of the harvested biomass after treatment and control culture. The carbohydrates fraction is the main fraction on the biomass generated (52.6%- 82.8%). The energetic compounds (carbohydrates and lipids) fraction was varied in the range of 65.6% to 88.1%. Meantime, lipid fraction represents the minority fraction and varied in the range of 5.45%-13.0%. On the other hand, the control experiment registered high lipid percentage up to 65.4% but the final biomass generation is less 3.13 times in term of TC.

After the microalga cultures, the values of the main quality parameters, oxygen dissolved, BOD_5, COD and total organic carbon (TOC) were improved from the point of view of treated water quality (Table 1).

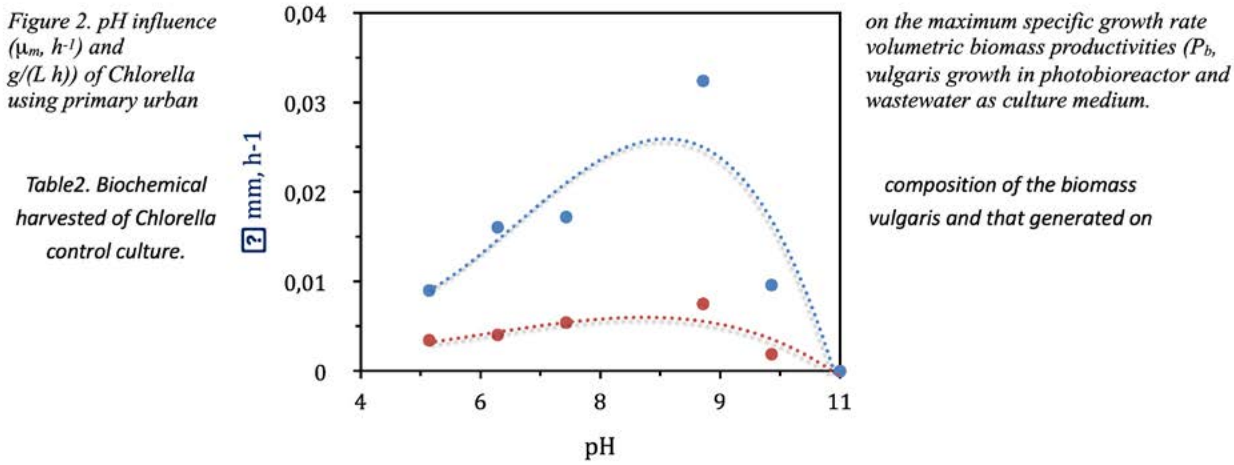

Figure 2. *pH influence on the maximum specific growth rate (m_m, h^{-1}) and volumetric biomass productivities (P_b, g/(L h)) of Chlorella vulgaris growth in photobioreactor and using primary urban wastewater as culture medium.*

Table2. *Biochemical composition of the biomass harvested of Chlorella vulgaris and that generated on control culture.*

C. vulgaris biomass	Parameter	Control experiment	*C. vulgaris* cultures at different pH-values					
			5	6	7	9	10	11
Biochemical composition	Total chlorophylls, %	0.40	0.05	0.08	0.03	0.27	0.25	0.00
	Carotenoids, %	0.13	0.02	0.21	0.02	0.10	0.09	0.00
	Crude Proteins, %*	8.39	20.5	13.5	8.13	10.9	21.9	17.8
	Carbohydrates, %	21.8	59.6	74.6	70.4	81.9	52.6	82.8
	Lipids, %	65.4	6.47	5.45	10.5	6.16	13.0	ND
Biomass in terms of carbon and nitrogen species	TC, mg/L	188	278	278	589	455	187	85.8
	TOC, mg/L	175	276	282	587	426	166	10.2
	IC, mg/L	13.4	1.25	4.39	2.07	28.4	21.2	75.6
	TN, mg/L	11.9	58.7	35.3	44.1	33.4	18.6	1.48
	IC, mg/L	13.4	1.25	4.39	2.07	28.4	21.2	75.6

Table 3 shows the removal percentages registered after microalgal cultures performed on primary urban wastewater. The TOC, IC, COD, TN, and BOD removal percentages were up to 68.2%, 99.6%, 98.7%, 95.6% and 99.2%, respectively. The negative values observed for COD and TOC are due to the bad biomass separation by centrifugation plus the cells breaks at the end of the cultures.

Table 3. *Removal percentages of TOC, IC, COD, TN, and BOD determinedafter C. vulgaris growth on primary urban wastewater.*

	% Removal after microalgal cultures at different pH					
Parameter	5	6	7	9	10	11
COD	32,5	-4,54	37,5	28,2	55,9	98,7
BOD	93,4	91,6	92,3	92,1	91,5	99,2
TOC	52,3	-106	68,2	48,3	59,3	50,6
IC	99,6	96,4	98,0	23,3	-33,7	-415
TN	90,6	65,2	92,4	93,7	94,1	95,6

CONCLUSIONS

As conclusions of this research work it can indicate the following points:

· The optimal maximum specific growth rate and volumetric biomass productivity were determined at pH = 9.

· The feasibility to use microalgae culture as secondary treatment by using UW-1 as culture media, which allows the reduction of sludge generated onthe conventional secondary treatment.

· The determination of higher removal percentages for the different water quality parameters after microalgal cultures, specifically for nitrogen removal with value up to 95.6%.

· The biochemical composition of the harvested biomass is rich on energetic compounds, which allows high conversion yields to biofuels.

· The final treated water quality after *C. vulgaris*growth complies with the current regulations established for direct discharges to waterways or for irrigation.

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TECHNO-ECONOMIC ANALYSIS OF MICROALGAE-BASED WASTEWATER TREATMENT IN SMALL POPULATIONS (SMALLWAT21)

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Abstract

A techno-economic analysis is carried out to determine the costs of different urban wastewater treatment processes in a small community (2000 p.e.). With the singularity that the secondary treatment is performedby phycoremediation with a bloom of microalgae. In this study, three projections were simulated, changing in each of them the hydraulic retention time (HRT) and solids retention time (SRT), tofind out the most economicnitrogen and phosphorus removal strategy for small populations. The times used in the three cases were 1) 5 days (HRT=SRT), 2) 20 days (HRT=SRT) and 3) 5 days HRT and20 days SRT. The costs obtained were 0.45, 0.73 and 0.43 €/m^3 of treated water, respectively.Although the third case achieves the lowest treatment cost, this scenario is not technically feasible, due to the high solid concentration required in the HRAP.

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INTRODUCTION

Eutrophication is one of the major problems causing loss of water quality due to excess nitrogen (N) and phosphorus (P)(Ansari et al., 2011). The main sources are agricultural runoff and, as a point source, urban wastewater(Ansari et al., 2011). Worldwide, 380 billion m3 of wastewater aregenerated each year(Qadir et al., 2020)and an estimated 80% of this is discharged untreated(WWAP, 2017).

In Europe, Directive 91/271/EEC regulates the collection, treatment and discharge of urban wastewater and wastewater from the agro-food industry. This directive stipulates that in populations above 10,000 population equivalents (p.e.)and which discharge into sensitive areas (prone to eutrophication), in addition to the removal of organic matter (COD and BOD5) and suspended solids (SS), N and P must also be removed. However, the Water Framework Directive (2000/60/EC) also stipulates in some cases that populations below 10,000 p.e. are obliged to remove N and P. For the latter, in contrast to large agglomerations, there is not economically viable and mature technology for the reduction of nutrients. This is because nutrient removal by conventional processes is expensive (2.5-8 €/kg N (Fux & Siegrist, 2018)and 2.5-14 €/kg P(Kroiss et al., 2008)) compared to removal of organic matter (0.32-1.13 €/kg BOD5(Pérez Sánchez & Egea Ruiz, 2014)). And that technologies typical of small populations rarely have high nutrient removal efficiencies. Also,small agglomerations do not benefit from the advantages of economies of scale, leading to higher and higher implementation and operating costs per inhabitant as the size of the agglomeration decreases(Ortega de Miguel et al., 2011). It is therefore of great importance to select the right technology.

A promising alternative is phycoremediation using microalgae/bacterial consortia, which achieves high nutrient removal efficiencies (72-100% (Delgadillo-Mirquez et al., 2016)). Open reactors “raceways” or HRAP (High Rate Algal Ponds), which are easy to operate and inexpensive(Kim, 2015), are used for this purpose. However, large-scale research is still needed to optimise the process, including cost analysis, as the cost of wastewater treatment with microalgae is poorly understood.This study performs a techno-economic analysis for different scenarios in a small population (2000 p.e.). The analysis aims to determine the best combination of phycoremediation, as well as to quantify and identify its advantages and disadvantages.

METHODS

A precise techno-economic evaluation was carried out for an urban wastewater treatment process (2000 p.e.) using a HRAP type reactor of microalgae in consortium with bacteria. The tool for the analysis was custom-built with the application of Microsoft Excel spreadsheets based on the study of (Ruiz et al., 2016)biorefinery and market exploitation for a 100 hectares facility in six locations. Our projections show a current cost per unit of dry biomass of 3.4 € kg−1 for microalgae cultivation in Spain (excluding biorefining products. The data as inputto support this analysis come from companies in the sector, databases, books, and articles.

Three scenarios were simulated, in each scenario different hydraulic residence times (HRT) and solids retention time (SRT) are used.In the first scenario, the time was set at 5 days, average time used in a HRAP(A. Shilton, 2005), (HRT=SRT) and follows the process shown in Figure 1-a. It was proposed to pump the purged sludge into the Imhoff tank to stabilise it. In the second case, the time was increased to 20 days (HRT=SRT), which follows the process shown in Figure 1-b.The sludge is sent to a gravity thickener because a stable sludge could be obtained in this process. Finally,in the third scenario, the time was decoupled, so that the HRT was set at 5 days and the SRT at 20 days (the process is shown in Figure 1-c). To achieve this SRT, it would be necessary to recirculate part of the biomass to the HRAP.

The flows in each step were calculated on the basis of the inlet flow, where a flow rate of 300 m^3/d, containing typical pollutant concentrations of 40 mg N/l, 8 mg P/l, COD 500 mg O_2/l, BOD_5 220 mg O_2/l and suspended solids 220 mg SS/l (Metcalf & Eddy, 2003). The assumed performance of each technology is shown in Table 1.Bacterial production in the HRAP was calculated based on a typical yield of 0.55 mgVSS/mgBOD_5(Metcalf & Eddy, 2003). Microalgae were considered to contain7-10% nitrogen and 1% phosphorus in its composition(Richmond, 2004). To comply with the discharge limits of Directive 91/271/EEC, we assumed a final concentration of nitrogen 15 mg N/l, phosphorus 2 mg P/l, BOD_5 25 mg O_2/l and suspended solids 35 mg/l.In this study it is assumed that microalgae can decant on their own.

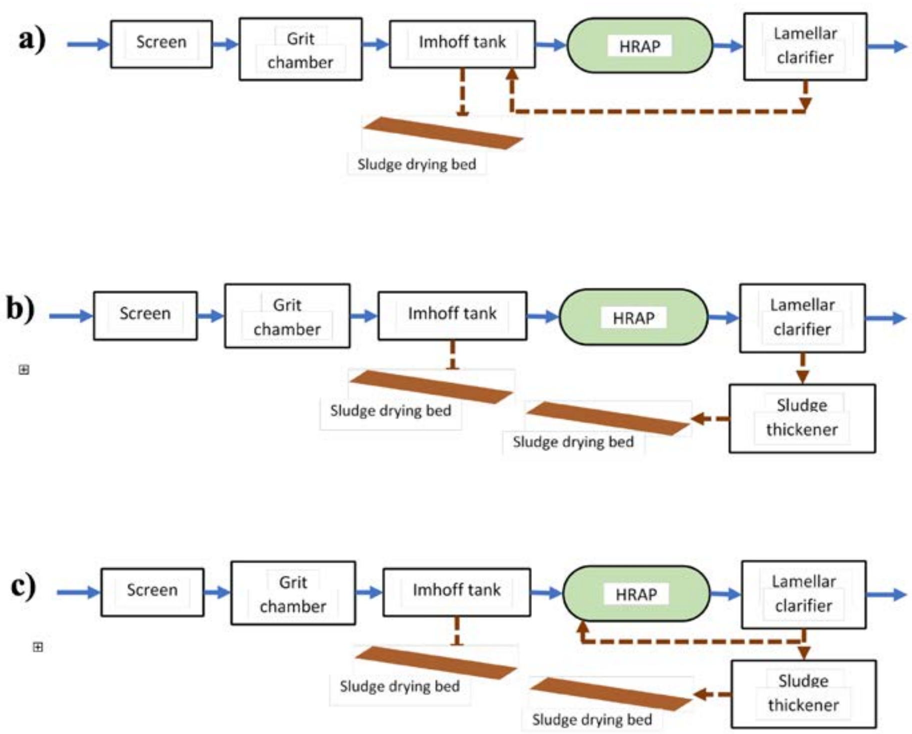

Figure 1: *Processes for wastewater treatment with microalgae.*
a) HRT=SRT= 5 days b) HRT=SRT= 20 days & c) HRT= 5 days, SRT= 20 days.

Table 1: *Removal performances of each technology (pre-treatment, primary treatment, and sludge line).*

Pre-treatment			
Treatment	% SS Removal	% BOD Removal	% Nutrient (N, P) Removal
Screen	Null	Null	Null
Grit	5	2.5	Null
Primary treatment			
Treatment	% SS Removal	% BOD Removal	% Nutrient (N, P) Removal
Imhoff tank	65	35	-
Sludge line			
Treatment	% SS Removal	% BOD Removal	% Nutrient (N, P) Removal
Gravity thickener	80	-	-
Sludge drying bed	94	-	-

RESULTS AND DISCUSSION

The projection for the first case (HRT=SRT= 5 days) shows that treating an annual flow of 102,095 m^3 per year would cost 0.45 €/m^3. 27% of thistotal cost iscapital costs and 73% corresponds to operating costs. In this scenario the energy consumption is 0.15 kWh/m^3 of treated water and the space requirements are around 2.5 m^2 per p.e. One remarkable issue in this scenario is the stability of the sludge, as 5 days of SRT is considered a young sludge with a high fermentation power.As a solution we proposed to stabilise it in the Imhoff tank (Figure 1-a). In this scenario the HRAP concentration of solids would be about 382 mg/l, which is reasonable according to literature (300-460 mg/l (Gutiérrez et al., 2016)since neither chemicals nor energy are needed. Indeed, bioflocculation may be promoted by recycling part of the harvested microalgal biomass to the photobioreactor in order to increase the predominance of rapidly settling microalgae species. The aim of the present study was to improve the recovery of microalgal biomass produced in wastewater treatment high rate algal ponds (HRAPs).

The second option (HRT=SRT= 20 days) arises from the mentioned poor sludge stability. Therefore, the times are increased based on typical values for the conventional prolonged aeration process (20-30 days (Metcalf & Eddy, 2003)), where stable sludge is obtained. In this case (Figure 1-b), treating 1 m^3 of water would result in a cost of 0.73 €/m^3, i.e., 64% greaterthan the first case. The reason for the increase is the higher land requirement, approximately 10 m^2/p.e., which means that a greater area of HRAP is needed. This also has an impact on the energy consumption, increasing by 3.5 times (0.52 kWh/m^3), mainly due to the power consumption of paddlewheels. In this scenario, a solids concentration of 402 mg/l would be required in the HRAP.

A strategy used by different authors is the decoupling of HRT and SRT(Torres-Franco et al., 2020)usually configured as high rate algal ponds (HRAP. This is the approach used in the latter case, where HRT=5days and SRT=20days. This requires recirculation of part of the biomass (Figure 1-c). The treatment cost for this scenario would be 0.43 €/m^3. Where 26% of the total cost is due tocapital costs and 74% to operating costs. In this scenario the energy consumption is 0.19 kWh/m^3 and the land requirements are 2.5 m^2/p.e. Although this scenario presents the lowest cost of the three cases, it could cause difficulties due to the high concentration of solids in the HRAP (1609 mg/l). It has the potential to inhibit the growth of microalgae and end up converting the reactor into a conventional activated sludge process, where microalgal activity is not relevant.

In all cases, the unitshowing the greatest influence on the total equipment costs is the secondary treatment (Figure 2 a). In terms of operating costs, in all cases “Other costs” is the most relevant, including this term the maintenance costs, operating supplies, contingencies and overheads (Figure 2 b).

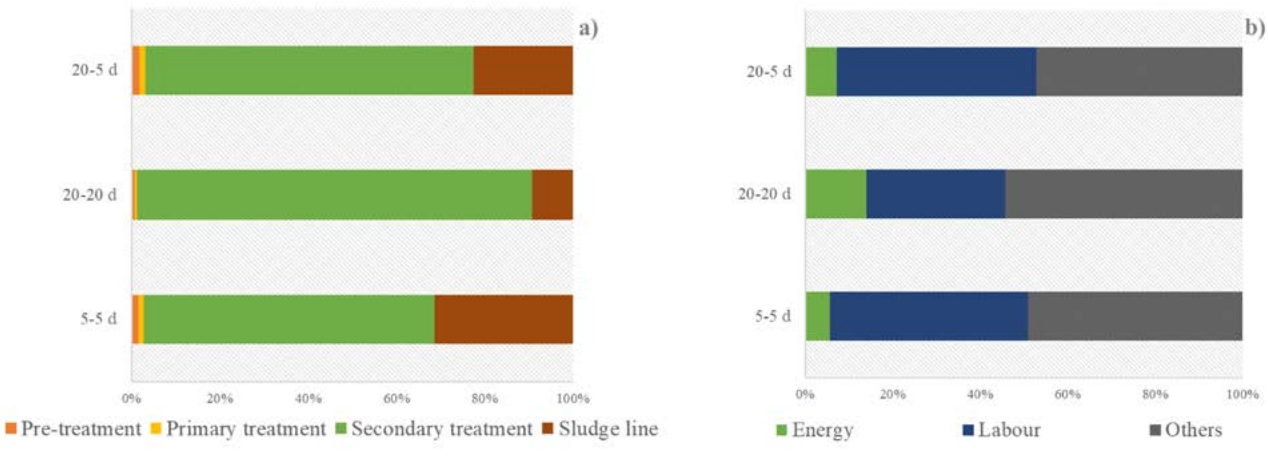

Figure 2: *Breakdown cost a) Equipment cost b) operation cost.*

CONCLUSIONS

The results of the wastewater treatment costs for the three projections simulated in this techno-economic analysis are competitive (0.45-0.73-0.43 €/m^3) when compared to the cost of conventional technologies in Europe (0.3-1€/m^3(UNEP, 2005)). Processes based on microalgae are much simpler and impose a low CAPEX. OPEX is also lower as maintenance is simple and does not require machinery and therefore lower energy consumption.

In addition, this process removes nitrogen and phosphorus without high costs. It is therefore a feasible solution for small populations, which have limited resources. However, the right compromise in operational conditions must be chosen. Since by working at different hydraulic and solids retention times, varying results can be obtained.

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MONITORING MICROALGAL BIOMASS GROWTH IN WASTEWATER IN LAB-SCALE EXPERIMENTS

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Abstract

Microalgal-based wastewater treatment is gaining attention as a way to achieve complete nutrient removal and reduce energy consumption in treatment processes. Valorisation of produced biomass should be aligned with the objectives of any wastewater treatment processes in order to increase sustainability. Correct estimation of microalgal growth has to be carried out when running R&D experiments in order to achieve these goals. The usual methods used for wastewater characterization (TSS, COD or BOD5) are neither suitable for routine laboratory work, where detailed time monitoring of a batch growth cycle is required, nor specific enough for microalgal biomass. Routine methods to monitor microalgal growth in synthetic media (optical density and sample dry weight) cannot be used with wastewater given the interferences of turbidity and suspended solids. In this work we have shown that monitoring the content of photosynthetic pigments is an indirect way to follow microalgal growth. This method is quick and uses small amounts of sample, making it appropriate for routine monitoring of a batch growth cycle with frequent periodical collection and analysis of samples. It is also specific for photosynthetic biomass. With this method, it is possible to estimate specific growth rates

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or duplication times of microalgal consortia growing in wastewater. The method has been applied to monitor microalgal growth in wastewater collected from two different stages of a WWTP: water after preliminary treatment and water after secondary treatment. Together with the results of the monitoring of other simple parameters, such as pH, an insight of biomass growth profiles and reasons for different behaviour dependent on wastewater composition can be understood.

INTRODUCTION

The use of microalgal-based wastewater treatment processes bears with it the promise of near complete removal of nutrients from domestic wastewater, while applying an extensive, low-cost treatment. Microalgal systems can be used in tertiary treatment to remove N and P. For that, suitable photobioreactors such as high-rate algal ponds, have to be installed in the treatment line, after secondary treatment (Abdel-Raouf *et al*., 2012). However, photobioreactor systems can also be used for secondary treatment, with consortia of photosynthetic microalgae and heterotrophic bacteria growing in symbiosis (Abdel-Raouf *et al*., 2012, Zhang *et al*., 2020). This solution has potential to cut the energy costs of the treatment process, because oxygen and carbon dioxide supplies are provided by the metabolic activities of these two groups of microorganisms.

Compared with conventional aerobic treatment, more complete nutrient removal is achieved via N and P assimilation into the extra microalgal biomass grown by photosynthesis. This mechanism of removal is advantageous, from a sustainability point of view, compared to conventional N removal via nitrification-denitrification cycles: Both the need for prolonged aeration for nitrification and the need for the presence of biodegradable organic matter in denitrification, involving forced costly changes in the oxidation state of nitrogen, are avoided (Abdel-Raouf *et al*., 2012, Shahid *et al*., 2020); Furthermore in microalgal treatment this element is fixed on the biomass, instead of being emitted as gaseous N_2 or traces of polluting nitrogen oxides. The biomass can be recovered (as waste sludge) and valorised in order to produce nitrogen-rich fertilizers and biomass-based products in dedicated biorefineries (Mata *et al*., 2010; Nagarajan *et al*., 2020; Shahid *et al*., 2020). This circular economy concept avoids production of significant amounts of synthetic fertilizers, as well as the associated carbon footprint and other negative environmental impacts (Mohan *et al*., 2019; Nagarajan *et al*., 2020). Such biomass valorisation processes have to be developed in tandem with microalgal-based wastewater treatment. Thus, it is important to get insights into the behaviour and characteristics of these biological systems in order to optimize all these goals.

Most often research and development starts at lab-scale, and there is a need for expedite methods to monitor the behaviour of biological systems. Assessment of microalgal growth is crucial. Expedite methods often used to monitor microalgal growth in synthetic growth media at lab scale are optical density (OD, absorbance of light of a given wavelength), or determination of solids dry weight of culture samples. Any of these methods readily yields results proportional to the content of biomass in the samples analysed. However, none of these methods can be applied when microalgae are grown in wastewater, because this has turbidity, colour and significant amounts of suspended solids, all of which interfere with the aforementioned analysis methods. As to the analytic methods commonly used for the characterization of wastewater (TSS, COD and BOD_5) they have as drawbacks long analysis time, large volume required, or lack of proportionality to the concentration of photosynthetic microalgal biomass alone.

As a consequence of the limitations of these methods, there is the need for a quick, reliable method to quantify microalgal growth in wastewater samples, suitable for routine laboratory work. Since these research activities usually involve several parallel assays with replicas in a single experiment, and the capacity of laboratory equipment is usually limited, such a method should be expeditious, reliable, require small volumes of sample, use current laboratory equipment and supplies, and yield results in a short time, so that it can contribute to decisions about the course of the experiments. In this study, we have assessed if the method of analysis for the quantification of photosynthetic pigments (Lichtenthaler, 1987) is suitable for this role.

METHODS

Wastewater was collected at the Faro Noroeste WWTP, located near the city of Faro, Portugal, where an extended aeration activated sludge process with UV disinfection is used to treat wastewater. Water collected after preliminary treatment (screening, degritting and degreasing), designated PT, and collected after secondary treatment, before disinfection, designated AD, was used as growth media for microalgal growth. Samples of wastewater were distributed in 1 L Erlenmeyer flasks containing 760 mL each. These were incubated in an orbital shaker at ~100 rpm and 22 ± 1 °C with continuous photoperiod irradiation by 18W fluorescent lamps (Phillips). The aspect of the wastewater samples and the experiment are shown in figure 1. PT water was incubated as such, with no inoculum, while AD water was inoculated with 6% (v/v) of PT water. Controls with tap water and non-inoculated AD water were also incubated. All experiments were carried out in triplicate for 28 days. A PAR intensity of 45.2 ± 3.4 $\mu mol.m^{-2}.s^{-1}$ (average ± standard deviation) was used, measured using a Skye Instruments SKE500 photometer, equipped with a SKE 510 sensor.
Periodically, samples were taken to analyse COD, TSS and photosynthetic pigments. pH was also monitored regularly. COD and TSS were analysed according to APHA/AWWA guidelines (Standard Methods, 1999). Photosynthetic pigments (chlorophyll A, chlorophyll B, total chlorophyll and total carotenoids) were determined according to the method of Lichtenthaler (1987).
Specific growth rates (*m*) were estimated as the slope of the linear part of the plot of ln *C* vs *t* with the concentrations of different photosynthetic pigments for each experiment. The duplica-

Figure 1. *Left picture: wastewater used for the experiments: PT wastewater to the left and AD wastewater to the right. Right picture: view of the experiment during incubation*

RESULTS AND DISCUSSION

Figure 2 shows the results of time monitoring of TSS and COD during batch microalgal growth in the different wastewaters tested.

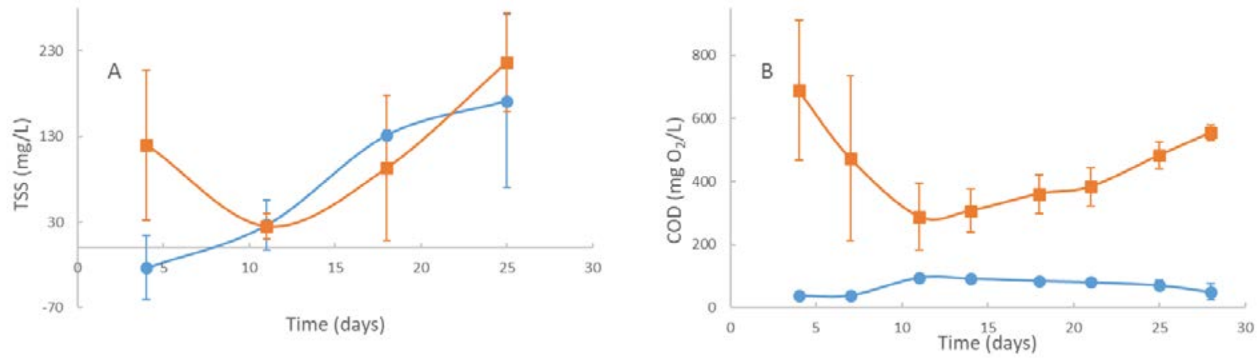

Figure 2. *Time monitoring of Total Suspended Solids (TSS) (A) and Chemical Oxygen Demand (COD) (B) of microalgal batch growth experiments in wastewater. Circles: AD wastewater; Squares: PT wastewater. Error bars represent the standard deviation of three replicates.*

Samples were collected only weekly to determine TSS, given the need for large volumes to have a reliable measurement of this parameter. As a result of this, only a few data points are available for each experiment, which does not allow processing to estimate growth kinetics parameters. Observing figure 2A, it can be seen that while in AD water a gradual increase is observed, for PT water the results are more difficult to interpret: there is an initial reduction of TSS, followed by a gradual increase after day 11. AD wastewater has a negligible TSS and biodegradable organic matter contents in the beginning of the incubation, thus photosynthetic microalgal activity is expected to dominate, and the gradual increase in TSS is a consequence of microalgal growth, much as is expected to happen during tertiary wastewater treatment. On the other hand, PT wastewater contains a significant amount of suspended solids, most of these consisting of biodegradable organic matter. This water is thus able to support heterotrophic metabolism of bacteria and microalgae present. The kinetics of heterotrophic metabolism is faster than autotrophic photosynthetic metabolism, which in turn is hampered by high wastewater turbidity. Thus, in the initial part of the incubation, consumption of particulate biodegradable organic matter occurs, mainly by heterotrophic bacteria, causing the observed decrease in TSS. This is the dominating mechanism in the first 11 days of incubation. After that, a gradual increase in TSS is observed. This corresponds to the overall increase in biomass content of the culture medium as the now dominating photosynthetic metabolism leads to microalgal growth, while particulate biodegradable organic matter content is already lowered. This combination of metabolisms is expected to occur during secondary treatment of wastewater in photobioreactors, with concurrent production of CO_2 by heterotrophs and of O_2 by photosynthetic autotrophs, requiring no external gas source. The COD profiles shown in figure 2B, for which more frequent sampling was possible, corroborate these deductions. Most of the COD corresponds to particulate matter (dissolved COD was nearly negligible for all samples, results not shown). For AD water there is a limited increase in COD on the first 11 days of the incubation, followed by a stabilization of the values. Given that particulate COD is absent from the initial water, this profile corresponds largely to the photosynthetic biomass grown during the incubation cycle. However, given the low values observed, it is not possible to calculate growth kinetics parameters from this profile. For PT wastewater the obtained profile is similar to that of figure 2A: a decrease during the first eleven days, corresponding to heterotrophic consumption of particulate organic matter, followed by an increase after that, corresponding to photosynthetic biomass growth. Again, a reliable determination of growth kinetics parameters is not possible with this set of data.

Figure 3 shows the results of time monitoring of photosynthetic pigments during batch microalgal growth in the different wastewaters tested.

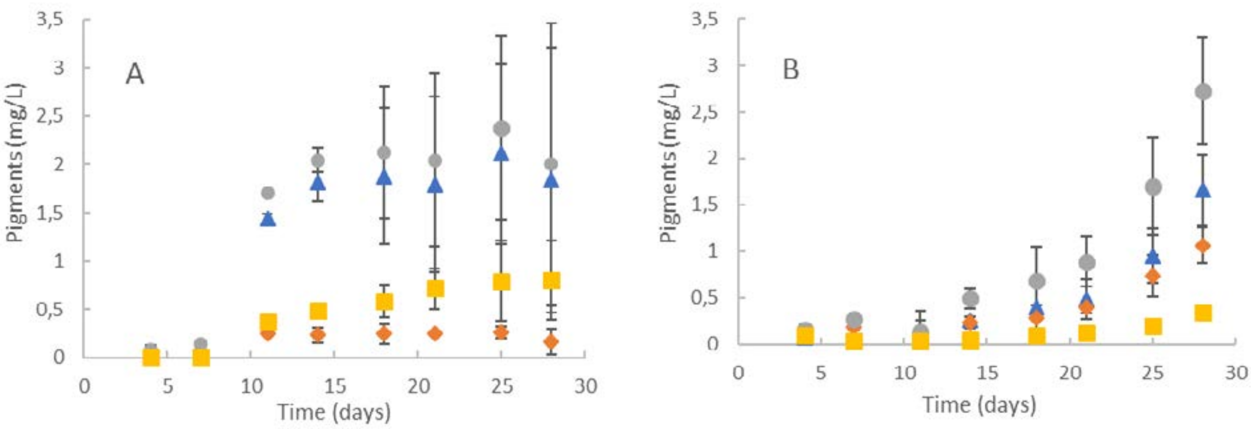

Figure 3. *Time monitoring of photosynthetic pigments during microalgal batch growth experiments in wastewater. (A) AD water (B) PT water. Triangles: Chlorophyll A; Diamonds: Chlorophyll B; Circles: Total Chlorophyll; Squares: total carotenoids. Error bars represent the standard deviation of three replicates.*

Even though there is significant dispersion between replicates it is possible to recognize typical profiles of batch microbial growth. The abundance of photosynthetic pigments can reasonably be considered proportional to microalgal biomass content during the growth cycle. Figure 3A shows that in AD water fast growth was limited to the first eleven days of incubation, followed by a stabilization phase that lasted to the end of the experiment. Given the near absence of turbidity in this water, fast photosynthetic growth is expected to occur from the very beginning of the incubation. However, the growth cycle was short. After 11 days the pH had risen to values above 11 (results not shown), indicating the carbon dioxide content of the growth medium had been depleted, and probably the depletion of nutrients (N and P) also limited microalgal growth. This agrees with conclusions taken with TSS and COD data. The growth profile observed in PT water (figure 3B) is quite different. Slow initial accumulation of photosynthetic pigments on the first 11 days of incubation is followed by an exponential growth observed until the end of the experiment. High initial water turbidity prevents vigorous photosynthetic growth, but once most of the particulate organic matter has been consumed the importance of this barrier decreases, and less obstructed microalgal growth proceeds. Furthermore, symbiosis between heterotrophic and photosynthetic metabolisms means that microalgal growth is not limited by depletion of CO_2. In this experiment the pH value ranged between 6 and 7, near neutral values (results not shown). In fact, it looks like microalgal growth would have continued if the experiment had been extended. Again, these observations agree with what had been observed with TSS and COD profiles.
A major difference is that with photosynthetic pigments data it is possible to estimate specific

growth rates of microalgal biomass. The results are shown in table 1.

Table 1. ***Estimated specific growth rates for microalgal growth***

Water	Chlorophyll A		Chlorophyll B		Total Chlorophyll		Carotenoids	
	μ (day^{-1})	R^2	μ (day^{-1})	R^2	μ (day^{-1})	R^2	μ (day^{-1})	R^2
AD	0.414 ± 0.084	0.923	0.252 ± 0.029	0.975	0.360 ± 0.070	0.929	0.477 ± 0.103	0.915
PT	0.117 ± 0.017	0.875	0.084 ± 0.018	0.759	0.100 ± 0.017	0.832	0.133 ± 0.005	0.993

The specific growth rates estimated with each pigment for the growth profiles in each water tested are in quite good agreement, with the exception of chlorophyll B. The specific growth rate for photosynthetic biomass is higher in AD water, where light penetration is easier, than in PT water, even though in this last case the exponential phase lasts much longer. Estimated average duplication times of photosynthetic biomass are 1.66 ± 0.34 day in AD water and 5.95 ± 0.67 day in PT water. These values are in reasonably good agreement with those obtained by Cabanelas *et al*. (2013) for microalgal growth in pre-treated and secondary treated wastewater.

CONCLUSIONS

The determination of photosynthetic pigments according to the method of Lichtenthaler (1987) is an expedite and reliable method to monitor microalgal growth in wastewater in lab scale. Only 1.5 mL samples are needed, and the results can be obtained within 4 hours after sampling. This makes it suitable for routine laboratory analysis of multiple replicate experiments of microalgal growth. Application of this method to the monitoring of microalgal growth in different wastewaters yielded results in accordance to the expected behaviour of microalgae in these media. By processing the raw photosynthetic pigment content data, it is possible to reliably estimate specific growth rates and duplication times. This is an invaluable tool for fundamental research on microalgal behaviour in wastewater.

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URBAN WASTEWATER TREATMENT BY CHLORELLA VULGARIS ON BUBBLE COLUMNS (SMALLWAT21)

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Abstract
Wastewater treatment plants are essential for social welfare. In this sense, urban wastewater treated by conventional plants either at the end of the plant or along the process (water from primary or secondary treatment) still contains nutrients that can be reused to produce high added-value algal biomass. In this work, *Chlorella vulgaris* grown on bubble columns with 4 L capacity using urban wastewater from secondary treatment as culture medium.Common operating conditions were pH of the culture medium = 9.0, ambient temperature,natural sunlightreceived inside the laboratory and the aeration flow was varied between 1 L/min to 5 L/min. The experimental results obtained indicate that the maximum specific growth rate varied between 0.00390 h^{-1} and 0.00942 h^{-1}. The volumetric biomass productivity has registered values between 0.0261 mg L^{-1} h^{-1} and 0.439 mg L^{-1} h^{-1}. The experimental optimum aeration level was obtained at 3 L/min with a maximum specific growth rate of 0.00943 h^{-1} and a volumetric biomass productivity of 0.439 mg L^{-1} h^{-1}. The biochemical composition of the biomass varied with the modification of the air flow rate supplied. The percentage of crude protein varied between 42.1% and 65.4%, the percentage of carbohydrates between 22.2% and 45.9% and the percentage of lipids between 5.1% and 9.0%. The removal percentages

of total organic carbon, COD, total nitrogen, and atmospheric-CO_2 have been 14% to 62.3%; 36.1% to 80%; 8.7 to 80.1 and 6.0% to 11.0%, respectively.

INTRODUCTION

Nowadays, the modern society is not perceived without the presence of wastewater treatment plants, which play a fundamental role in the preservation of the environment. Recently, wastewater treatment has begun to be seen to offer solutions not only in terms of bioremediation but also beyond that, especially in the generation of energy in the form of biogas. Throughout the urban wastewater treatment process, which normally consists of primary and secondary treatment, the treated wastewater still contains nutrients (organic and inorganic) that may be of interest for the growth of microalgae, opening the possibility of achieving the following objectives: i) Completion of urban wastewater treatment, ii) Generation of algal biomass rich in energy compounds and iii) The incorporation of carbon dioxide from the atmosphere and/or industry into the culture medium and thus reducing the greenhouse effect (Malvis et al., 2019). In this work, urban wastewater from secondary treatment has been used as a culture media to produce*Chlorella vulgaris* biomass in bubbling columns of 4 L capacity. The influence of the aeration rate (from 1 L/min to 5 L/min) on algal growth, the quality of the final treated water obtained, and the biochemical composition of the harvested biomass were studied.

METHODS

Microorganism used

The experiments were carried out with the microalga *Chlorella vulgaris* available in the research laboratories of the Chemical Engineering Area of the Pablo de Olavide University. The inoculum used was taken from a previous culture of *C. vulgaris* in wastewater from secondary treatment, specifically, from a photobioreactor with a working volume of 1L, where continuous artificial illumination was provided by means of white fluorescent light with a luminous intensity on the surface of the photoreactors of 359 $\mu E/(m^2\ s)$. The illumination is supplied from one side of the reactors (Hodaifa et al., 2012) at room temperature, with continuous air bubbling.

Experimental set-up and culture conditions

The columns used were made of acrylic material with dimensions of 7 cm inner diameter, 3 mm thick and 104 cm high with a total volumetric capacity of 4 L (Fig. 1).

In the experiments, effluent from secondary treatment of the urban wastewater treatment plant in the province of Seville was used as a culture media. The crude urban wastewater was used directly (without pre-treatment on non-sterilization conditions) as a culture media and once it was introduced into the bubbling columns (3.6 L work volume), the microalga was inoculated (Fig. 1).

Figure 1. *C. vulgaris cultures in bubbler columns operated at aeration rate = 3 L/min with secondary treatment effluent: A) at time = 0 h and B) at the end of culture.*

During the experiments, the aeration rate was varied between 1- and 5 L/min and the common operating conditions maintained were pH = 9.0, ambient temperature and natural sunlight. During the experiments, the pH of the culture media was adjusted by the addition of a 0.1 M NaOH and 0.1 M HCl solutions.

Microalgal growth measurement

Biomass concentration (x, g L^{-1}) was monitored over the course of the experiments. To that end, a volume of 5 ml of *C. vulgaris* suspension was centrifuged at 3000 rpm during 5 min to allow the separation of biomass and culture medium. Then, biomass was washed with ultrapure water three times. The measurement of the absorbance of the cell suspension was performed at 600 nm by using a UV-visible Spectrophotometer.

Biochemical composition of microalgal biomass

The harvested biomass obtained at the end of each experiment was characterized in terms of lipids, proteins, and carbohydrates. For this purpose, obtained biomass was separated by centrifugation at 3000 rpm for 5 min and washed three times with distilled water. After drying at 105 °C, total lipids wasdetermined. Total lipids were extracted during 24 h in a micro-Soxhlet extractor with 50 ml of n-hexane as solvent. Then, n-hexane was removed, and samples were dried and weighed.

The crude protein content was calculated from the total nitrogen percentage according to the following equation: %Crude proteins = %TN x 6.25 (Becker, 2007). Total nitrogen was determined with a Total Carbon and Nitrogen Analyser provided by Skalar Company, mod. Formacs^{HT} and Formacs^{TN}.

The percentage of carbohydrates was determined in dry weight by the DNS method (dinitro-salicylic acid method) with previously sample sonication for half an hour to break the cell wall of the microalgae and dispose of most of the sugars inside (Miller, 1959).

The percentage of genetic material was considered equal to 1% as indicated by (Maaitah et al., 2020).

Analytical Methods

The following parameters were determined for urban wastewater: electric conductivity (EC), turbidity, chemical oxygen demand (COD), total phenolic compounds (TPCs), total carbon (TC), total organic carbon (TOC), total nitrogen (TN), inorganic carbon (IC), total iron, sulphate, and ammonium.

EC and turbidity were measured with a pH-meter Crison (mod. GLP 22C), a Conductimeter-type Crison, mod. GLP31 and a Hanna Turbidimeter, mod. HI93703, respectively.

COD was determined photometrically at 620 nm according to German Standard methods (DIN 38409 H41, 1980).

TPCs were measured according to standard methods through their reaction with a derivative thiazol, resulting in a purple azo dye which was determined photometrically at 475 nm (DIN 38402 A51, 1986).

TC, TOC, IC and TN concentrations (mg/L) were measured with a Total Carbon and Nitrogen Analyser provided by Skalar Company, mod. FormacsHT and FormacsTN.

Total iron ions were determined according to the standard methods throughout their reduction to iron (II) ions in a thioglycolate medium with a derivative of triazine, resulting in a reddish-purple complex which was photometrically determined at 565 nm (ISO 8466-1, 1990; ISO 8466-1, 1990).

Sulphates and chloride were determined photometrically at 420 nm and 450 nm, respectively (ISO 8466-1; DIN 38402 A51).

The determination of K^+, Ca^{2+} and NH_4^+ concentrations were performed by using a selective ion electrode (Crison, mod. GLP 22C and Hach HQ 40d).

RESULTS AND DISCUSSION

Table 1 shows the characterization of urban wastewater before and after microalga cultures at different aeration flow rate.

Table 1. *Physicochemical composition of urban wastewater from secondary treatment before and after bioprocessing.*

Parameters	Initial	Microalga cultures on bubble columns (Air flow rate, L/min)			
		1	1.5	3	5
pH	7.80	9.21	9.08	9.19	10.08
Conductivity,µs/cm	1545	2000	2200	1760	1784
Turbidity, NTU	8.45	1.87	2.54	5.06	3.39
%Total solids, w/w	0.151	0.0022	0.0024	0.0020	0.0018
%Ash, w/w	0.120	0.0006	0.0009	0.0005	0.0005
%Organic matter, w/w	0.031	0.0016	0.0015	0.0015	0.0013
COD, mg O_2/L	76.8	60.8	57.2	32.2	17.9
Dissolved oxygen,mg O_2/L	4.61	7.53	8.15	8.27	8.19
Total organic carbon,mg/L	15.7	3.51	4.65	3.98	8.19
Total carbon,mg/L	99.7	109	123	125	86.7
Inorganic carbon,mg/L	84.0	105	119	121	76.8
Total nitrogen,mg/L	41.9	15.0	36.9	9.67	39.3
NO_2^-+NO_3^-,mg/L	5.75	31.3	27.3	4.94	26.2
K^+,mg/L	10.7	5.34	6.50	3.73	11.73
Ca^{2+},mg/L	26.4	35.1	28.51	1.79	3.27
NH_4^+,mg/L	25.2	0.0290	0.00400	0.0140	0.946
Cl^-, mg/L	178	163	160	108	73.7
Total-Fe,mg/L	0.644	0.0981	0.0403	0.117	0.0588
SO_4^{2-},mg/L	672	668	621	619	648

After the microalga cultures performed in bubbler columns using wastewater from secondary treatment as culture media, the values of organic matter (OM), COD and total organic carbon (TOC) were reduced, due to the assimilation of organic carbon as a source of carbon and energy by the microalga through aerobic respiration indicating a mixotrophic growth (photoautotrophic and heterotrophic) is carried out. Also, turbidity values decreased after bioprocessing in the bubbler columns.

Fig. 2 shows the variation of the kinetic parameters of *C. vulgaris* when the air flow rate is varied between 1- and 5 L/min. Both parameters were increased with the augment of the aeration flow rate up to 3 L/min, then the values were decreased.The highest values of the maximum specific growth rate (0.00942 h^{-1}) and the volumetric biomass productivities (0.000439 g/(L h)) were determined when the cultures supplied by 3 L/min of air (Fig. 2).

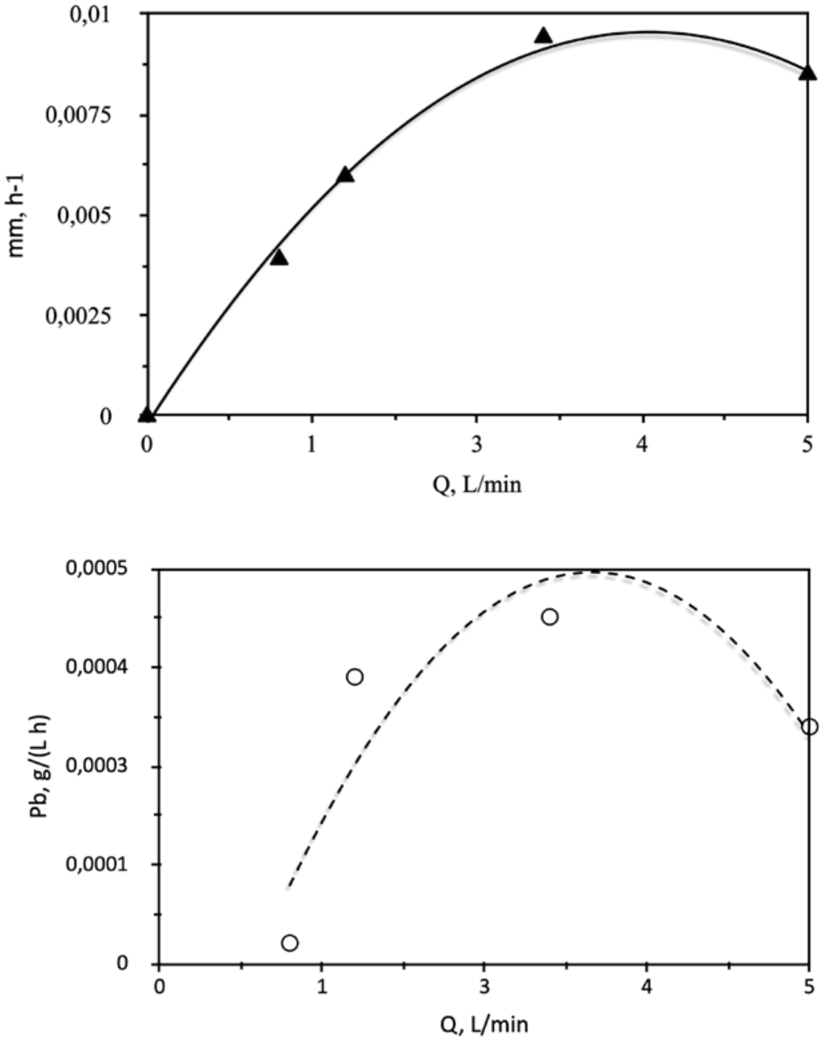

Figure 2. *Aeration influence on the maximum specific growth rate (m_m, h^{-1}) and volumetric biomass productivities (P_b, g/(L h)) of Chlorella vulgaris growth in bubbler columns and using secondary urban wastewater as culture medium.*

In Fig. 3 it can shows the biochemical composition of the harvested biomass. The proteins fraction is the main fraction on the biomass generated (42.2%-65.5%). The energetic compounds (carbohydrates and lipids) fraction was varied in the range of 31.1% to 51.0%. Meantime, lipid fraction represents the minority fraction and varied in the range of 5.1%-8.9%.

Regarding the final treated water, Table 1 shows the crude wastewater from secondary treatment and the treated water after each of the culture media performed. In all cases, it can be observed that the dissolved oxygen in the crude urban wastewater was 4.61 mg O_2/L and after algal culture this value substantially increased to higher than 7.5 mg O_2/L.

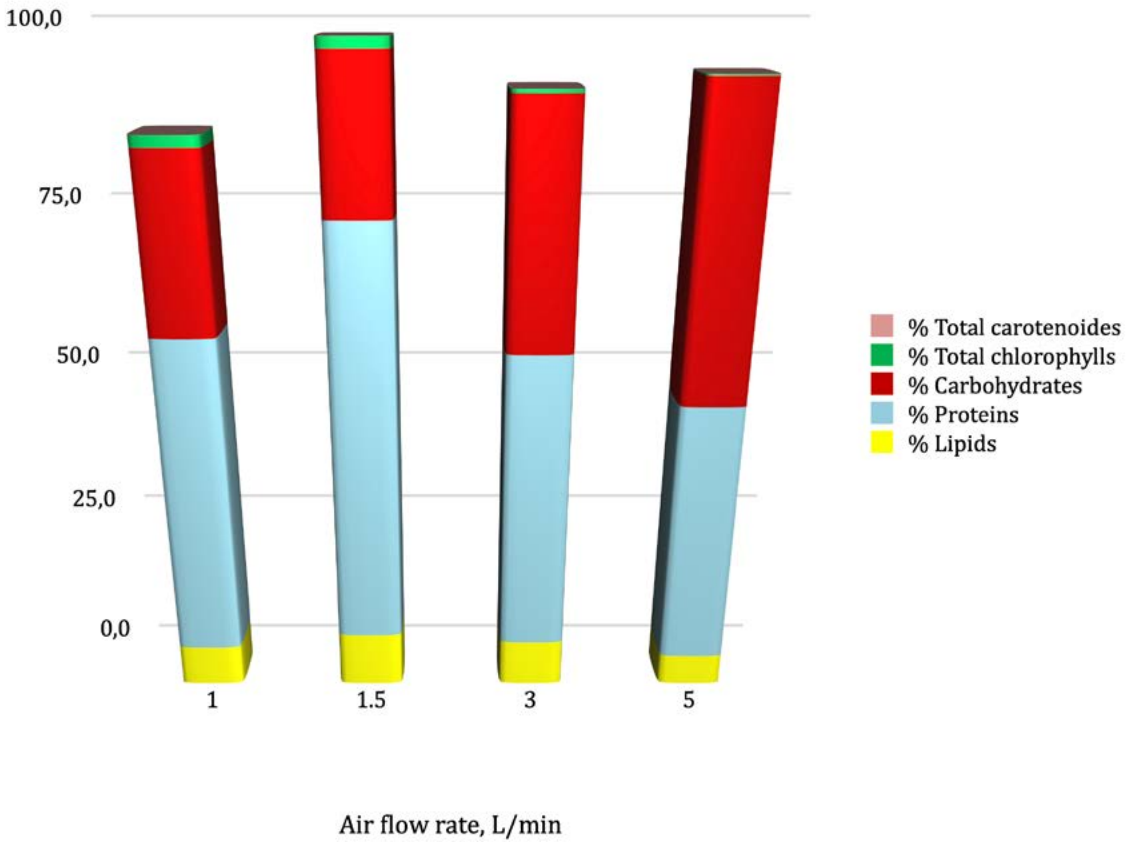

Figure 3. *Biochemical composition of the Chlorella vulgaris biomass harvested after cultures.*

Table 2 shows the removal percentages registered after microalgal cultures performed on secondary urban wastewater. The TOC, COD, TN, and atmospheric CO_2 removal percentages were up to 86%, 81.2%, 91% and 11.1%, respectively.

Table 2. *Removal percentages determined of TOC, COD, TN, and CO2 after C. vulgaris growth on secondary urban wastewater.*

Parameter	%Removal after microalgal cultures at different aeration rates, L/min			
	1	1.5	3	5
TOC	65.2	65.7	85.7	14.0
COD	5.56	33.3	81.2	73.7
TN	40.7	10.6	90.8	21.2
CO_2(atmospheric)	5.99	11.1	6.0	5.90

Considering the values of the different parameters determined in Table 1 and 2, the final treated water has enough quality to be discharge to the waterways or to reuse in irrigation.

CONCLUSIONS

As conclusions of this research work it can indicate the following points:

· The maximum specific growth rate was higher at an aeration rate equal to 3 L/min and the volumetric biomass productivity increasing with the aeration augment on microalgal cultures.

· The total nitrogen removal percentages values are higher than that registered for the COD values, which indicates that a mixotrophic culture is efficient in removing nitrogen from urban wastewater.

· The biochemical composition of the harvested microalga shows higher percentages on energy compounds (carbohydrates = 22.2% to 46.0% and lipids = 5.1% to 9.0%) and proteins (42.2%-65.5%). The final biomass could be used for biogas production or for its direct use for hot gases generation.

· The final water quality after treatment with *C. vulgaris* complies with the current regulations established for direct discharges to waterways or for irrigation.

· The incorporation of microalgae cultures into treatment plants can be an alternative of great economic and social interest.

Acknowledgements:

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PHYTO-TREATMENT OF AQUACULTURE EFFLUENTS USING MICROALGAE TECHNOLOGY (SMALLWAT21)

A.Perales. Pérez[1*] • J.A.Perales-Vargas Machuga[1], C.Garrido. Pérez[2].

Abstract

In this work, a purification treatment for the concentrated current of solids in an aquaculture farm dedicated to the cultivation of soles (*Solea Senegalensis*) with a recirculation aquaculture systems (RAS). The current was treated by a consortium of microalgae-bacterial. The mixotrophic microalgae-bacterial bloom was cultivated in 8-liter reactors and operated in discontinuous and semi-continuous mode to determine the kinetic modeling of organic matter and nutrients removal from the decanted water of this stream and the production of algal biomass. The culture showed a potential for dissolved nitrogen and phosphorus assimilation.

The algal biomass extracted when operating the reactors in semi-continuous was harvested by means of previous treatment of coagulation-flocculation. Some experiments of harvesting were carried out with two techniques: flotation and decantation. The decantation was the most viable harvesting technique because it used less quantity of coagulant and it obtained a greater biomass recovery efficiency. Furthermore, the biomass obtained was dehydrated by centrifugation and the water obtained

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in this process was filtered by means of a sand filter in order to obtain a quality effluent.
The algal biomass together with the concentrated sludge after the decantation of the concentrated current will be taken to an anaerobic digestion to obtain biogas, and the digestate can be used as a biofertilizer in agriculture. In this way, life cycle impacts of microalgae systems will be able to get environmental and economic benefits.

INTRODUCTION

Marine aquaculture has considerably increased its production growth in recent decades. Intensive aquaculture generates currents of concentrated solids corresponding to uneaten feed remains and fish faeces. These residues contain big amounts of organic matter and nutrients; species of nitrogen and phosphorus, which worsen water quality and produce eutrophication, facilitating the appearance of bacteria which endanger aquatic plants and animals. (Jasmin et al. 2020; Nie et al. 2020).

Another environmental pressure is the high water footprint aquaculture generates. The use of recirculation aquaculture systems (RAS) is more and more frequent due to the demand for water is much lower than conventional systems (Tanikawa et al. 2018). The mass flows of organic matter and nutrients generated in traditional systems are diluted in large volumes of water. However, these mass flows of pollutants are more concentrated in RAS. For this reason, the flows can be treated and utilized through some technologies. RAS consists of two streams. The mainstream or process water is screened to remove accumulated solids. Then, this water is subjected to a biological nitrification-denitrification process to remove nitrogenous compounds that are dissolved in the water (Zhang et al. 2020). These compounds, for example, total ammonia nitrogen is toxic to fish and it must be removed (Tanikawa et al. 2018). The next stage is disinfection to remove pathogens, e.g., bacteria of the genus Vibrio (Villar-Navarro et al. 2019). The water is free of solids, ammonium and bacteria and it is recirculated at the inlet of the farm and it is fed back into the culture systems. The second stream in the RAS is generated after the sieving of the initial current. This water has a high content of organic matter and nutrients in particulate form and it must be treated.

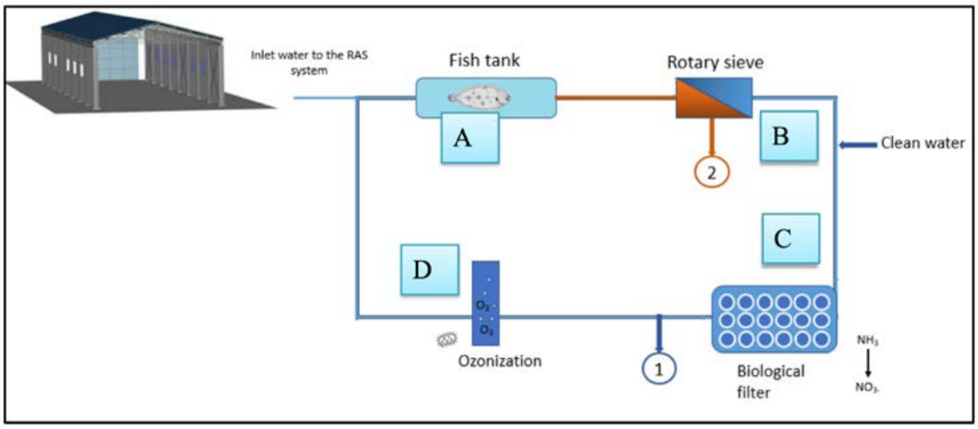
Microalgae-bacterial technology is used to purify the solids and nutrient-laden current generated in RAS (Marella et al. 2020; Milhazes-Cunha and Otero 2017; Zabed et al. 2020). One of the main challenges of microalgae technology is biomass harvesting due to the energy costs of the process (Morais Junior et al. 2020). Centrifugation is the most efficient technique for solids recovery and concentration, but this technique is not economically sustainable because of its high energy consumption. Currently, some studies have been conducted combining flotation or decantation technologies with a previous coagulation-flocculation. The purpose of this stage is to pre-concentrate algal biomass and reduce operational costs of centrifugation (Ferreira et al. 2020; Najjar and Abu-Shamleh 2020).

METHODS

This research work has been carried out with the current waters of an aquaculture farm, which is located in the natural park of the Bay of Cadiz. The farm produces 500 tons of sole (*Solea Senegalensis*) per year. The plant has three main facilities: hatchery, nursery, and fattening. A RAS has been implemented in the last facility. The current flowing out of the fattening ponds (A) is loaded with feed not ingested and fish faeces. This current flows through a rotary sieve (B) generating stream number 2, in which solids and nutrients are concentrated, and stream number 1, corresponding to screened water, which circulates through the system and is treated by a biological filter (C). Part of this water is purged and replaced by clean water. Then, the water is disinfected with ozone (D) and part of the organic matter is removed. Finally, this stream is returned to the system.

1.Current

work



Figure

treatment.

This

research

Figure 1. ***Current treatment.***

This research work proposes an alternative treatment to stream number 2 (Figure 2). The treatment consists of subjecting the decanted water from the decanter (from stream 2, Figure 1) to a biological treatment based on microalgae technology in consortium with bacteria, in which microalgae assimilate the nutrients in the water. Microalgae generate organic matter from light and inorganic carbon through their autotrophic metabolism. Photosynthesis produces O_2, which is used by bacteria in their heterotrophic metabolism to degrade organic matter present in the water. In this process of biological elimination of organic matter, bacteria generate CO_2, which is assimilated by microalgae for photosynthesis.

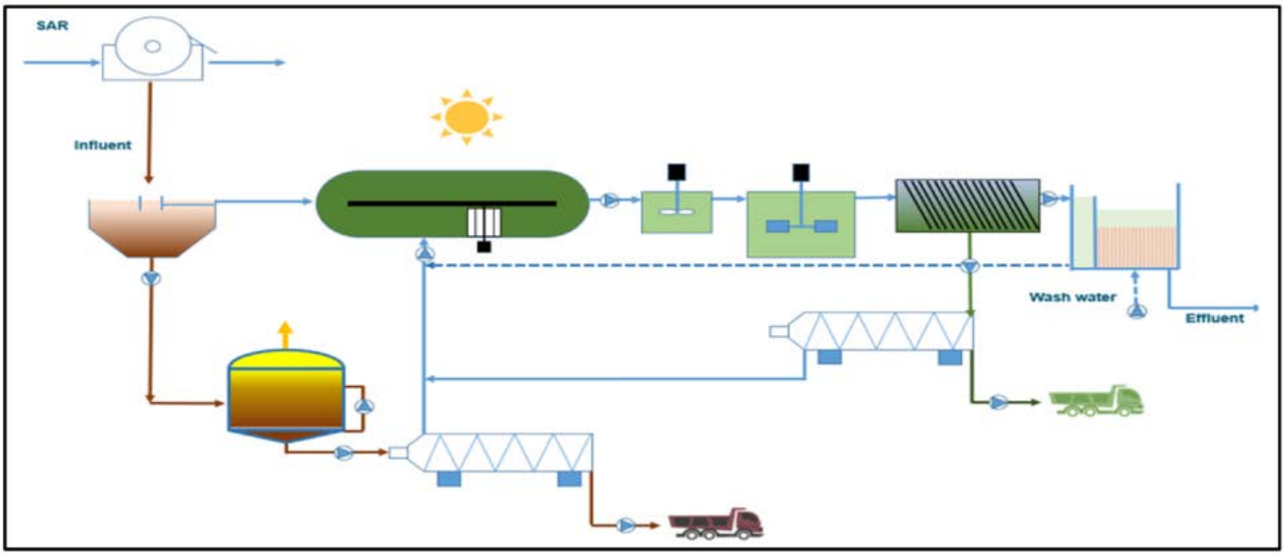


Figure 2. *Alternative treatment.*

Some experiments were conducted to study the treatment of the solids and nutrient-laden stream from a RAS in the farm studied:

1. Decanting of the stream number 2 for 30 minutes using imhoff cones and sedimentation column to determine the upward velocity of the particles that constitute the sludge and analysis of total suspended solids (TSS) and volatile suspended solids (VSS) of sludge and decanted water.
2. Mixotrophic culture of microalgae bacteria in 8-liter reactors operating discontinuously and semi-continuously. The Verhulst logistic equation was used to determine the biomass production in Batch mode (discontinuously). Experimental data of biomass concentration during Batch operation were modeled by Verhulst (Verhulst 1844) and compared with the values predicted by this model. The maximum productivity reached during this stage occurs at the midpoint of the exponential growth stage. The parameters of initial concentration (Xo), maximum concentration reached (Xmax) and specific growth rate (μ) calculated by modeling were used to know the theoretical initial (xi) and final (xe) concentration of the reactors operated semi-continuously through 10 feeding cycles. This operating mode reaches the maximum biomass productivity in the reactors.
3. The volume extracted in the feeding cycles was subjected to Jar test assays with increasing doses of $FeCl_3 \cdot H_2O$. Subsequently, some flotations and decantations tests were performed to separate the biomass from the supernatant water.
4. After selecting the best harvesting technique (decantation or flotation) and determining the optimal reagent dose, the biomass obtained was centrifuged for 5 minutes at 3220 rpm and supernatant water was filtered after the harvesting stage through a sand filter.

RESULTS AND DISCUSSION

9.22 ± 1.74 mL/L settleable solids were obtained after settling and the solids recovery was 52%. The ascensional velocity in which sedimentation of more than 90% of the particles constituting the sludge occurred was 2.8 cm/h. The percentage solids recovery and the VSS fraction of the decanted sludge being 73% were used to estimate the biogas production that would be produced in an anaerobic digester. In this stage, 40% of the VSS is reduced and the estimated biogas flow obtained is 108 m^3/d. This biogas would generate 220 kWh/d of electricity per day.

The experimental data of biomass concentration presented a good compromise for the theoretical results of the Verhulst model, the values of the regression coefficients were close to 1. The maximum productivities reached in the 3 reactors were similar.

Table 1. *Parameters obtained from Batch operation of 3 reactors.*

Reactor	X_0 (mgSS L^{-1})	X_{max} (mgSS L^{-1})	μ (d^{-1})	P_{max} (mgSS L^{-1} d^{-1})	R^2
1	43.6	179.5	0.7	31.4	0.999
2	32.1	180.7	0.9	39.4	0.986
3	40.5	194.2	0.6	29.2	0.997
Media	38.7	184.8	0.7	33.3	0.994

The average value of the initial concentration in the reactors during the semi-continuous operation was similar to the theoretical value. However, the biomass concentration values reached were higher than the predicted values, for this reason, the obtained productivities were almost twice the theoretical productivity and the hydraulic residence time (THR) was lower than expected, so the process was faster.

Table 2. *Comparison of theorical and experimental average values.*

	Xo (mgSS/L)	Xe (mgSS/L)	P_{max} (mgSS/L d)	μ (1/d)	THR (d)
Theorical	75.9	108.9	32.9	0.7	3.4
Experimental	77.9 ± 2.2	140.4 ± 12.2	62.5 ± 12.3	0.8 ± 0.1	2.6 ± 0.3

Moreover, samples taken from the extracted volume at the end of the feed cycle were analyzed for initial and final concentrations of total phosphorus (TP), total nitrogen (TN), and total organic carbon (TOC). The removal performance of these contaminants were 98%, 87%, and 52%, respectively.

The previously coagulated-flocculated biomass obtained a recovery percentage of 93% with a coagulant dose of 60 mg/L in the decantation while in the flotation a recovery percentage of 86% was obtained with a coagulant dose of 80mg/L.

The percent solids capture after centrifugation of the biomass was 88% and the total suspended solids (TSS) concentration of the supernatant water was 15 mg/L. In the sand filter, the bulk density of the retained solids was 1.32 kg/m^3. Besides, the filter efficiency was 42% and TSS concentration of the effluent was 35 mg/L.

CONCLUSIONS

An alternative treatment for the solids-laden stream of a RAS has been evaluated:

The conclusions obtained from the results obtained are:

– The sludge in the stream has good sedimentability and a yield of 52%. This sludge has the potential to be treated by anaerobic digestion. The generation of biogas can satisfy the energy demand of the aquaculture facility.
– The mixotrophic microalgae-bacteria culture presents good nutrient elimination yields (phosphorus and nitrogen) in addition to eliminating part of the organic matter present in the stream.
– The coagulation-flocculation-decantation technique has been selected as the most optimal because this technique uses less coagulant concentration, higher percentage recovery, and energy costs are lower because flotation needs a tank to pressurize water.
– Centrifugation is an efficient technique as a final stage of biomass harvesting. The chemical composition of this biomass will be analyzed in future studies for its potential as a biofertilizer.
– Filtration of the selected supernatant water generates an effluent with 35 mg/L of TSS. This concentration is the maximum admissible in urban wastewater treatment regulations.

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ASSESSMENT OF THE TREATABILITY OF A STEEL MILL EFFLUENT USING MICROALGAE BIOTECHNOLOGY

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Abstract

In the present study, the treatability of a steel mill effluent with microalgae technology is evaluated. This effluent is characterized by a high nitrate load, the presence of heavy metals and a low organic matter load, which makes it a very difficult effluent to treat. To this end, a two-stage test was carried out: in the first stage, a synthetic water was created that imitated the effluent, in order to have an approximation of the behavior of the microalgae in the medium. In a second stage, screening tests were carried out with the real effluent (effluents from two different collectors, C and D), to determine which macro and micronutrients are essential for the growth of the algae in this medium. The species used was *Scenedesmus obliquus.*

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INTRODUCTION

Recent worldwide interest in the cultivation of microalgae for energy purposes using inexpensive culture media, coupled with the need for more environmentally sustainable wastewater treatment technologies, has made wastewater treatment using microalgae a promising alternative from an economic and environmental point of view. Microalgae are autotrophic organisms that fix solar energy into chemical energy through the process of photosynthesis. They convert inorganic elements of C, N, P... into organic compounds of different complexity. Water treatment systems using microalgae technology are based on this process considering that the inorganic elements are provided by the wastewater, which in this technology are considered as a culture medium for these microorganisms. In this way, nutrients that are considered pollutants are "eliminated" from the water, and these nutrients are transformed into a "sludge" or highly valuable microalgae biomass (energy production, biofertilizers, etc.) (Mallick 2002).

It is known that microalgae have a great potential to remove nitrogen and phosphorus from wastewater to levels below the limit levels imposed by environmental regulations (Mennaa, Arbib, and Perales 2015). Another environmental benefit offered by the use of microalgae is the capture of CO2 as it is the main source of inorganic carbon that they assimilate for photosynthesis, increasing biomass productivity (Jiang et al. 2011) (Tang et al. 2011), and reducing the carbon footprint of the purification process.

Microalgae have been found to be effective in removing some heavy metals from industrial waters (Anastopoulos and Kyzas 2015), reaching good removal efficiencies for some metals, such as more than 60% for zinc (Ferraro et al. 2018). However, and although the scientific literature provides data on the ability of microalgae to assimilate metals, no published work has been found on microalgae purification with effluents that have such a complex mixture of heavy metals and at concentrations typical of steel mill effluents. Moreover, there are many published papers showing that at certain concentrations, some heavy metals can be toxic to microalgae. Also, complex mixtures of these metals can synergistically affect toxicity in these organisms by increasing the effects beyond those expected with each compound individually (US 2002).

In summary, in terms of nitrogen, steel mill wastewater can provide one of the fundamental nutrients for the growth of microalgae, although it is provided in very high concentrations and much higher than those studied in other effluents or culture media. It is necessary to advance in the effect that these high concentrations can have on microalgae. Likewise, the presence of metals that can generate toxicity in microalgae can jeopardize the applicability of these treatments. Nevertheless, and although there are several handicaps, it is important to advance in the study of the treatability of these effluents with novel technologies such as microalgae technologies.

METHODS

The algae selected for this study was the species *Scenedesmus obliquus*, as it is a resistant algae suitable for wastewater treatment. To create the synthetic water, the public data of the company was consulted, and the majority of its elements (N, P, Zn, Cu, Ni and Cr) were considered. The experimental design with this water consisted of 5 one-litre reactors, 4 with synthetic wastewater and a control reactor with distilled water and nutrient solution (COMBO medium): ACE reactor (synthetic water), ACE++ reactor (ACE reactor with COMBO nutrient medium), ACE_1/10 reactor (ACE reactor in 1:10 dilution), ACE_1/10++ reactor (ACE++ reactor in 1:10 dilution). A batch test was carried out, where algae growth and N and heavy metals removal were measured.

As for the actual effluent, four screening tests were designed and carried out in two stages. In the first stage, work was carried out with undiluted effluent, testing different additions of macro and micronutrients using the elements and concentrations of the Combo medium as a basis, giving a total of ten combinations. In the second screening stage, the experimental design was repeated, but with the effluents diluted in a 1/10 ratio. In this case, glass tubes of 16 mm diameter and 10-12 ml capacity were used, especially for the development of toxicity microassays with microalgae. *Table 1* shows the planning of the test.

Table 1. *Screening test.*

Nº	Name	Distilled water	Collector (C o D)	N	P	Macro_I	Macro_II	Micro_I	Micro_II	Vitamins
1	CONTROL	Yes	No	Yes	Yes	Yes	Yes	Yes	Yes	Yes
2	A	No	Yes	No	No	No	No	No	No	No
3	B	No	Yes	No	Yes	No	No	No	No	No
4	C	No	Yes	No	No	Yes	Yes	Yes	Yes	Yes
5	D	No	Yes	No	Yes	No	Yes	Yes	Yes	Yes
6	E	No	Yes	No	Yes	Yes	No	Yes	Yes	Yes
7	F	No	Yes	No	Yes	Yes	Yes	No	Yes	Yes
8	G	No	Yes	No	Yes	Yes	Yes	Yes	No	Yes
9	H	No	Yes	No	Yes	Yes	Yes	Yes	Yes	No
10	I	No	Yes	No	Yes	Yes	Yes	Yes	Yes	Yes

RESULTS AND DISCUSSION

Tests with synthetic water.

Figure 1 shows typical microalgae biomass growth curves in terms of suspended solids during the batch stage.

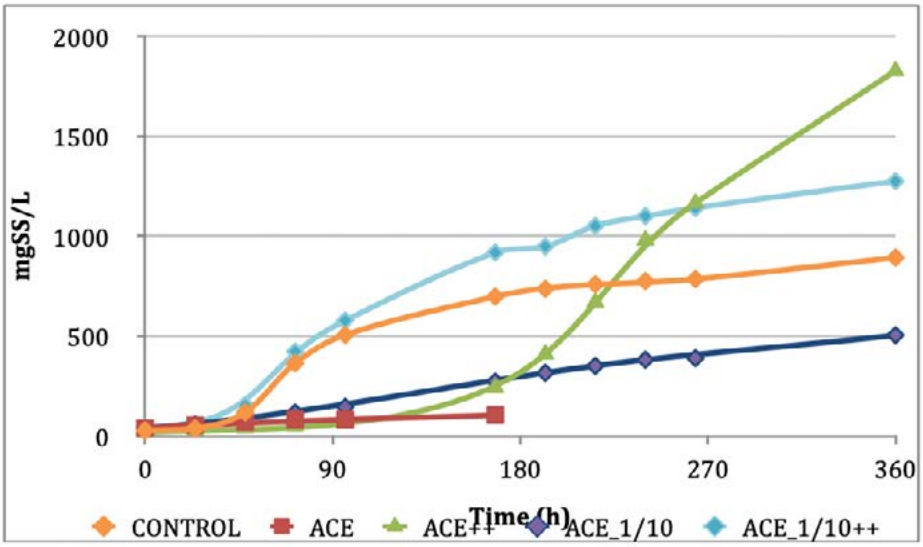

Figure 1. *Evolution of biomass.*

Table 2 shows the removal of metals in each reactor during the test.

Table 2. *Heavy metals removal.*

Aqueous samples (mg/L)			
Heavy Metal	**Reactor**	**Batch start**	**End of batch**
Cu	**ACE**	0,022	0,0242
	ACE++	0,022	0,0183
	ACE_1/10	0,0041	0,003
	ACE_1/10++	0,0037	0,00104
Cr	**ACE**	0,0262	0,019
	ACE++	0,26	0,0302
	ACE_1/10	0,0032	0,0016
	ACE_1/10++	0,00311	0,0012
Ni	**ACE**	0,0538	0,062
	ACE++	0,0535	0,0842
	ACE_1/10	0,0068	0,00935
	ACE_1/10++	0,00714	0,0062
Zn	**ACE**	0,242	0,205
	ACE++	0,239	0,223
	ACE_1/10	0,109	0,297
	ACE_1/10++	0,146	0,088

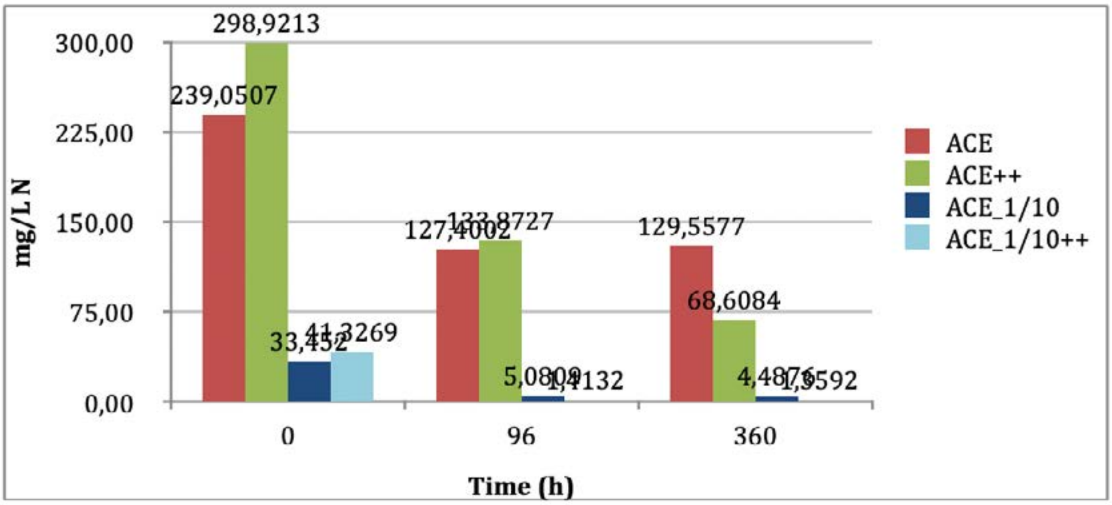

Figure 2, *shows the nitrogen removal during batch test.*

Figure 2. Nitrogen removal.

The results obtained in this first stage show that those reactors with a nutrient medium supply are those with the best growth and elimination of nitrogen and heavy metals. The microalgae are capable of growing in this medium, but as it is a medium with a low organic matter load, they need a nutrient supply. Even so, they show a marked phase of latency derived from the need to adapt to the environment due to the complex mixture of metals, minerals, and other compounds.

<u>Real effluent test</u>

Figures 4, 5, 6 y 7, show the results obtained in the test with the real effluent.

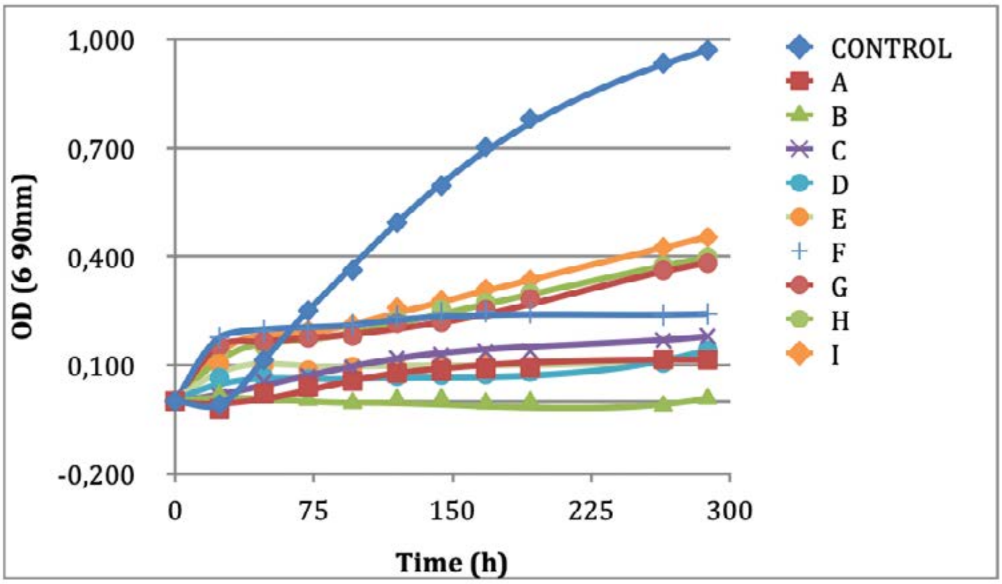

Figure 3. *Test with collector C effluent.*

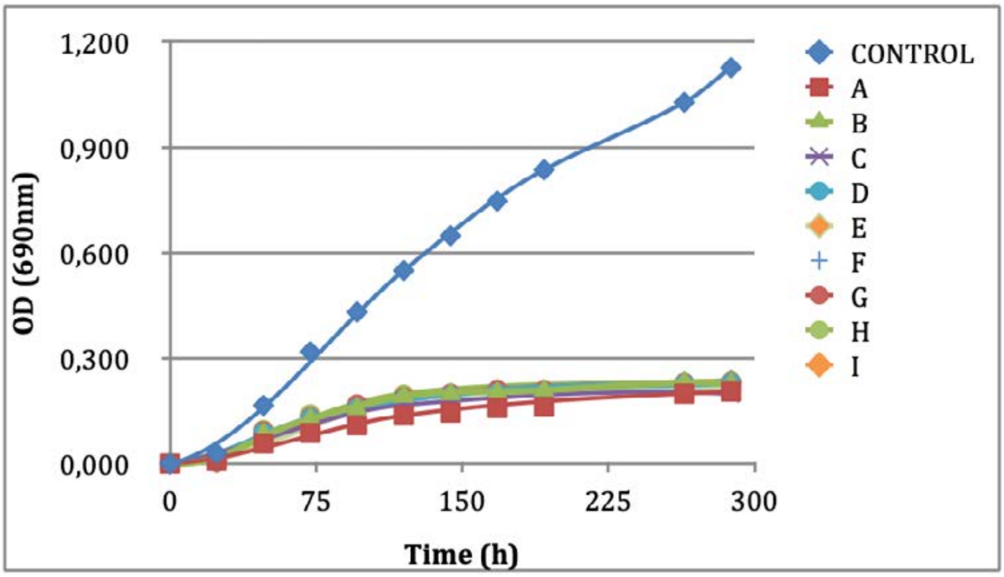

Figure 4. *Test with collector D effluent.*

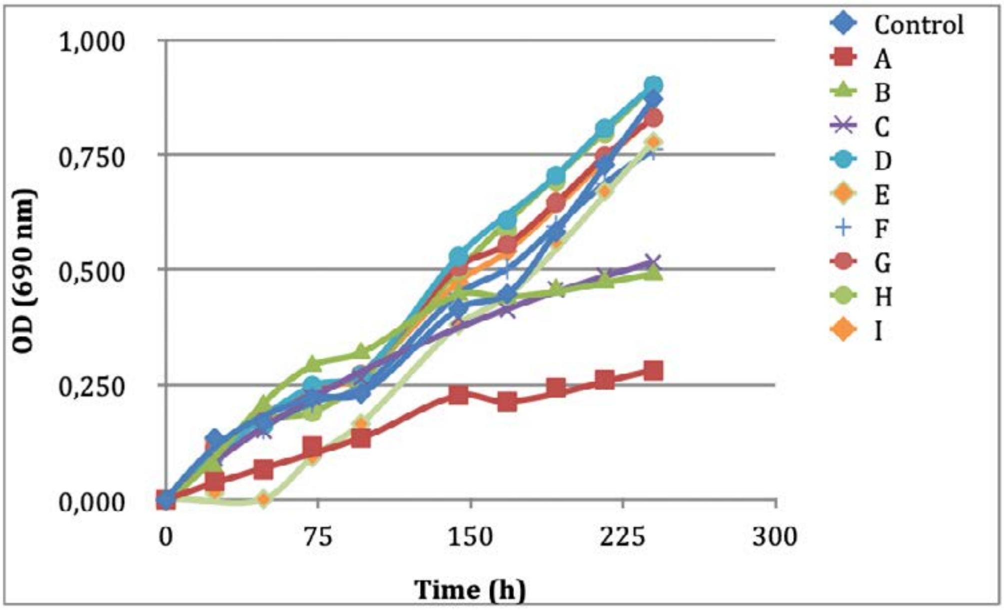

Figure 5. *Test with collector C effluent diluted 1:10.*

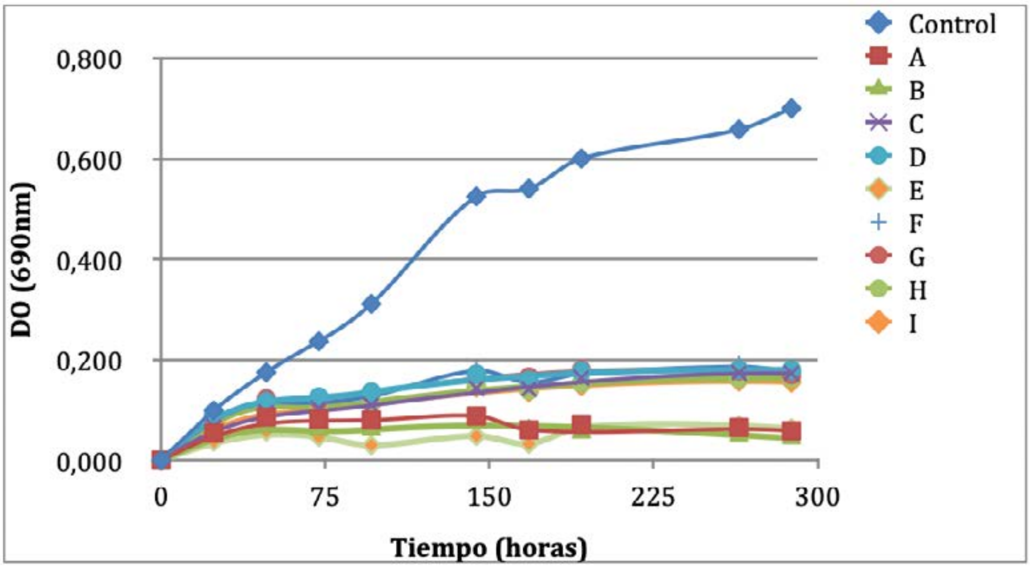

Figure 6. *Test with collector D effluent diluted 1:10.*

Based on the data, it can be seen that the real effluent is a less favourable medium for the growth of microalgae, as they grow better when the medium is diluted 1:10. It is important to note that the results obtained parallel to the first stage, as the undiluted treatments, which contain all the elements of the COMBO medium, are the ones that grow best. Furthermore, comparing both effluents, the microalgae grow better in the C.

CONCLUSIONS

These waters lack certain elements that are necessary for the growth of these micro-organisms. These effluents are complex mixtures of compounds derived from industrial processes that inhibit algal growth and crop development. Even in trials in which the effluents have been supplemented with macro and micronutrients, the mixture of metals and other compounds means that the microalgae have a period of adaptation to the environment.

In the trials with synthetic water and effluent water C and D, and with a dilution of 1/10, the treatability with microalgae was superior, producing cultures with similar growth to an optimal growth control.

It has been proven that these cultures in 1:10 dilution, reduce the nitrogen in the medium, which from the beginning has been considered as the main element to be eliminated from the effluents. The algae also act as heavy metal absorbers, removing metals from the medium and accumulating them in the biomass.

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05

SEWAGE SLUDGE TREATMENTS

SEWAGE SLUDGE: CONVERSION INTO AGRONOMIC BIOSTIMULANTS

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Abstract
This communication presents the pilot-scale design of a process for converting sewage sludge into agronomic products through aerobic fermentation with plant growth-promoting microorganisms (PGPR).The new product obtained in this fermentation process falls within the category of biostimulants, consisting of the Bacillus licheniformis biomass,which is the microorganism in charge of fermentation, hydrolyzed biomolecules, mainly peptides, and functional molecules secreted by Bacillus during fermentation.The pilot plant is divided into 3 parts that fulfill the functions of conditioning, physical pre-treatment and fermentation, respectively.Although itwould be easily scalable, the fermentation unit has the capacity to treat the sludge produced by an urban nucleus of 1350 equivalent inhabitants (54 Kg of sludge DM/day), producing 4.5 m3 of biostimulants in 6-day cycles, which would be ready for local application. In this way, it is proposed as a means of sludge management applicable to small and medium-sized urban and/or agro-industrial wastewater treatment plants.

INTRODUCTION

The activated sludge process is the most common process for treating sewage or industrial wastewaters(Pérez-Elvira et al., 2006). This generates large amounts of sludge, which management has a high cost for wastewater treatment plants (WWTP) (Collivignarelli et al., 2015)both at national and international level, in particular due to the uncertain recovery/disposal future options. Therefore, it is clear that the development of new technologies that can mitigate the problem at the source by reducing sludge production is necessary, such as the European Directive 2008/98/EC prescribes. This work shows the results obtained with a thermophilic membrane reactor, for processing a biological sludge derived from a wastewater treatment plant (WWTP. According to data from the National Sludge Registry, in Spain around 1,200,000 tons (in dry matter) of sewage sludge are produced annually.

Due to its high content of organic matter and nutrients, agronomic application is an important sink for this by-product, since its edaphological application can improve the chemical and biological characteristics of the soil and supply nutrients for plant growth(Cheng et al., 2007; Fernández et al., 2009)and the impacts of CSS amendment on soil physical and chemical properties. Soils amended with ≤20% CSS did not significantly affect the seedling emergence, while the contents of chlorophyll, nitrogen, phosphorous, and potassium of perennial ryegrass grown in such soils were greatly improved. Bulk density, water retention, and nutrient contents of the soil were also improved with the amendment of CSS, but high CSS contents introduced excessive amounts of heavy metals and soluble salts. Results show that Cu, Zn, and Pb accumulated slightly (up to 2.3 times. However, since they can contain heavy metals, pathogens and organic pollutants, they can pose a health risk(Guerrini et al., 2017; B. Rodríguez-Morgado et al., 2019) State of São Paulo, Brazil, therefore its direct application is prohibited by the Directive 86/278/EEC, being necessary to submit it to stabilization treatments and eliminate the potential pathogenic burden (Cheng et al., 2007).

Conventional biological technologies for sewage sludge valorization allowed by Spanish legislation (Real Decreto 506/2013, June 28, about fertilizer products) are mainly focused on composting,, for agronomic purposes, and anaerobic digestion, for both energy and agronomic purposes. While composting is a suitable method for small and medium WWTPs, due to its technical requirements, anaerobic digestion is applied mainly in large EDRs (Wei et al., 2000)

There are other technologies that, although not yet covered by the current legislative regulation, they have received great interest because they allow converting organic waste such as sewage sludge into biostimulant products applicable in the field of agronomy and bioremediation of soils contaminated by organic xenobiotics (Gómez et al., 2014; Tejada et al., 2018, 2016, 2014, 2011, 2010)obtained from rice bran, RB1 and RB2; municipal solid waste, MSW; and sheep manure, SM. Biostimulants are defined by the European Council of the Biostimulant Industry (EBIC) as "*substances and/or microorganisms that, applied to plants or to the rhizosphere, stimulate natural processes, improving/benefiting the absorption of nutrients, nutritional efficiency, tolerance to* *abiotic stress and crop quality*". In the particular case of fermentation and/or enzymatic processes applied to sewage sludge, obtained biostimulants are mainly constituted by low molecular weight peptides and free amino acids, microbial metabolites, such as phytohormone analogues, polysaccharides, humic substances, etc. as well as the micro-organisms of agronomic interest that carry out the fermentation(Parrado et al., 2008; Bruno Rodríguez-Morgado et al., 2019; Rodríguez-Morgado et al., 2015)we describe a biological process that converts carob germ (CG.

In this communication, a pilot-scale sludge fermentation unit is presented for its conversion into biostimulants. The fermentation process integrated in this pilot unit consists of a two-phase process: an initial sanitizing phase by a heat and pressure pretreatment, followed by a biological phase in which the sanitized sludge serves as a substrate for the growth of amicroorganism of agronomic interest (*Bacillus licheniformis*). The whole process will modify sludge, increasing its solubility and bioavailability, transforming it into a biostimulant mainly composed of protein hydrolyzate, *Bacillus* biomass and its secretome maily composed of hydrolytic enzymes and metabolites of agronomic interest (phytohormone analogs, chelating agents, siderophore compounds, biocontrol activity compounds...).

RESULTS AND DISCUSSION

The process integrated in the fermentation unit has been optimized at the laboratory level and the equipment has been selected based on optimization tests. The fermentation unit consists of three parts:

1. Sludge conditioning
2. Physical pretreatment of the sludge (thermal-pressure)
3. Aerobic fermentation

<u>1.- Sludge conditioning:</u>The conditioning consist of concentrating the sludge up to 6% w/vby a rotary drum sieve concentration system(Figure 1, right). This concentration is intended to improve the efficiency of the fermentation process.

Figure 1. *The figure shows the concentrator system composed of the flocculation tube (left) and the rotary drum sieve (right).*

The concentration system requires sludge to be previously mixed with a flocculating polyelectrolyte that agglomerates the sludge flocs, preventing their passage through the sieve, and separating them from the water. For that purpose a flocculation tube (Figure 1, left) is coupled upstream of the concentration system allowing the sludge to circulate through a sinuous path homogenizing it with the flocculating polyelectrolyte.

2.- Physical pretreatment of sludge:The second stage,consist of a thermal pretreatment for sanitation purposes. The pretreatment system integrates the *Cascade Flash (C&F)* technology that is detailed in other communication. Briefly, it consists of a continuous thermal and pressure pre-treatment by means of a sudden steam explosion (SSE), in which sludge reaches 120 °C and 1.2 bar overpressure by injecting steam in the opposite direction to the flow of sludge. After a short residence period subjected to these operational conditions, the pressure is suddenly released, causing the sanitization, hydrolysis and solubilizationof sludge by breaking cell membranes of the microorganisms. Said system integrates a sludge preheating unit, made up of a 2 m^3 buffer tank and a tubular heat exchanger, and the *C&F* device, made up of a sterilizer and a sudden decompression tank, coupled to cooling system based in a decompression tank by vacuum pump (Figura 2).

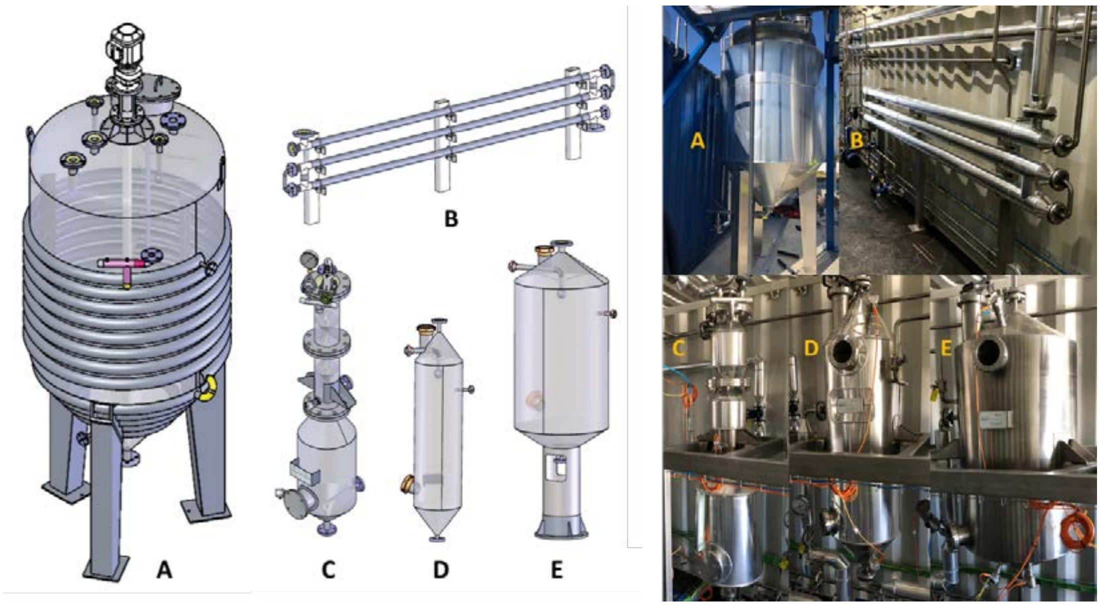

Figure 2. *The figure shows the different systems that integrates the pretreatment system. A – Buffer tank; B – Tubular heat exchanger; C – C&F device; D – Flash 1; E – Flash 2.*

The operation of the pretreatment system is detailed below. Concentrated sludge is stored in the buffer tank (Figure 2, A) being slowly preheated to 80 °C. Then sludge is pumped through a tubular heat exchanger (Figure 2, B) reaching 90 °C, taking advantage of residual heat from the *C&F* device. Next, sludge pass into the*C&F* device (Figure 2, C) which is made up of two parts: the upper part is a mixer that allows steam to spread quickly through the sludge, and the bottom part is a tank that retains sludge, maintaining operational conditions for a short period. Here, in the bottom, is where the steam is injected (12 Kg h^{-1}, 3 bar). After the short period, pressure is suddenly releasedand sludge violently pass to a flash tank(Flash 1, figure 2, D) favoring the sanitizing effect. Finally, in the cooler system (Flash 2, figure 2, E)sludge are subjected to adecompressioncoolingdownupto 45 °C. Steam for heating requirements is produced by an electric steam boiler.

The sludge flow through the pretreatment system is 0,18 m^3 h^{-1}. Considering treatment cycles of 5 days, and 5 hours each day, this system is capable of treating 4.5 m^3 of sludge per cycle (6 % w/v DM), generating a sanitized product, with an approximate pre-hydrolyzed soluble content of 8%, ready to be used as a fermentation substrate.

Within the process encompassed in the fermentation unit, the pretreatment system represents a bottleneck, presenting a treatment flow of0.18 m^3 h^{-1}, while the rotating drum sieve is able to concentrate about 0.2 - 1 m^3 h^{-1}, and the fermenter has a capacity of 8 m^3. However, treatment capacity ofpretreatment systemcould be easily increasedby coupling several sterilizers in parallel, and increasing the power of the boiler.

3.- Aerobic fermentation: Cooled sludge is now a suitable substrate to grow *Bacillus*. This biological-nature stage takes places in an 8 m^3 fermenter(Figura 3) which keeps the fermentation conditions constant: temperature 45 °C, stirring 60 rpm.

Figure 3. *8 m3 fermentation tank. Scheme (left) and image of the equipment installed in the fermentation unit, outside the maritime container (right).*

The fermentation cycle lasts for 6 days, after which 4.5 m^3 of fermented sludge are harvested. 10% of the total fermentation volume remains as inoculum for the following fermentation.

As mentioned above, the pretreatment system is the bottleneck of the sludge recovery unit. For this reason, the fermentation, which takes place in a discontinuous regime, is carried out in such a way that 0.9 m^3 of pretreated sludge are loaded each day on the fermentation that is already in progress (Figure 4).\

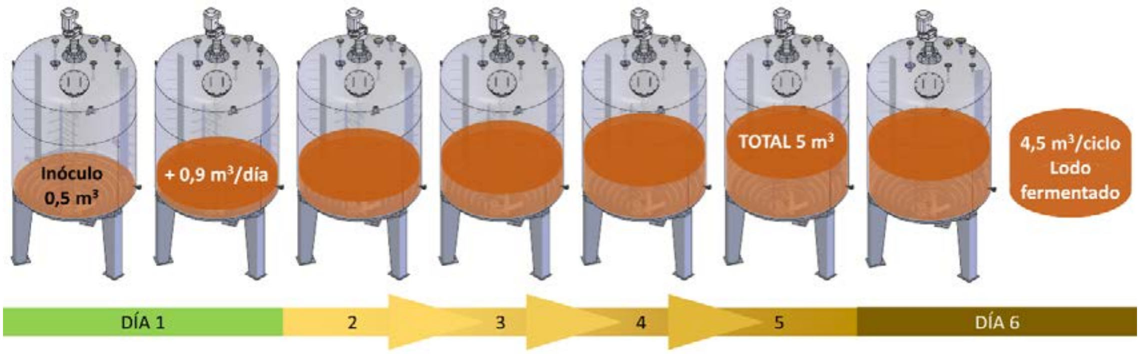

Figure 4. *Sludge fermentation cycle scheme.*

<u>Sludge fermentation unit:</u>The integration of all equipment is shown in figure5 and 6. Therotary sieve drum is arranged on an elevated walkway, 3.35 meters high. The rest of the equipment and pumps, with the exception of the 2 m^3 buffer tank and the fermenter, are located inside a 12 m long maritime container while facilitating its installation and transport as a portable unit.

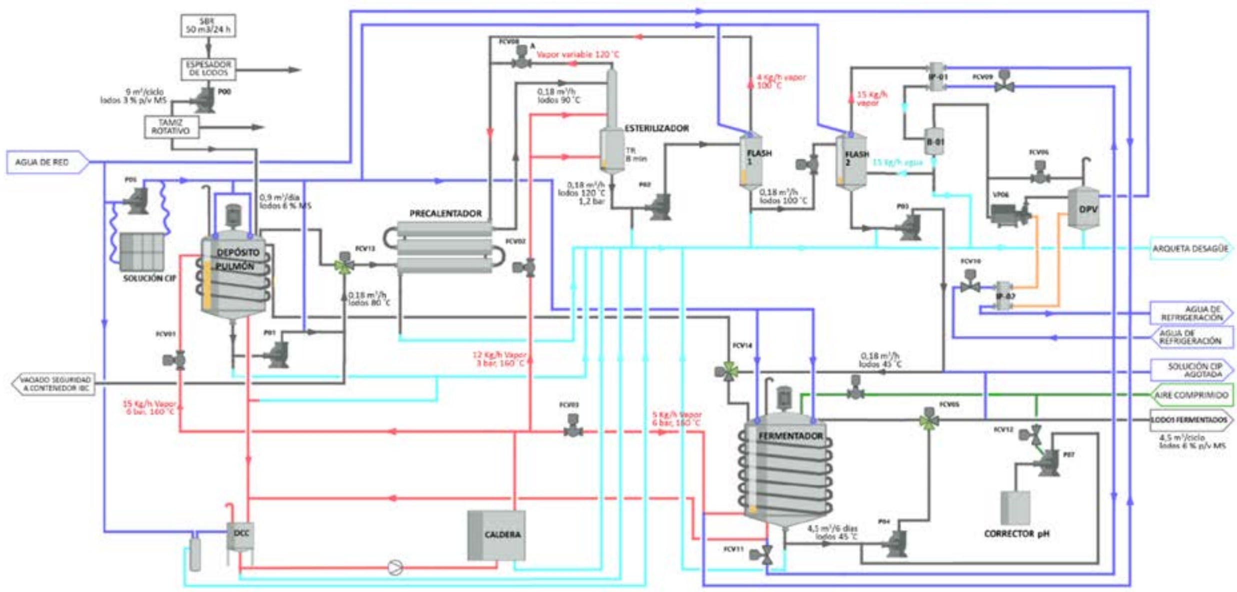

Figure 5. *The figure shows diagram of the fermentation unit.*

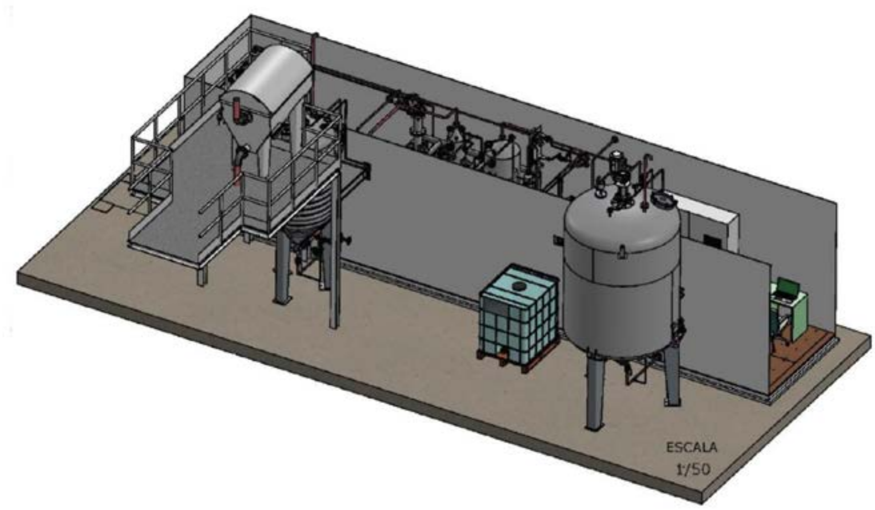

Figure 6. *The figure shows the equipment arrangement in the sludge fermentation unit.*

Global balance of the process on an industrial scale: Considering that the fermentation process takes place in 6-day cycles, and that these, in turn, include work cycles of 5 days and 5 hours a day for conditioning the sludge as a fermentation substrate, at the end of the process 9 $m^3$of sludge will be treated. These comes from the treatment of about 150 - 225 m^3 of wastewater. The process generates 4.5 m^3 of fermented sludge per cycle.

Table 1 shows an estimate of the global balance of the fermentation process carried out in the fermentation unit.

Estimated electricity consumption	Potency (KWh)	N° hours/cycle	Consume/cycle (KW)
Pump P00	0,37	18	6,6
PumpP01	0,75	25	18,75
PumpP02	0,75	25	18,75
PumpP03	0,75	25	18,75
PumpP04	1,1	10	11
PumpP05	1,8	2,5	4,5
Vacuum PumpVP06	1,5	25	37,5
PumpP07	-	-	-
Rotary sieve	0,75	18	13,5
Buffer tank	1,1	25	27,5
Fermenter	4	25	100
Air compressor	2,2	1	2,2
Electric steam boiler	30	7,5	225
			Total = 484,05

Table 1. *The table showsan estimate of the balance of the industrial sludge fermentation process developed in the fermentation unit.Electricity consumption to treat 9 m^3 of sludge(3 % w/v DM).*

The total electrical consumption estimated by the fermentation unit in each fermentation cycle is 484.05 KW; 46.5% corresponds to the consumption of the electric steam boiler. However, in practice, this consumption may be lower due to the thermal insulation of every equipment. On the other hand, although it is not contemplated in this unit, the installation of a heating system using renewable energy (thermosolar) would considerably reduce the total electricity consumptionby around 62.5%.

The fermentation unit does not have an associated consumption of water, since the cooling system use water from a closed circuit.

This fermentation unit can be coupled to small and medium-sized urban and agro-industrial WWTPs, being an alternative to sludge management. This unit has the capacity to treat daily 1.8 m3 of sludge (3% w/v DM), which is equivalent to 54 Kg of sludge (DM). Considering that the estimated daily production of sludge per capita is 0.04 Kg (DM) (Karagiannidis et al., 2011), this unit would have the capacity to manage the sludge produced by an urban nucleus of 1350 equivalent inhabitants (he). However, as also mentioned previously, the sludge pretreatment system, which is the bottleneck of the process, is easily expandable, which would allow managing the sludge produced for a greater number of he.

CONCLUSIONS

A fermentation unit has been designed for the conversion of sewage sludge into biostimulants. This unit is applicable to small and medium urban and/or agro-industrial wastewater treatment plants. After a 6-day cycle, the process developed in the fermentation unit produces 4.5 m3 of biostimulant product constituted by the biomass of *Bacillus licheniformis*, which is the microorganism in charge of fermentation, hydrolyzed biomolecules, mainly peptides, and functional molecules secreted by the microorganism during fermentation.

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CONVERSION OF DRYING BEDS INTO MACROPHYTE BEDS TO DEWATER SLUDGE AT THE ALCOUTIM WWTP

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Abstract

The sludge treatment and dewatering is a critical step in the biological treatment of wastewater. An efficient dewatering contributes to the good performance of the treatment system, and there are several technologies on the market that allow the achievement of high degrees of dewatering. However, the application of equipment and the consumption of energy increase the investment and operating costs of the systems. In this context, the application of solar sludge drying systems and macrophyte beds appear as viable alternatives, for certain situations, in relation to the conventional sludge dewatering process, allowing to considerably reduce the moisture content and the pathogens, thus improving the sludge quality. The objective of this work is to present a case study in which to two of the three drying beds of the Alcoutim WWTP were converted to a dewatering system by macrophyte bed. It is an innovative system for the Algarve region and as far as we know, there are still few full scale systems applied worldwide. On the other hand, there remains uncertainty about specific aspects of design and operation of the systems that require research to

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make the systems more sustainable, such as the applied sludge loads and its variation throughout the year, taking into account climatic conditions, the type of sludge produced and the seasonality of the region. This communication aims to present the results of the start-up and operation phases of this system.

INTRODUCTION

In the WWTP, the sewage sludge produced is intended to have the best possible quality, with high levels of dry matter and complying with the quality required for the intended destination. This would allow the sludge to be used as close as possible to the source, thus reducing the respective carbon footprint, as well as the respective operating costs, which can represent about 20 to 60% of the total operating costs (von Sperling and Gonçalves, 2007).

Given the national reality in which a good part of the soils are lacking in terms of organic matter and nutrients, the preferred destination of the sewage sludge, after being subjected to temporary storage and, or treatment, e.g. composting, has been on agricultural soils.

On the other hand, in a society where aspects such as climate change, circular economy, decarbonisation of the economy and energy efficiency and sustainability are increasingly pressing, it is of interest to adopt new solutions that are in line with such designs, also in the water sector.

In this context, the application of solar sludge drying systems and macrophyte beds appear as possible alternatives, in certain situations, in relation to the conventional sludge dehydration process, allowing to considerably reduce the moisture content and increase its quality from the microbiological view.

It was from these premises that Águas do Algarve, SA joined the consortium of the INTERREG POCTEP SECASOL project with the objective of improving the quality of the final sewage sludge and, at the same time, reducing the costs associated with its transport, using macrophyte plants that, associated with solar energy, they can contribute to the drying and promote sludge valorisation, allowing eventually to fulfil the values of the microbiological parameters necessary for direct agricultural recovery (Decree-Law n.º 276/2009).

It is an innovative system for the Algarve region and even at an international level few systems are implemented atfull scale. On the other hand, there remains uncertainty about specific aspects of systems designing and exploration that require further research to make these systems more sustainable, namely the need or not of the system to have pipes for aeration and the optimal sludge loads criterion applied and its variation along the year, taking into account climatic conditions, the type of sludge produced and the seasonality of the region. This communication aims to present the results of the start-up and operation phases of this system.

METHODS

The Alcoutim WWTP is located in the northwest of the Algarve and was designed for an equivalent population of 868 e.p., corresponding to a flow of 183 m^3/day. The facility currently receives an average daily flow, on an annual basis, of around 50 m^3/day. The raw tributary to the WWTP is initially subjected to harrowing and sanding, followed by biological treatment with activated sludge. Regarding excess sludge, they are removed from the bottom of the decanter and are sent to the drying beds, leading to an annual sludge production of about 20 t OM

Two of the three drying beds of the Alcoutim WWTP (Bed 2 and Bed 3) were converted into a dehydration system by macrophyte beds, *Phragmite australis*. In Bed 2, ventilation was promoted through the installation of the respective pipes, in order to compare the performance of the system with (Bed 2) and without aeration (Bed 3). The following table (Table 1) shows the dimensions of each Bed.

Table 1. *Dimensions of each bed of macrophytes.*

Measure	Value	Units
Length	10	m
Width	6	m
Height	2,4	m

As for the filling of each bed, it consists of a layer of stones at the bottom (granulometry 0.5-1 mm) with a thickness of 15 to 20 cm, followed by an intermediate layer of 20 to 30 cm of gravel (granulometry 2 -10 mm), and a layer of 10 to 15 cm of sand (granulometry of 0,5 to 1 mm), following the one proposed by Uggeti et. al (2011).

In the exploration phase, changes were made to the level of mud applied to the beds, in order to ascertain their response, namely with regard to possible effects on the level of accumulation /infiltration of the liquid on the surface of the beds, as well as the plant development.

The load applied to each bed (kg/m^2.year) can be determined using the following formula, in which the extraction time corresponds to the sum of the pump extraction times throughout each month and the concentration of the sludge fed is estimated based on the average of the available results of the sludge extracted from the decanter.

The feeding must be alternated between the available beds, with intervals of 2 to 3 days, in order to allow the percolation of the beds, as well as to maintain the balance between the development of the plants and the sludge load.

RESULTS AND DISCUSSION

The system started in April 2020, with the beds being fed alternately, with spacing of 4 to 7 days. In the first month of exploration, the system was fed for 16 min, followed by longer times, up to a maximum of 56 min in the months of July and September (Table 2).

Table 2. *Feeding times to macrophyte beds*

Load applied to macrophyte beds	Feeding Bed 2 (min)	Feeding Bed 3 (min)
Month	min	min
April	16	16
May	20	20
June	24	24
July	36	56
August	48	48
September	48	56

The surface load of solids applied to each bed was determined based on the capacity of the extraction pump, the feeding times (Table 1) and the concentration of total suspended solids in the sludge recirculation, which was considered identical to that existing at the bottom of the secondary decanter, showing the respective values in Table 3.

Table 3. *Load applied to each bed of macrophytes in the last months and visual observation at the level of clogging*

Load applied to macrophyte beds	Average load applied Bed 2 (aeration)	Average load applied Bed 3 (without aeration	Clogging Leito 2	Clogging Leito 3
Month	$kgMS{\times}m^{-2}.{\cdot}year^{-1}$		Yes / Some / No	
April	42	42	No	No
May	53	53	No	No
June	63	63	No	No
July	95	148	Some	No
August	126	126	Yes	No
September	126	148	Yes	No

In Figure 1, the applied loads of both macrophyte beds (2 and 3) are compared with the reference value 60 kg $MS{\times}m^{-2}{\times}year^{-1}$ (Uggeti et. al, 2011).

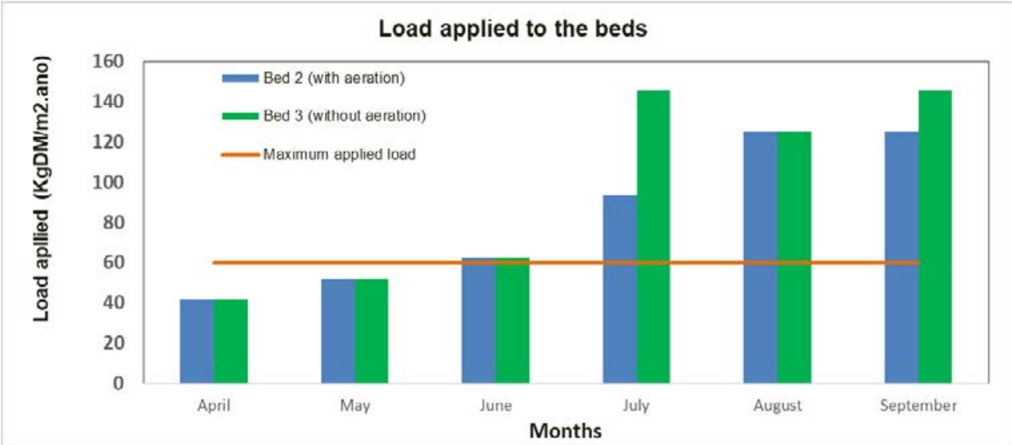

Figure 1. *Loads applied to macrophytebeds (2 and 3) in the Alcoutim WWTP*

In the first three months of operation, the load applied to each bed was close to the recommended value (60 kg MS×m^{-2}×year^{-1}).(Uggeti et. al, 2011).After the month of July, the extraction of mud increased considerably, with the applied load increasing to values of 95-126 kg MS×m^{-2}×year^{-1} e 126-148 kg MS×m^{-2}×year^{-1}, respectively in beds 2 and 3.Thus, it appears that bed 3 (without aeration) has been more overloaded than bed 2, namely in the months of July and September. In Figure 2, it is possible to visually evaluate the development of plants, as well as the balance between their appearance and the respective accumulation/infiltration in both beds, from June to September.

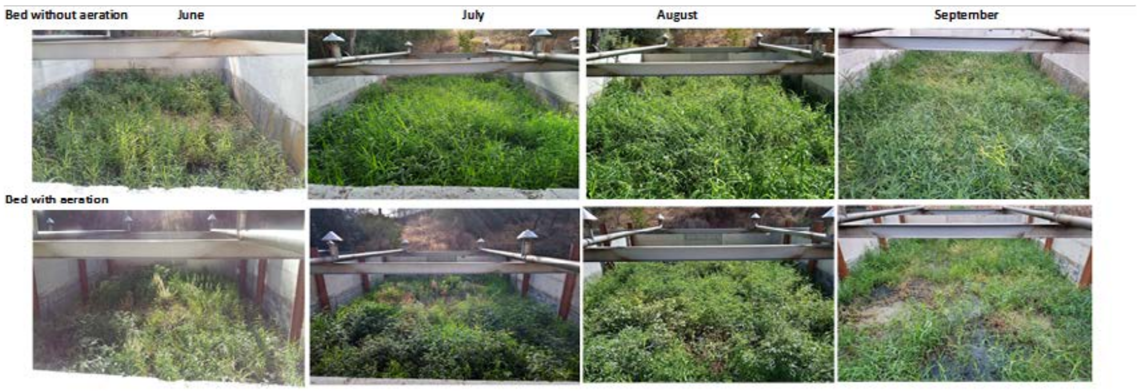

Figure 2. *Evolution of macrophyte beds 2 and 3 in the period from June to September 2020*

Thus, it appears that although Bed 2 has an aeration system with perforated pipes at the bottom in order to allow better infiltration and percolation of runoff, in practice it is the one that presents the greatest flow problems, showing some clogging. On the other hand, the Bed 3, despite not having any aeration and pipes for percolation of runoff, has withstood higher loads, without showing signs of clogging. With some surprise, the implementation of the aeration system, with perforated pipes, did not prove to be more effective. The system's operating period is, however, still very short, so it is necessary to wait for longer periods of operation.

Additionally, Bed 3 (not ventilated) showed a good infiltration as well as a better balance in plant growth. As of September, there is a slight change in the appearance of the plants in both beds, with the appearance of yellowish tips on the leaves. The sludge overload applied to the beds, with values much higher than the recommended values (60 kg MS×m^{-2}×year^{-1}), seemed to have compromised the balance of the development of *Phragmite australis*, which is in line with other studies (Uggeti et. al, 2011).

The sludge load was then decrease by changing thesludge extraction times for the macrophyte beds to about 22 min. On the other hand, with the decrease in atmospheric temperature and the increase in rainfall in the autumn/winter period, as well as the decrease in evapotranspiration (from 25 to 30 mm /d in the summer to around 2 to 10 mm/d in the winter; Pedescoll, 2010), can still lead to some adjustments of the applied surface load.It is therefore necessary to continue to closely monitor the operation of the macrophyte beds.

CONCLUSIONS

This article presents the results obtained in the first six months (between April and September 2020) of the operation of the sludge dewatering system by macrophyte beds (*Phragmite australis*) that was implemented at the Alcoutim WWTP.

In the first quarter of operation (between April and June) the system was operated with applied surface loads close to the reference value (60 kg MS×m^{-2}×year^{-1}),having good performance in terms of water infiltration and development and aspect of the plants (lush and with a strong green color) for both beds. With the considerable increase in the surface load, in both beds, in the following three months (July to September), there was a slight deterioration in the appearance of the plants, with Bed 2 presenting some accumulation of liquid on the surface. This aspect requires verification for a longer time of operation since it was not expected to occur in the bed that has aeration.

The continued operation of the system and the careful monitoring of operating conditions will allow to optimize its operation throughout the year. The results will serve as a guide for the design of similar systems in other facilities located in the Algarve region, or in other regions with identical climates.

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MODELING OF WWTP SLUDGE DEHYDRATION IN DRYING BEDS

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Abstract

The by-product of wastewater treatment is called sewage sludge and must also be treated for final disposal. Its treatment encompasses three main objectives: Stabilization of organic matter, reduction of volume, and reduction of pathogens load. Sludge dehydration is an essential step in the process since the reduction in volume translates into greater ease and lower costs of transportation and landfill. Some fast and compact methods used in dewatering sludge have high energy consumption and are expensive. Contrary to that are the drying beds, which due to the simplicity of construction and operation and the low cost, are a priority alternative. The objective of this work was the formulation of mathematical models that describe the dewatering of WWTP sludge in drying beds, depending on the climatic conditions. The results of the models were satisfactory and a significant reduction in volume was observed, even in cold and humid weather.

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INTRODUCTION

Many conferences related to the theme of wastewater treatment have been carried out in the last decades (UN, 2017), highlighting a problem related to the subject and the need for debate and research.

Effluent treatment generates a by-product, the called sewage sludge, which must undergo appropriate treatment before final disposal.Sludge is considered solid waste from wastewater treatment, however, it has approximately 95% water (Sperling and Franci, 2001). A large volume is generated continuously, so its dehydration is an essential step in the process, and the reduction in volume translates into greater ease of operation and a lower cost, both for transportation and landfill.

There are several techniques for sludge dehydration, with the selection of the most favorable method depending on several factors: area available for its implementation, the volume of sludge generated, availability of technology, and climatic factors. Mechanical methods are faster and more compact but are associated with high energy costs, in addition to the complexity of installation and operation. In the drying beds (DB), on the other hand, the sludge is exposed to the environment, resulting in the evaporation and leaching of water by natural effects. In this way, they are exempt from spending on electricity and are easy to install and operate, requiring only time and space superior to energy methods (Sperling, 2007).

The choice of drying beds takes into account the volume of sludge generated, the available space, and the weather patterns of the region. Thus, the existence of mathematical models that relate these data is essential, allowing the evaluation of the technical feasibility of the process for each situation. This study intends to develop a mathematical model that describes the dehydration of sludge from Wastewater Treatment Plants - WWTP in drying beds, relating meteorological data in the water mass balance.

MATERIALSAND METHODS

The experimental study consisted of the design, construction and application of drying beds for sludge dewatering at the WWTP in Bragança, Portugal. The sludge treatment station using drying beds was set up at the Campus Santa Apolónia, Polytechnic Institute of Bragança - IPB. Built using four tanks (TK), with a square section (1m x 1m) and a useful volume of 1000 L, perfectly watertight, with a sampling port on the base for collecting leachate. All TKs had a graduated rod inside, to monitor the thickness of the sludge layer.

The draining bed of each DB consists of three overlapping layers of inert material; at the base, a layer of 30 cm of coarse gravel (20-32 mm), followed by 10 cm of fine gravel (8-12 mm), and finally, 10 cm of sand (granulometry up to 0.6 mm). Subsequently, the drying beds received a certain volume of sludge: 20 cm high in TKs 1 and 2 and 30 cm for tanks 3 and 4, as shown in Fig

1. These thicknesses were used following the recommendations of Metcalf and Eddy (1995). For TKs 1 and 3, the sludge was mixed and homogenized daily, with a metal shovel, in order to assess the influence on the dehydration process.

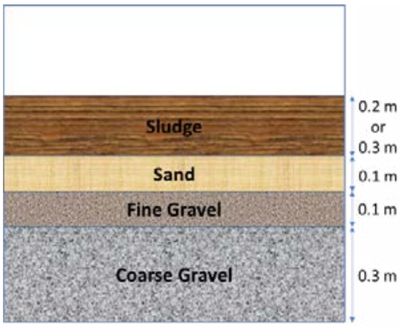

Figure 1. *Materials and composition of the drying bed*

Two dehydration cycles were carried out, each lasting 30 days, corresponding to different seasons (dry season and rainy season). The parameters monitored daily during the tests correspond to: sludge Solid Content (SC); sludge layer thickness; and drained volume of water. SC analyzes, in duplicate, were performed according to Standard Methods (APHA, 1998). The meteorological data were collected at the IPB meteorological station.

The mathematical modeling was elaborated from: the estimate of evaporated water (daily), by energy conversion equations, using meteorological data; the volume of water drained (daily), relative to the system's initial water; precipitation (volume of water) over the bed area. With these data, the water mass balance can be estimated. Fig 2 shows the flowchart for the elaboration of the models, where correction factors were established, in order to correct the evaporation and drainage of water, according to the instantaneous SC each day (Lima, 2020, p. 46 - 55).

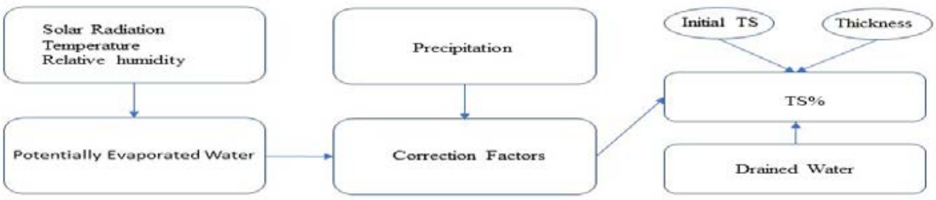

Figure 2. *Flowchart of mathematical modeling*

For the preparation of the mass balance, initially, the water potentially evaporated was found for each day of the experimental test, and then the volume of precipitated water was determined. Finally, it was estimated the water drained from the beds, referring to the initial water content in the sludge. By the trial-and-error method, observing the approximation of the theoretical curve to the experimental SC curve, some correction factors were attributed to the correction of evaporation and drainage, since these factors are influenced bythe instantaneous SC value of the sludge.

RESULTS AND DISCUSSION

Since each TK had different characteristics, 4 mathematical models were obtained. Fig 3 shows the experimental results (blue line) and the theoretical results (orange line), found from the models, for the 4 DB (cycle 1 and 2, left and right side, respectively). The figure also shows the occurrence of precipitation during the testing period.

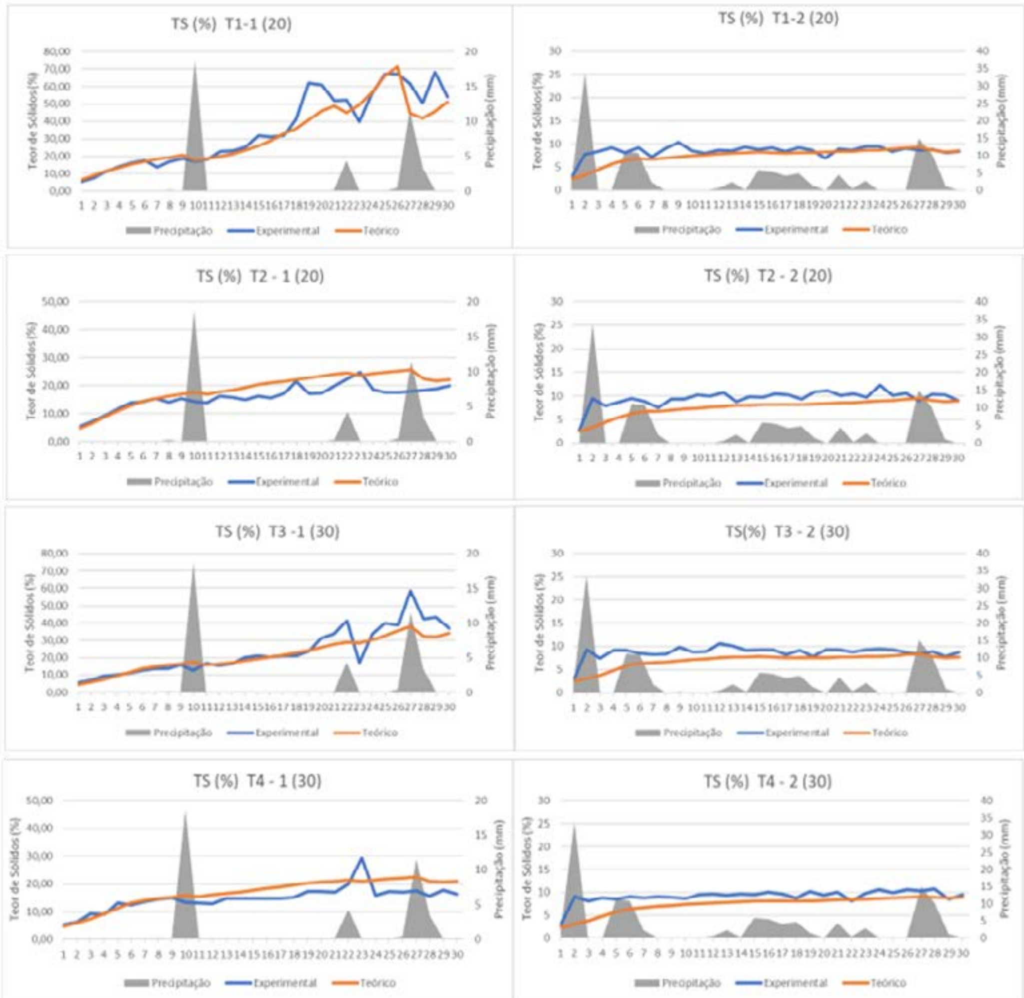

Figure 3. *Mathematical models and experimental data (Solid Content)*

An approximation of the theoretical and experimental curves is observed in most of the period, for both cycles, showing that the model is adapted to different climatic conditions. A peak of decrease in SC in the experimental curves occurred after the occurrence of precipitations, when the SC was close to or greater than 20%. This peak is less accentuated in the theoretical curve;however, the two curves return to similar values soon after the event. This factor happens, because experimentally a portion of the precipitation is incorporated in the upper part of the sludge and evaporates later, since the model does not foresee this effect and only considers that it is drained immediately.

In cycle 1, the TK 1 and 2, whose initial layer of sludge was 20 cm, showed better dehydration in relation to the other tanks, as a result of the lower thickness of the sludge layer. It is also observed that for beds where daily homogenization was increased, the SC values were higher than those without turning, such as for DB1, which recorded SC values close to 70% and the TK 2 of only 30%.

In cycle 2, the results showed no significant differences between the tanks, and the SC values at the end of the tests were approximately equal, in the range of 10% - 15%. But it was observed that even in cold and humid weather, the volume reduction was significant, reaching values that exceeded 75%, of reduction of the initial volume, tank 3.Tanks 1 and 2 had volume reduction efficiency of 70% and 75%, respectively. Tank 4, on the other hand, with the lowest performance, obtained a 2/3 reduction. All tanks achieved volume reduction within the range of 50% and 80%, which is expected in sludge dehydration (Hazel, 2015)

CONCLUSIONS

The mathematical modeling of the drying sludge dehydration process proved to be quite satisfactory, since it adequately represented the experimental results in different climatic conditions, following the experimental concentration of SC in practically the entire period of the tests. The drying beds, even in unfavorable climatic conditions for water evaporation, showed a good performance, significantly reducing the volume of sludge from the water that was drained from the system.

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SOLUTIONS FOR SLUDGE MANAGEMENT IN SMALL AGGLOMERATIONS: DECENTRALISED ANAEROBIC DIGESTION (SMALLWAT21)

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Abstract

In Spain, almost 30% of the total sludge is produced in small WWTPs, which serve less than 20.000 population (XV Estudio Nacional Suministro de Agua Potable y SaneamientoenEspaña (2018)). This sludge is collected from these WWTPs without specific treatment, leading to management costs and environmental impact. Therefore, Aqualia is studying the possibility to develop sludge treatment systems for small-medium populations in which the biogas produced is used to heat the digester without losing the decentralised and low cost horizon.Two different systems are being developed depending on the WWTP size: small (less than 2.000 population) or medium-small (less than 20.000 pupulation). Within H2020 Scalibur project activities, a system for small populations is being developed based in polyethylene bags. In parallel, Aqualia is developing a different design for medium-small populations based on covered lagoons.It is necessary to assess the amount of WWTPs without sludge treatment on site. From these WWTPs a techno-economic analysis will be assessed in order to understand which of them may benefit from sludge transportation to a centralised WWTP or which of them are isolated and would benefit from in-situ sludge treatment.

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INTRODUCTION

Current regulations for sludge management is becoming stricter both at National and European level.In Spain, almost 30% of the total sludge is produced in small WWTPs, which serve less than 20.000 population (XV Estudio Nacional Suministro de Agua Potable y SaneamientoenEspaña (2018)). This sludge is collected from theseWWTPs without specific treatment, leading to management costs and environmental impact. In addition, working towards reducing the amount of sludge produced will help to achieve the new targets of coming regulations.

The anaerobic digestion systems called "low cost" aim to treat sludge in a decentralised way and are characterised by its easy operation and maintenance (Garfí et al., 2019). Due to the high influence of temperaturein its implementation, these systems are mostly used in regions close to the tropicswhere digestion temperature reach 20-30°C (Garfí et al., 2019).

The effect of temperaturein biogas production is shown in Table 1(resultsobtained in a bench scale Aqualia R&D study).These results show that the operation of anaerobic digestion at low temperatures (<20°C - psychrophilic) reduces sludge production in 8-18%, while operation at higher temperatures (>30°C – mesophilic) reduces sludge production in 32-35%. Therefore, Aqualia is studying the possibility to develop systems for small-medium populations in which the biogas produced is used to heat the digester without losing the decentralised and low cost horizon.

Table 1. *Effect of temperature in biogas production and Volatile Solids removal*

Tempe.	Yield	Elim. VS
(°C)	($NLCH_4/gVS$)	(%)
37	0.39	49.0
35	0.36	46.0
33	0.35	45.0
30	0.31	39.0
20	0.21	25.9
15	0.15	19.0
10	0.10	12.1

METHODS

Two different systems are being developeddepending on the WWTP size: small (less than 2.000 population) or medium-small (less than 20.000 pupulation). Within H2020 Scalibur project activities,a system for small populations is being developed (see Fig 1).

This system consists in three10 m^3polyethylene bags with gas collection and mixing in the form of recirculation. From the 10m^3, 7 m^3 are used for digestate and the remaining 3 m^3 for gas storage.Each bag will be operated with different conditions: one bag kept at atmosphere temperature (B3), one with thermal isolation (B2) and the last one with thermal isolation and heating (B1). Different conditions of hydraulic retention time and recirculation flow will be tested. Currently the three bags are working with a hydraulic retention time of 20 days, being fed with 300 L / day each, in total 0,9 m^3 per day. Recirculation rate is 7 m^3 per hour, meaning that the content of each bag is moved once every hour.

In parallel, Aqualia is developing a different design for medium-small populations based on covered lagoons with mechanical mixing and heating.

Figure 1. *Anaerobic Digestion for small populations (less than 2.000 population)*

RESULTS AND DISCUSSION

During the time that the three bags have been in operation, the heating system was not in place, therefore B1 and B2 were in the same conditions (with a isolating layer) and B3 exposed at atmospheric temperatures.

Figure 2 shows that Volatile Solids removal has been similar in the three bags and achieving good values around 50% but with a high variability, which might be linked to the mixing strategy. Biogas quality is also good with an average of 55% of methane. Biogas production is variable ranging between 0,12 and 1,26 m^3 per day.

Next stage of the analysis will be analysis the temperature that can be reached in the system using the biogas produced.

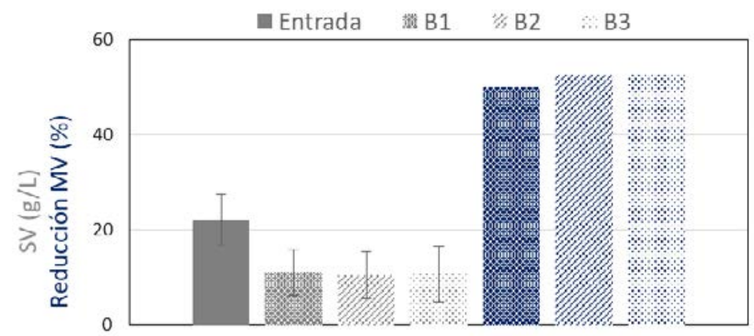

Figure 2. *VolatileSolids and volatilesolidsreductionduringMarch 2021*

CONCLUSIONS

It is necessary to assess the amount of WWTPs without sludge treatment on site. From these WWTPs a techno-economic analysis will be assessedin order to understand which of them may benefit from sludge transportation to a centralised WWTP or which of them are isolated and would benefit from in-situ sludge treatment. This analysis will give a length in km within which the transportation of sludge would be viable. Outside this range, the systems developed by the activities within H2020 Scalibur project and Aqualia will try to offer solutions with low capital and operational costs that will help to minimise the impact of sludge production to both companies and environment.

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ECODIGESTION: A CONTROL TOOL FOR ANAEROBIC CO-DIGESTION IN WWTP

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Abstract
A control and management tool for the anaerobic co-digestion in WWTP was developed by using a dynamic mathematical model, based on the IWA Anaerobic Digestion Model No. 1. The developed tool was able to calculate the amount of co-substrate to be fed to the system in order to achieve a desired biogas production to cover energy demand of the WWTP y using the prediction of the biogas production. The main kinetic and stoichiometric parameters were obtained from batch laboratory-scale experiments and the data from continuous operated demo plant was used to validate this model using as co-substrates orange waste and organic acids. The obtained results showed a good agreement between simulation and target biogas production and between model prediction and experimental values.

INTRODUCTION

The energy consumption in wastewater treatment plants (WWTPs) varies depending on volume of water treated, organic load and the type of treatment technology used and it is estimated to entail in typical WWTPs between 25 and 40 % of total operating costs (Panepinto *et al.*, 2016). This value varies between the range of approximately 0.3-2.1 kWh/m^3 of treated wastewater (Gandiglio *et al.*, 2017). Albadalejo *et al.* (2014) concluded that the energy consumption increases in an exponential way while the size of WWTP decreases, from values of 0.286 kWh/m^3 of wastewater treated in facilities for the treatment of 200,000 population equivalent (p.e.) to 2.807 kWh/m^3 of wastewater treated for the treatment of 250 p.e. In addition to this, the production of the energy needed by WWTPs in the European Union (EU) causes the emission of 27 million tonnes of CO_2 (Lotti, 2016).

In wastewater treatment it is generated a stream of sludge that requires its stabilization prior to its final disposal. This stabilization can be done by aerobic or anaerobic biological processes. Aerobic stabilization presents higher operating costs since it requires oxygen supply, high retention times and, therefore, large reaction volumes. Anaerobic digestion stabilizes this sludge with a lower operating cost, enabling the recovery of energy from the wastewater generating a biogas rich in methane, but for this it is necessary to have a higher investment cost. Gretzschel *et al.* (2014) studied the costs associated with the transformation of aerobic systems into anaerobic one and concluded that this transformation was economically favourable for facilities up to 7500 p.e.

On the other hand, the EU agri-food sector produce about 88 million tonnes of animal and vegetable waste per year (Edwards *et al.*, 2015), resulting in management costs of 143 billion euros (Fusion EU project). The management of this waste using current alternatives also entails an environmental cost, with greenhouse gas (GHG) emissions that account for up to 5 % of total EU GHG emissions in the form of CO_2 and CH_4, which is 25 times more potent than CO_2 as greenhouse gas.

Despite all this, sewage sludge and agri-food waste have great potential for energy production in the form of biogas. Biogas is considered to be amongst the most sustainable alternatives for the production of renewable energy from agri-food waste, through the application of different anaerobic co-digestion technologies, that offers buffer capacity, provide better nutrient balance, manage mixed wastes easily and improve fertilizer value of digested residues (Holm-Nielsen *et al.*, 2009; Jagadabhi *et al.*, 2008). Around 50 million tonnes of agri-food waste is produced each year in Spain, with a potential to generate 2.6 billion cubic metres of biogas (Probiogas Project), the equivalent of 4.2% of annual natural gas production. It has been calculated that 2.7 kWh/m^3 (Batstone *et al.*, 2002) of energy can be extracted from wastewater.

The application of co-digestion of sewage sludge and agri-food waste in small and medium sized WWTPs needs a change in the way both wastes are managed. The use of centralized digesters that covers the treatment for several WWTPs could be cost-effective feasible in rural areas with an economy based on primary sector. By this way, reliability and efficiency of anaerobic digesters will be improved for its application in small towns.

However, anaerobic co-digestion is a complex biological process which stability depends on the metabolic balance of different groups of microorganisms. Therefore, the addition of a readily biodegradable substrate, such as certain agri-food waste, can increase the speed at which some of the stages occur, thereby destabilising the system. Due to this, a control tool based on mathematical model is definitely useful for the optimization of the operation and biogas production prevent process failures in full scale digesters.

In this study a control tool of co-digestion in WWTP to achieve biogas production on demand based on modified IWA Anaerobic Digestion Model No. 1 (ADM1) is presented. ADM1 is a structure model that covers multiple steps of the anaerobic digestion of biomass in WWTPs (Batstone *et al.*, 2002), so it should be implemented the kinetic and stoichiometric coefficients for the simulation and modelation of co-digestion of sewage sludge and agri-food waste. The agri-food wastes co-digested in this study were orange waste and organic acids.

METHODS

<u>BMP tests:</u>

Biochemical methane potential (BMP) tests were carried out for primary, secondary sludges and for co-substrates individually, orange waste or organic acids, in 500 mL bottles, loaded with different concentrations of the corresponding substrate with a substrate/inoculum ratio of 0.5 kgVS/m^3. The inoculum was obtained from real scale mesophilic digesters treating the mixture of primary and secondary sludges and the mentioned co-substrates. Tests were conducted in Bioprocess Control® system during 15 days or until no biogas production occurred in thermostatic bath maintained at 35 °C. Results were plotted as CH_4 volume produced per substrate fed. All these assays were performed in triplicate and control tests, only inoculum, were conducted to discount CH_4 produced from remaining organic matter present in the inoculum.

<u>Continuous digesters:</u>

Two identical anaerobic digesters were employed and simultaneously operated, one treating primary and secondary sludge and the other the mentioned sludge and a mixture of orange waste and organic acids as co-substrates. They contained 1.5 m^3 of reaction volume and 0.5 m^3 of headspace. Both were mixed by sludge recirculation and were operated at 35 °C, with hydraulic retention time of 38 days and organic loading rate of 0.63 kg VS/ m^3/d. Operation was automatically controlled by temperature, redox potential and pH sensors and biogas produces was analysed online for CH_4, CO_2, H_2 and O_2.

<u>Analytical methods:</u>

Solids and chemical oxygen demand (COD) were measured according to Standard Methods for the Examination of Water and Wastewater (2017). Soluble fraction of COD was measured after centrifugation (4000 rpm, 15 min) and filtering of the supernatant by 0.4 µm. Biogas composition was determined by continuous biogas analyser (GA3000PLUS, Geotech). Volatile fatty acids (VFA) were measured by gas chromatography, lipids by gravimetric hydrolysis, proteins by titration and carbohydrate by calculation.

RESULTS AND DISCUSSION

ADM1 implementation in Ecodigestion tool

The ADM1 was implemented in Scilab as a differential and algebraic equation system in an easy to handle and windows base program. The implementation was made following the guidelines suggested parameters suggested by Batstone *et al.* (2002), with the exception of ones calculated and calibrated ad-hoc for the use of orange waste and organic acids as co-substrates. The simulation of the biodegradation of co-substrates was included in the ADM1 model to predict biogas production during a specific period. The expected biogas production was calculated by the model from the results obtained in BMP assays.

The developed tool based on ADM1 enables manage and control of the process. Therefore, the desired biogas or methane production to cover energy demand of the plant every 15 minutes is introduced into Ecodigestion tool. Then, the implemented model simulates the process to achieve the indicated biogas or methane production while the amount needed of each co-substrate is calculate. Once the co-substrates are dosage automatically considering the simulation results, Ecodigestion tool use a strategy to control the process based on the comparison between real and simulated data. Therefore, when real data differed from simulated one in either pH, biogas production or percentages of CO_2 and CH_4, modification in process parameters are made in order to avoid destabilization of the system.

Characterization of co-substrates and inoculum and parameter estimation

ADM1 requires several inputs to define the substrate and inoculum. For this reason, exhaustive COD characterization was made in this work for the primary and secondary sludge and co-substrates. The results obtained are included in Table 1. The substrates fed into the digesters, including primary and secondary sludge, were fractionated in their biodegradable and inert parts by using biodegradability data obtained in BMP tests. This was assumed as an approximation because BMP tests do not provide information to specifically determine the biodegradability of soluble and particulate organic matter separately.

Table 1. *COD characterization of primary and secondary sludge and co-substrates and BMP from co-substrates*

	Inoculum	Primary sludge	Secondary sludge	Organic acids	Orange waste
TS(%)	1.45±0.26	3.49±1.18	2.46±0.26	32.46±14.13	8.13±4.92
VS(%)	57.26±3.31	65.45±3.73	77.91±0.96	50.77±2.53	73.40±13.51
VFA (mg/L)	77±36	2562±1252	398±455	220400±25173	22930±3917
COD (g/L)	13±3	43±15	27±4	615±245	72±9
CODs (mg/L)	484±94	5484±2043	1352±1501	48400±7425	50200±8200
Protein (g/100g)	4.00±1.64	7.67±1.66	11.39±2.47	3.82±3.50	6.70±3.93
Lipids (g/100g)	3.85±1.81	6.86±4.91	4.73±1.95	4.00±1.67	4.25±1.50
Carbohydrates (g/100g)	0.44±2.78	8.24±4.01	3.07±3.48	164.65±72.72	64.49±53.09
Acetic acid (mg/kg)	155±396	2222±1423	550±457	137703±254470	8925±4532
Propionic acid (mg/kg)	49±209	1571±565	219±207	813±849	6738±6607
Isobutiric acid (mg/kg)	7±9	106±38	38±42	81±66	48±53
Butiric acid (mg/kg)	26±113	650±277	65±88	63830±50900	3444±3615
Isovaleric acid (mg/kg)	8±16	156±65	64±72	5±0	30±29
Valeric acid (mg/kg)	8±12	171±72	22±23	6±2	548±755
Isocaproic acid (mg/kg)	5±0	5±0	5±0	10±10	12±8
Caproic acid (mg/kg)	5±2	24±8	5±0	8±5	232±203
Heptanoic acid (mg/kg)	5±0	5±0	5±0	9±7	7±3
BMP (Nm^3/kg VS)		0.6	0.15	2.1	0.45

In order to fit the model to the experimental results, the simulation was undertaken to fit the outputs to the experimental data by changing the most sensitive parameters until finding the best values. Similar methodology was used to determine maximum speed of each metabolic group by varying the feeding of substrates and analysing metabolites evolution. A correction factor based on van't Hoff equation was used to consider the temperature influence, that considers the exponential trend of Arrhenius law.

Model validation

To assess the accuracy and applicability of the calibrated parameters in the anaerobic co-digestion of orange waste and organic acids with Ecodigestion tool, the model validation was undertaken basing on a comparison first between the target production of biogas and the model predictions and in a subsequent stage the comparison between experimental results and the calibrated model predictions with the previous calibrated parameters.

Ecodigestion tool sensibility was ascertained by comparing target biogas during a time of period and simulated one with and without determining co-substrate to be fed. There was good agreement of model predictions and constant target production even the model presents a slight overestimation with desired biogas production. Consequently, it was achieved an adjustment of 2.7 % in both circumstances, with and without considering co-digestion. When a variable target biogas production is considered, adjustment between desired biogas production and the model prediction increases until achieved a difference of only 2%, s it is shown in Figure 1.

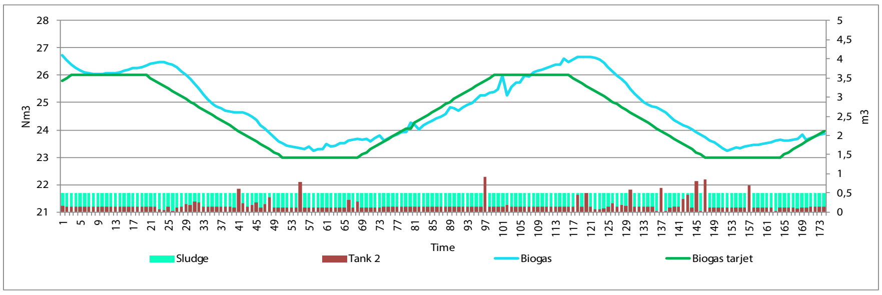

Figure 1. *Example of comparison between simulated biogas production and target biogas production*

The model with calibrated kinetic parameters was validated in mesophilic demo scale reactors. The mesophilic reactors (35 °C) were continuously fed with sock loads of co-substrates to cover peaks in biogas demand. As shown in Figure 2, the simulated biogas production values (one each 15 min) corresponded well to experimental data, ensuring biogas availability for satisfy energy needs of the WWTP even in peaks of demand.

Figure 2. *Example of comparison between target biogas production and experimental biogas production in continuous digesters*

CONCLUSIONS

Ecodigestion tool was demonstrated as a powerful tool for the control of anaerobic co-digestion to cover energy demand in WWTPs even in peaks of demand. Therefore, this tool is postulated as an alternative on the management of small and medium-sized WWTPs, enabling the reduction in operation costs and the recovery of the energy contained in the wastewater. Also, the model is useful to better understand the dynamic behaviour of anaerobic co-digestion of sewage sludge and organic wastes resulting from load variations as well as the temporally addition of high strength of waste.

ACKNOWLEDGMENT

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STUDY OF THE SUITABLE FEED DIET FOR DRY ANAEROBIC CO-DIGESTION BETWEEN SLUDGE FROM A WWTP AND TWO AGRO-INDUSTRIAL WASTES

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Abstract

This paper evaluates the use of two agro-industrial by-products, refined from fruit concentrate and wine lees, together with dehydrated sludge from a wastewater treatment plant with oxidation dams' scheme, to feed a dry anaerobic digestion process in order to assess the feasibility of co-digestion.

The wastewater treatment plants of small and medium-sized towns in the regions of Southwest Europe (Spain, Portugal, and Southern France) are characterised by a configuration based on oxidation dams which, with the current and future requirements of the legislation, are inefficient for sludge treatment. In this context dry anaerobic digestion is presented as a viable alternative for small and medium sized WWTPs. In order to implement the quality and quantity of biogas produced, as well as to provide a valorisation route for waste from the agri-food industry, dry basis anaerobic co-digestion is proposed.

To carry out the study, a series of mixtures were established between the dehydrated sludge from the WWTP serving the municipality of Frege-

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nal de la Sierra (Badajoz-Spain) and each of the aforementioned wastes. These mixtures have been introduced into individual anaerobic digesters to monitor digestion by measuring the quality and quantity of biogas produced daily.
This research is part of the CirCRuRaL4.0 project approved in the second call of the Interreg-SUDOE programme, co-financed with FEDER funds.

INTRODUCTION

The CircRuRaL4.0 project (www.circrural4.0.com) suggests a radical transformation of wastewater treatment in rural areas in line with a new conception of wastewater treatment based on the efficient use of resources.

The SUDOE area (Spain, South of France and Portugal) has hundreds of small and medium-sized towns located in rural environments where wastewater management and treatment still respond to the needs of the past, when the only priority was to guarantee the quality of the treated water. Usually, the WWTPs of small and medium-sized towns respond to a scheme based on oxidation dams (more than 1000 installations) where the aim is to stabilise the sludge in the aeration ponds themselves by means of sludge endogenesis.

Currently, one of the most common ways of managing sludge from a WWTP is field application for agricultural purposes, for which it is necessary to comply with Directive 86/278/EEC. The national legislation of several EU member states goes further by including limits to pathogens, so the criterion of sludge sanitisation is of particular importance.

One of the objectives of CirCRuRaL4.0 project is to improve the management of sludge from small and medium sized wastewater treatment plants, with the intention to promote the implementation of energy efficient technologies common in large WWTPs such as anaerobic digestion.

In this context, dry anaerobic digestion is presented, with number of advantages over conventional wet digestion, such as lower space requirements, lower investment costs and improvements in energy efficiency. In order to improve digestion performance and increase the amount of nutrients present in the digestate, sludge from extended aeration WWTPs will be co-digested together with agro-industrial co-residues.

In this work, different diets for dry anaerobic digestion of dehydrated sludge from the WWTP of Fregenal de la Sierra (Badajoz-Spain) together with, on the one hand, wine lees and, on the other hand, refined from fruit concentrate are analysed.

METHODS

In order to carry out the different tests, total volume 6 L stainless steel laboratory anaerobic digesters have been used, with smaller useful volume to leave space for the biogas generated.

To maintain a constant temperature during the process, digesters are covered by a hot water recirculating outer jacket, regulated by a thermostat. In this study, we have worked in the mesophilic temperature range (38°C).

The mixed substrate to be digested was kept completely homogenised by means of a central agitator, electrically driven and with adjustable speed by means of a potentiometer.

Biogas composition and volume generated was automatically *in-situ*monitored throughout the experiments thanks to an Awite System of Analysis Process 9 series analyser (Bioenergie GmbH, Germany). This analyser consists of two IR sensors that measure methane and carbon dioxide contents of the generated biogas, and three electrochemical sensors that measure hydrogen, hydrogen sulphide and oxygen content of the biogas. The system also has individual gas meters (Ritter model MGC-1 V3.2 PMMA, Germany) capable of measuring the amount of biogas produced in each test. The biogas produced is subsequently stored in Tedlar bags.

The initial characterisation of the 3 substrates to be used is shown in Table 1. Taking this into account, the different concentrations of the substrates are determined and presented in Table 2.

Table 1. *Initial characteristics of substrates used in dry anaerobic co-digestion.*

Parameter	Dehydrated sludge	Refined from fruit concentration	Wine lees
Total Solids, %	13.68	19.78	26.34
Volatile Solids, %	11.36	16.12	17.83
Ammoniacal Nitrogen, mg/L	629	400	569
C/N ratio	7.99	54.42	17.89

Table 2 .*Mixtures used for each experiment carried out.*

RESIDUE	Dehydrated sludge	Refined from fruit concentration	Wine lees
Amount Exp 1, g	1500	1000	-
Amount Exp 2, g	2125	375	-
Amount Exp 3, g	1176	-	784
Amount Exp 4, g	1875	-	563
Concentration Exp 1, %	60	40	-
Concentration Exp 2, %	85	15	-
Concentration Exp 3, %	60	-	40
Concentration Exp 4, %	75	-	25

All experiments also contained a portion of inoculum (1,500 g) of an active digestate. This ensures that the micro-organisms specific for methane production are present in the digestion medium, making the digestion faster and more efficient.

For this reason, control experiments (carried out in duplicate) containing a mixture of inoculum and water subjected to anaerobic digestion under the same conditions as the other experiments. In these experiments, the amount of biogas generated by the inoculum used was evaluated.

RESULTS AND DISCUSSION

Figures 1 to 4 (experiments 1 and 2) correspond to the results of the dehydrated sludge and the refining of the fruit concentrate by-product. **Figure 1** shows the amount of biogas produced for each experiment and its replication. Figures 2 to 4 show the biogas composition for methane, carbon dioxide and hydrogen sulphide respectively.

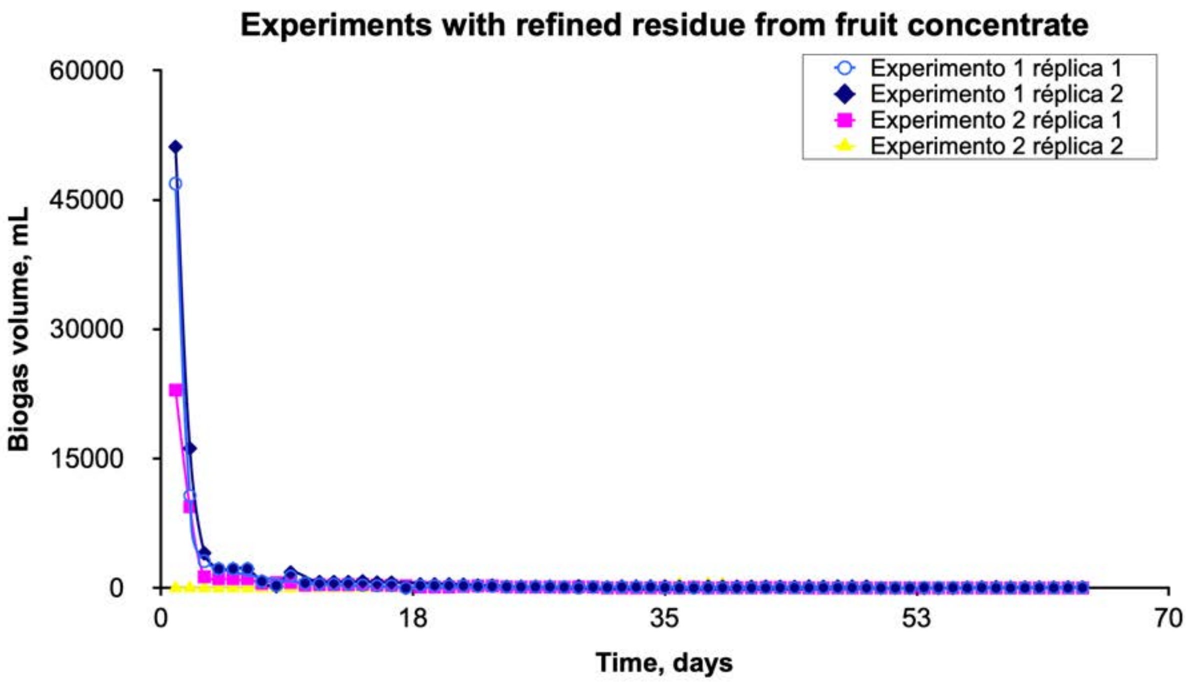

Figure 1. *Evolution of the volume of biogas produced for each of the replicates of the experiments developed with fruit concentrate refining residue.*

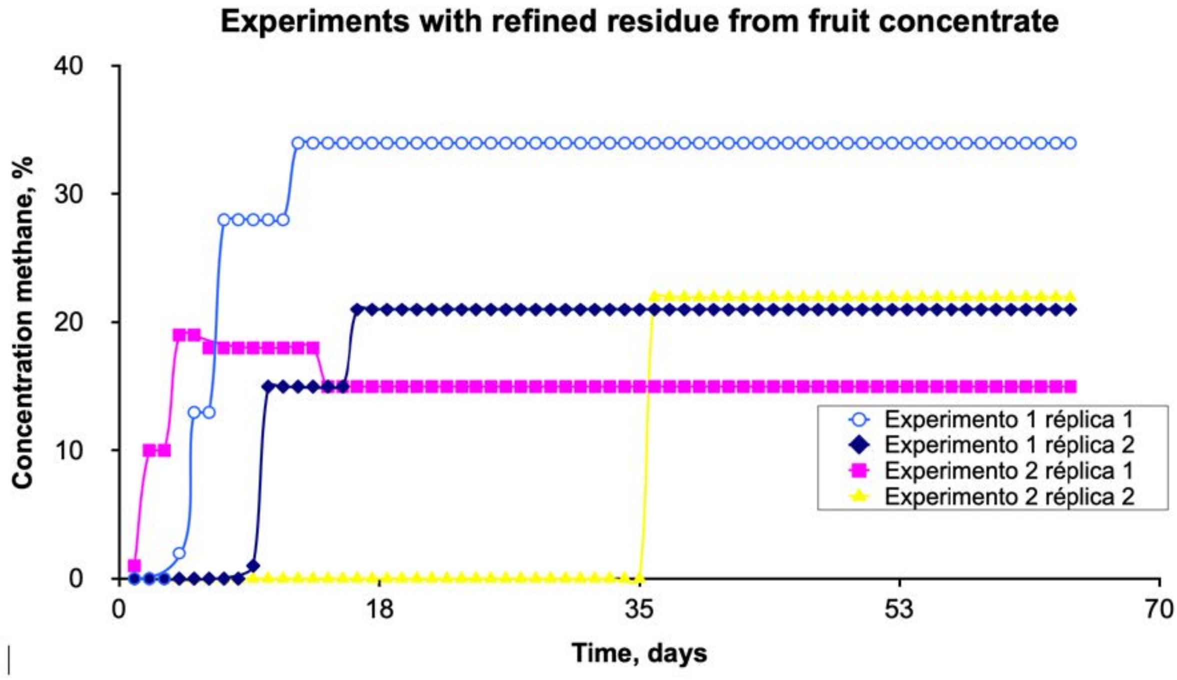

Figure 2. *Evolution of the concentration of methane produced for each of the replicates of the experiments developed with fruit concentrate refining residue.*

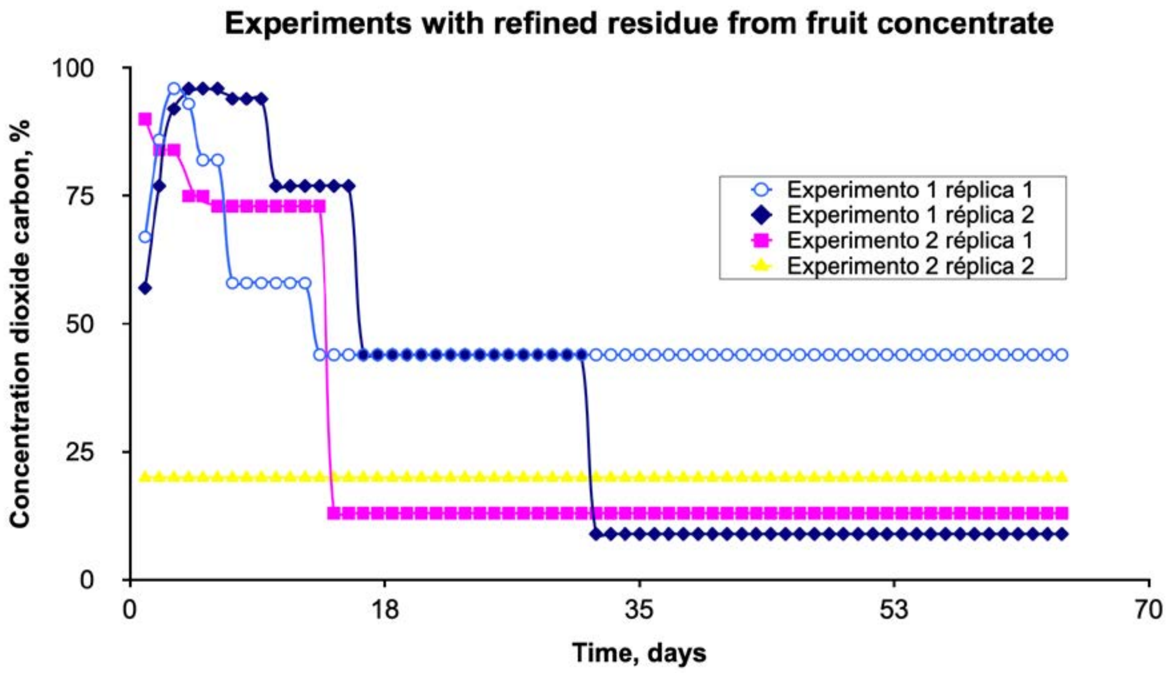

Figure 3. *Evolution of the concentration of carbon dioxide produced for each of the replicates of the experiments carried out with refining residue from fruit concentrate.*

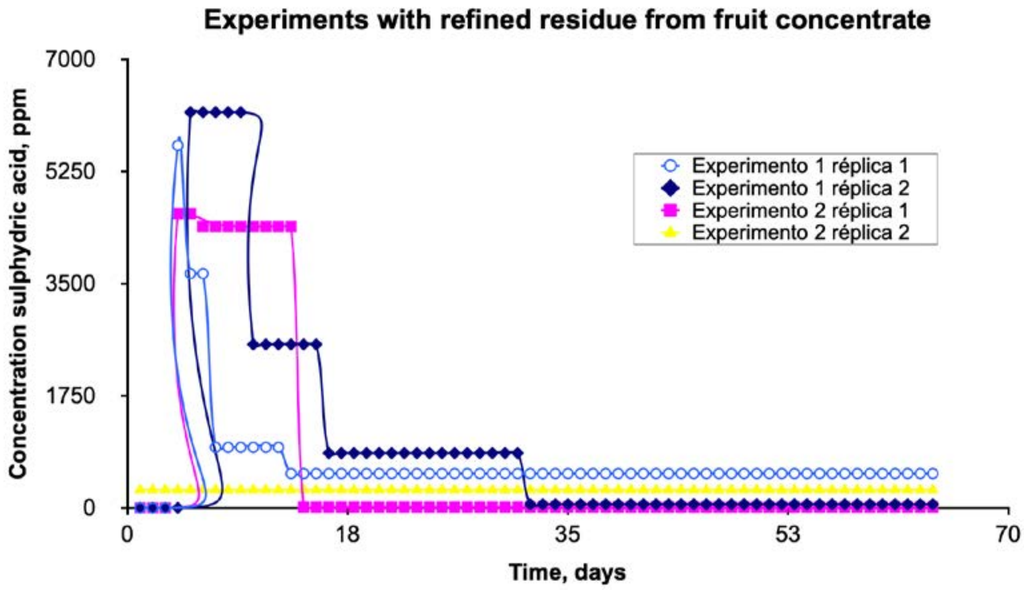

Figure 4. *Evolution of the concentration of sulphydric acid produced for each of the replicates of the experiments carried out with fruit concentrate refining residue*

The experiments carried out with fruit concentrate refining residue did not manage to reach the methanogenic stage in either of the two proportions studied (40 % and 15 % fruit concentrate refining residue), as the methane concentration was less than 50 % throughout the experiment (Figure 2). Moreover, the volume of total biogas obtained was not very high (Table 3), as from the first 20 days it declined in all cases evaluated for this waste (Figure 1). However, the values obtained for carbon dioxide and hydrogen sulphide were within normal values for this type of process, despite the high values of hydrogen sulphide obtained during the first 15 days. **Table 3** specifies both the characteristics of the mixtures evaluated and the methane yields obtained.

Table 3. *Initial characteristics and methane yields of experiments carried out with fruit concentrate refining residue*

Parameter/Experiment	Exp.1 rep.1	Exp.1 rep.2	Exp. 2 rep. 1	Exp.2 rep.2
Total Solids %	10.08	10.08	9.12	9.12
Volatile Solids, %	8.29	8.29	7.55	7.55
Ammoniacal Nitrogen, mg/L	336	336	372	372
CN Ratio	16.6	16.6	9.35	9.35
Methane yield, L/kg SV	0.171	0.138	0.156	0.091
Methane Yield, L/kg m.f.	6.2	4.64	1.65	0.093
Average methane concentration, %	33	20	15	22

Figures 5 to 8 correspond to experiments 3 and 4, those referring to the dehydrated sludge together with the wine lees. Figure 5 shows the amount of biogas produced for each experiment and its replication. Figures 6 to 8 show the biogas composition for methane, carbon dioxide and hydrogen sulphide respectively.

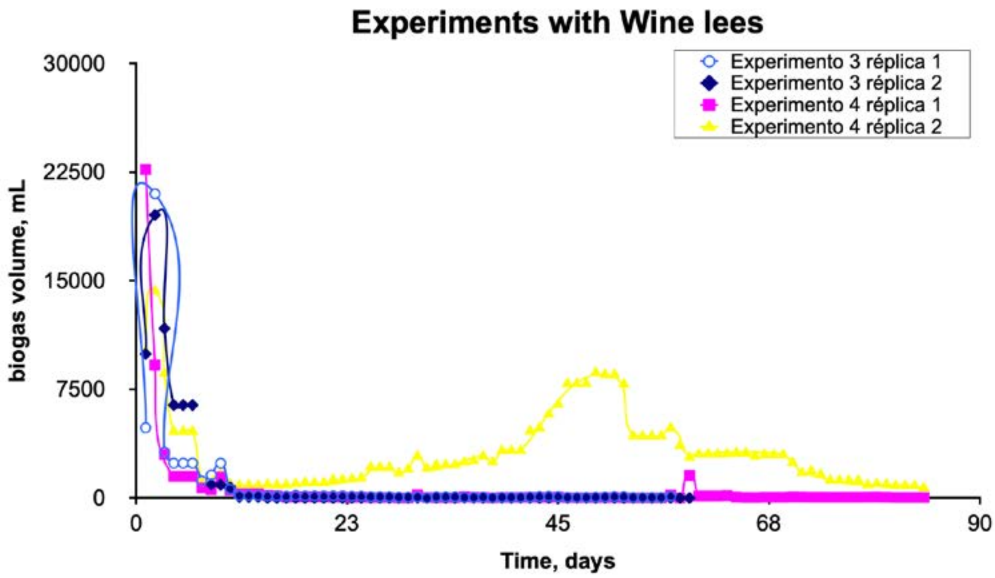

Figure 5. *Evolution of the volume of biogas produced for each of the replicates of the experiments carried out with wine lees.*

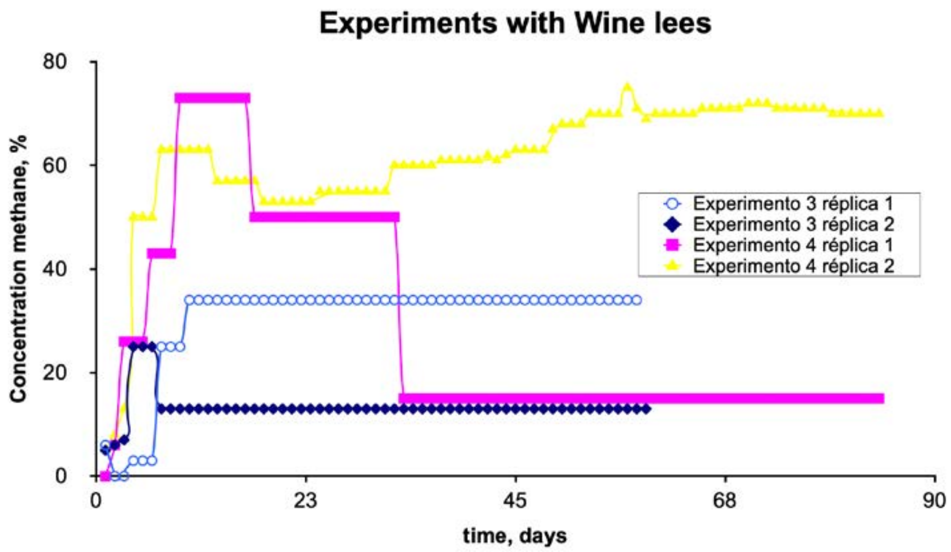

Figure 6. *Evolution of the concentration of methane produced for each of the replicates of the experiments carried out with wine lees.*

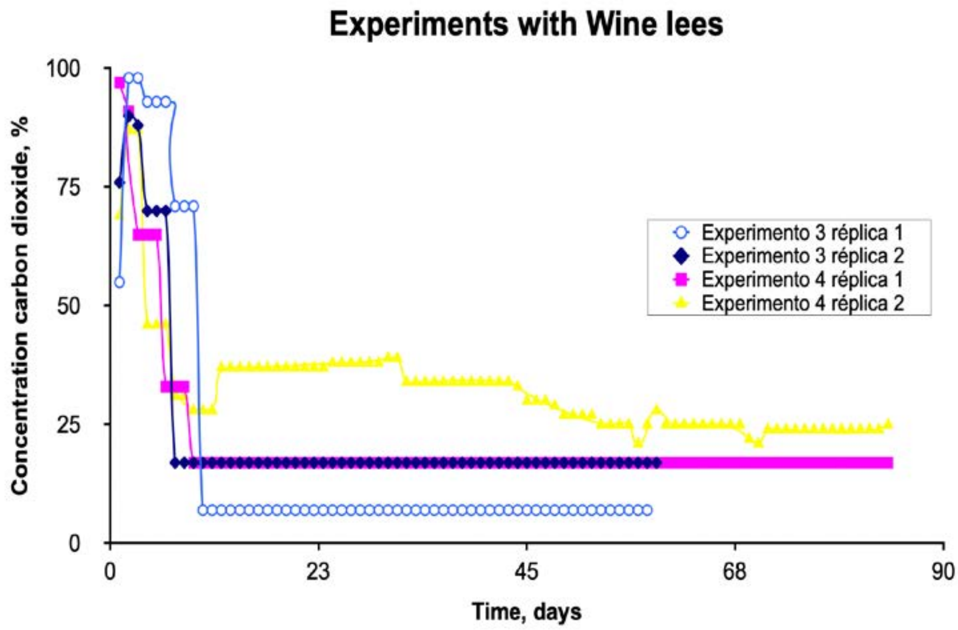

Figure 7. *Evolution of the concentration of carbon dioxide produced for each of the replicates of the experiments carried out with wine lees.*

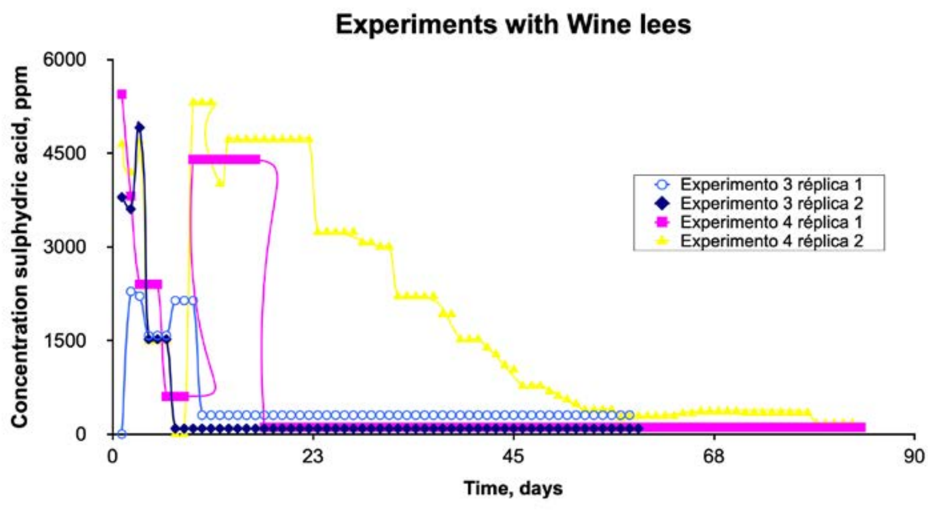

Figure 8. *Evolution of the concentration of hydrogen sulphide produced for each of the replicates of the experiments carried out with wine lees.*

The experiments developed with wine lees have determined that the only proportion that is viable in the anaerobic co-digestion process is the one carried out in Exp. 4 rep. 2 (25 % wine lees), as can be seen in Figures 5 and 6 referring to the evolution of biogas volume and methane concentration. The observed evolution of carbon dioxide and hydrogen sulphide concentration is similar to that obtained in experiments 1 and 2.

The starting data for experiments 3 and 4 as well as their methane yield are shown in **Table 4**.

Tabla 4. *Características iniciales y rendimiento de metano de los experimentos l levados a cabo con lías de vinificación*

Parámetro/Experimento	Exp.3 rep.1	Exp.3 rep.2	Exp. 4 rep. 1	Exp.4 rep.2
Parameter/Experiment	10.62	10.62	10.28	10.28
Total Solids %	7.90	7.90	7.96	7.96
Volatile Solids, %	343	343	381	381
Ammoniacal Nitrogen, mg/L	6.77	6.77	6.36	6.36
CN Ratio	9.315	20.872	18.494	450.989
Methane yield, L/kg SV	0.99	2.22	1,81	40.36
Methane Yield, L/kg m.f.	30	13	29	62

The pH and redox potential values obtained after the end of the anaerobic digestion process in the experiments Exp. 3 rep. 1 and Exp. 3 rep. 2 were 5.27 and -100 mV, and 4.71 and -96 mV, respectively. In the case of Exp. 4 rep. 1, a pH value of 5.39 and a redox potential of -255 mV were obtained. The other replicate carried out with lees in Exp. 4 rep. 2 obtained a final pH of 7.76 and a redox potential value of -421 mV.

CONCLUSIONS

Refined from fruit concentrationby-productis not suitable for dry anaerobic digestion. At concentrations above 15%, the acidity limits the bacterial growth suitable for the process.

In the case of the wine lees (Exp.3) it was observed that low pH values, probably due to the acidity of the co-residue itself, means that the digestion medium is not suitable for reaching the methanogenic stage.

For Exp.4, it was observed that one of the replicates (Exp.4 1,) obtained a pH value of 5.39, compared to the other, with a pH of 7.76, the latter value being suitable for the methanogenic microorganisms to develop perfectly, hence its high methane yield. This difference between the replicates could indicate that the studied proportion of 25% of wine lees together with dehydrated sludge is at the limit at which the microorganisms are able to degrade organic matter and thus carry out a good dry anaerobic digestion.

CASCADE FLASH (C&F) SYSTEM: THERMAL - PRESSURE SLUDGE PRETREATMENT

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Abstract
The technology called "*Cascade Flash (C&F)*" proposes a system designed to sanitize and hydrolyze sludge by injecting steam to sludge in a cascade-counter current device followed by a flash decompression. Thus, sludge is converted into a sanitized and highly bioavailable product. Due to its properties, this new substrate is ideal both for the formulation of biostimulants using enzymatic or fermentative technology, and for energy production through anaerobic digestion, since it has an increased biochemical potential for biogas production. This system is proposed as a means of sludge management applicable to small and medium-sized urban and agro-industrial wastewater treatment plants.

INTRODUCTION

Sewage sludge is generated in large quantities in the treatment of wastewater, being a problem of difficult management(Collivignarelli et al., 2015)both at national and international level, in particular due to the uncertain recovery/disposal future options. Therefore, it is clear that the development of new technologies that can mitigate the problem at the source by reducing sludge production is necessary, such as the European Directive 2008/98/EC prescribes. This work shows the results obtained with a thermophilic membrane reactor, for processing a biological sludge derived from a wastewater treatment plant (WWTP. Every year, 11.5 million tons (DM) of sludge are produced in Europe, and the cost of its treatment is around € 400/ton (MS).

Its agronomic use requires sanitization and stabilization pre-treatments, which can be composting or anaerobic digestion, among others. However, the former is a poorly controlled process and the latter is inefficient in terms of energy, requiring the development of new technologies aimed at enhancing this by-product. In this direction, new sludge valorization technologies through aerobic fermentation and enzymatic hydrolysis are being developed to obtain highly value-added biofertilizers.

The present developed technology allows sludge to be sanitized, increasing in turn its bioavailability, transforming it into a new substrate with biofertilizer potency and upper energy generation capacity.

The main purposes of this technology are to produce a sanitized product with higher bioavailability and solubility. Thus, sludge is transformed into a sanitized substrate easily metabolizable by plants and microorganisms that can be used for:

1. Energy production by anaerobic digestion.
2. Directly use as biofertilizer.
3. Substrate for producing plant growth promoting bacteria in bioreactor.

Thermal hydrolysis of sludge technology is widely used as a pretreatment to enhance the biochemical methane potency of sludge.Two examples of similar systems already in the market are Cambi's Thermal Hydrolysis and BIO THELYSTM from Veolia. Their purpose/function is to increase the biochemical methane potential of sludge in order to increase the anaerobic digestion yield. However, there are some differences. Firstly, *C&F* system works at softer conditions of temperature and pressure. The operational temperature is 120 °C, while the other technologies work at least at 160 °C. Other difference is that they are typically applicable only to large volumes of sludge not being applicable to small scale WWTP. Its small scale of the system allows coupling it to a renewable energy system, which would considerably reduce its energy consumption compared to other technologies.

Since conventional systems are developed for treating big volumes of sludge, they cannot be implemented in small WWTP, which have few options of managing this waste, resorting in the best of cases to composting, or, failing that, to landfilling. Thus, *Cascade Flash (C&F)* is a continuous pretreatment system thathas been designed to be implemented in small WWTP being proposed as a small-scale sludge management model in order to obtain locally applicable agronomical products, as well as for energetic production through anaerobic digestion.

RESULTS AND DISCUSSION

The *Cascade Flash (C&F)* system presented in this communication is a small-scale system for continuous thermal treatment of sludge, applicable to small and medium-sized wastewater treatment plants, which converts the sewage sludge into a sanitized substrate with a higher soluble content and higher bioavailability.This system arises as a result of the European H2020 project“*Water2Return*”, as a small-scale solution for the necessary sanitization-heat treatment prior to the fermentation process to obtain biostimulants from sewage sludge.This technology is currently under review to achieve ETV certification (*Environmental Technology Verification Program*, within the Eco-innovation Action Plan, promoted by the European Union). Environmental technological development uses these types of tools as a way to promote innovative and scientifically consistent technologies to the market. For this, the Verification Bodies are in charge of verifying the robustness of the claims made about technological innovation, and issuing a Verification Statement that serves as evidence of reliability, and facilitates access to the market of this technology, since they reduce the risk technological for your buyers.

Optimization of operational parameters: The operational parameters have been fine-tuned at the laboratory level. The lowest treatment temperature that achieves complete sterilization of the sludge has been sought, improving the solubility and hydrolysis of the organic matter. The results of these optimization tests were collected in a scientific publication (Caballero et al., 2020). Three working temperatures were evaluated, 120, 140 and 160 °C, finally selecting the lowest of the three temperatures.

Figure 1 shows solubilization degree of sludge after its treatment at different temperatures expressed interms ofTSS removal (%) compared to untreated sludge.Figure 2 and table 1 show the molecular exclusion HPLC chromatogram (215 nm) representingthe molecular weight profile of the soluble organic componentsof treated sludge with thermal hydrolysis and suddendecompression (THSD) at different temperatures (120, 140 and 160 °C) after each treatment compared with untreated sludge (WT) and autoclaved sludge (TH).

Figure 1. *Removal of TSS (%) under each experimental condition. TH: autoclaved sludge (120 °C); THSD 120, 140, 160: Thermal hydrolysis with sudden decompression at different temperatures.*

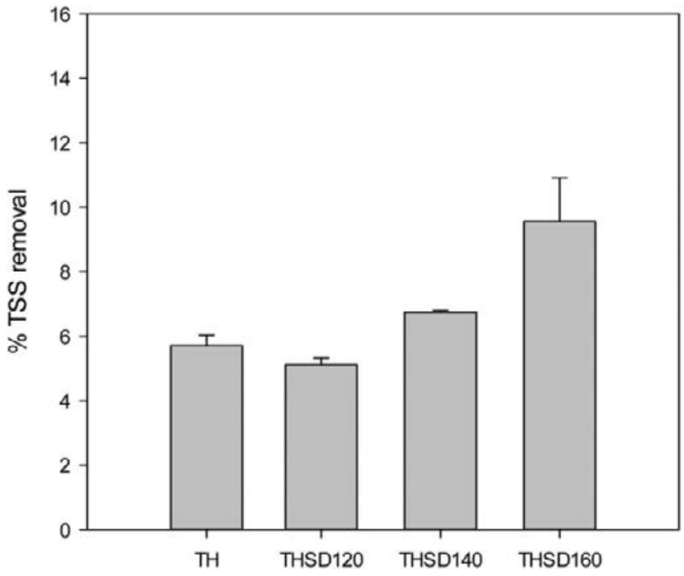
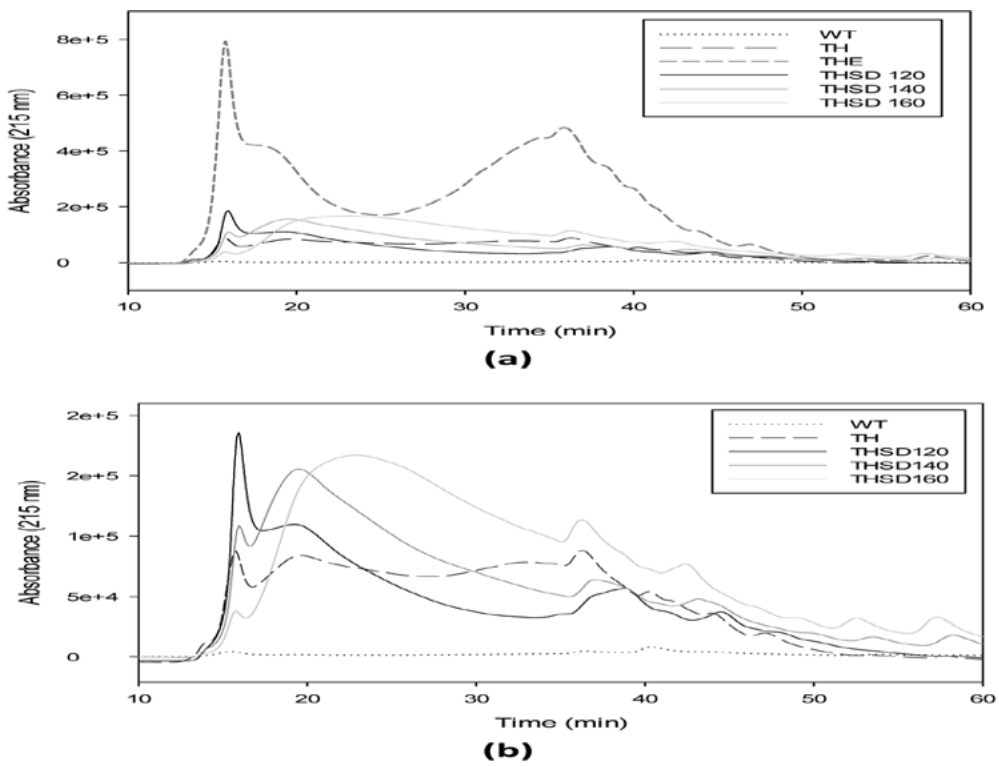


Figure 2. *Molecular exclusion HPLC chromatogram (215 nm) representingthe molecular weight profile of the soluble organic componentsof sludge after each treatment. WT: untreated sludge (120 °C); TH: autoclaved sludge; THSD 120, 140, 160: Thermal hydrolysis with sudden decompression at different temperatures.*



Molecular weight (kDa)	WT (%)	TH (%)	THSD_120 (%)	THSD_140 (%)	THSD_160 (%)
>5	ND	37.8	50.8	48.0	36.5
3–5	ND	5.09	4.19	5.33	6.44
1–3	ND	11.6	7.25	9.67	12.7
<1	ND	45.7	37.8	37.0	44.33
0.3–1	ND	13.9	6.94	8.32	11.4
<0.3	ND	31.7	30.8	28.7	33.0

Table 1. *Molecular weight distribution of the soluble organic fraction of treated sludge after applying the different treatments. ND: not detected; WT: untreated sludge (120 °C); TH: autoclaved sludge; THSD 120, 140, 160: Thermal hydrolysis with sudden decompression at different temperatures.*

Design of the Cascade Flash (C&F) system: The innovative aspect of the *C&F* system is the cascade flash device that is a continuous sludge pre-treatment device. It is composed in the top by a cascade mixer that is an inverted funnel system that allows mixing the sludge with counter current steam, quickly reaching operational parameters, and a reservoir at the bottom side that maintains pre-treatment conditions during a short period. It also includes valves that release the pressure suddenly, achieving the sanitization of sludge continuously, making microbial spores unfeasible, and breaking cellular structures (cover and internal membranes of the microbial cells), improving the solubility and bioavailability of the sludge, giving rise to a product of greater added value with which to formulate agronomic compounds and also increase biomethanisation yield.

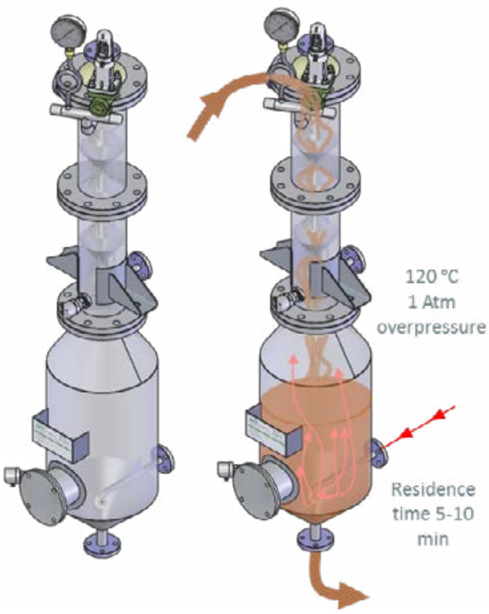


Image of the C&F device

The sludge treatment process where the C&F device is integrated is described below (Figure 3).Firstly, sludge is preheated in the buffer tank (Figure 4, A) up to a temperature of 80 °C. Here, sludge will be waiting to enter in the pretreatment system. When entering in the pretreatment system, sludge pass through the preheater (Figure 4, B) at a flow rate of 0.18 m^3/h, increasing its temperature from 80 to 90 °C, and then, it pass through the C&F device (Figure 4, C), reaching 120 °C. Here, in the bottom side of the C&F, it will remain for a short period of time (5-10 minutes). After that,pressure is released and sludge pass to a flash vessel (Figure 4, D) being subjected to a rapid decompressionthat produces a sudden evaporation of intracellular water, leading to a hydrolyzed and sanitized product.

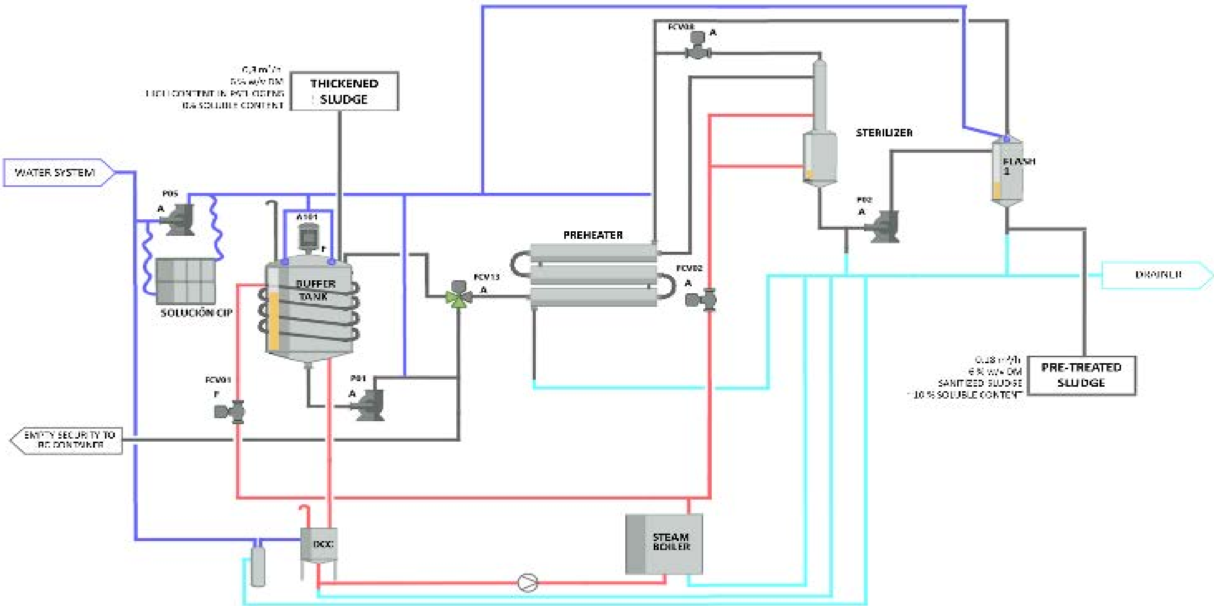


Figure 3. *Figure shows the image of the C&F whole process.*

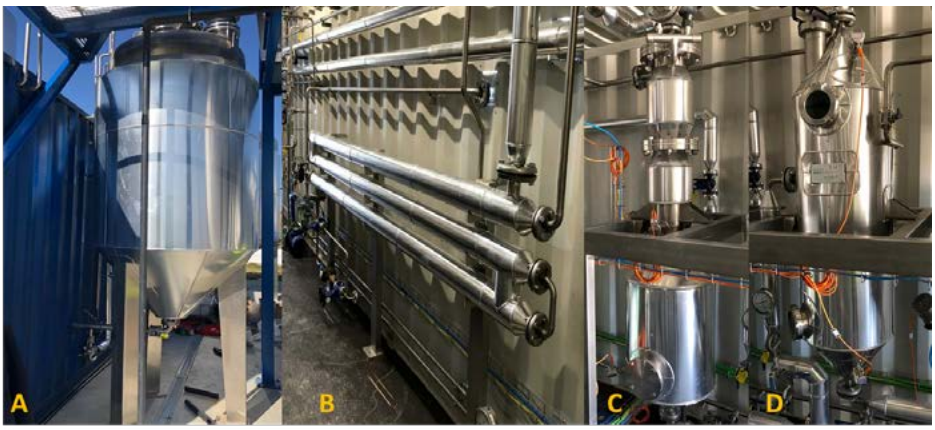


Figure 4. *The figure shows the different systems that integrates the pretreatment system. A – Buffer tank; B – Tubular heat exchanger; C – Cascade Flash device; D – Flash vessel.*

Although the equipment is scalable, the current prototype presents an operation flow of 0.18 m^3/h, operating in continuous regime.The hydrolysis temperature (120 °C) is reached gradually. In the buffer tank,sludge is heated until 80 °C, in the preheater it is heated until 90 °C, and in the sterilizer it is quickly heated until 120 °C.

Sludge has a residence time of 5-10 minutes in the C&F device, during which operational conditions (120 °C and pressure associated to this temperature) are maintained. After this residence time, sterilizer content is completely emptied to flash1 tank, reducing temperature to 100 °C and releasing the pressure.

C&F technology, applied as a thermal pretreatment of sewage sludge, achieves:

1. Total sanitization of the product by removing 100% of viable microorganisms and spores in sludge.
2. Increase the solubilization of total dry matter by at least 8%.
3. Hydrolysis of the organic matter resulting in a soluble content at least 45 % composed of under-1 KDa molecules.

One negative environmental aspect is that the process is not energetically optimized, relying in a electric steam boiler to generate steam to heat sludge. However, renewable energy systems could be adapted to reduce the consumption. The parabolic cylindrical collector technology would be ideal for this purpose, being able to considerably reduce the steam requirements to heat the sludge.

Given that there is a demand on the market for cheap and effective technologies for sewage sludge pretreatment applicable on a small scale, as small WWTPs do not have solutions available to manage their sludge, C&F technology could be established meeting this demand.

CONCLUSIONS

A continuous system for treating sludge using sudden decompression thermal hydrolysis has been designed. This system is proposed as a means of sludge management applicable to small and medium-sized urban and agro-industrial wastewater treatment plants. C&F system converts sludge into a sanitized and highly bioavailable substratewith biofertilizer potency and upper energy generation capacity.

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MODELING OF SEWAGE SLUDGE DEWATERING IN DRYING BEDS, ANGOLA CASE

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Summary
Sludge dehydration is an essential step in the wastewater treatment process. The main objective of this study was to contribute to the development of a mathematical model that describes the sludge dewatering of a Wastewater Treatment Plant in Drying Beds, based on the climatic conditions of Angola. A pilot installation consisting of six sludge drying beds and a portable weather station was designed, built and operated. During the experimental part, two cycles of sludge dehydration were carried out, with sample collection for laboratory measurement of parameters of interest. During the process, there was a greater removal of water by drainage and evaporation, consequently reducing the thickness of the sludge layer and increasing the concentration of total solids (TS) as a function of the time of exposure to air. In beds with a lower initial thickness of the sludge layer, the TS concentration increased from 5.6 % to 84.49 %.
Keywords: sewage sludge, drying beds, monitoring, modeling, sludge treatment

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INTRODUCTION

In developing countries, such as Angola, more than 90 % of all wastewater produced is not treated or has an appropriate treatment. To reverse this situation, it will be necessary to increase coverage in basic sanitation, with the expectation of building more wastewater treatment plants in the country. However, the treatment of sewage another great problem, the management of sludge. The environmental legislation of some countries already requires the technical specification of the final destination of the sludge in the licensing processes. Sludge dewatering, is an essential step in the wastewater treatment process, since the reduction volume, translates into easier management, lower transportation, landfill and incineration costs. There are several technologies for treating sludge. Natural techniques, such as drying beds (DB), are simple processes with low operating, energy and installation costs, but require more space compared to mechanical technologies, Metcalf & Eddy, (2003).

MATERIALS AND METHODS

The experimental study of the dewatering of WWTP sludge in drying beds was carried out in Luanda – Angola for the following reasons: favorable climatic conditions, low operation/maintenance cost, cheap and available space.

The experimental study was carried out in five stages:

First stage – foundations building and waterproofing of the site;

Second step - acquisition and assembly of circular deposits (0.12 m^3 in volume and 0.2 m^2 of internal surface), obtaining the draining medium of the drying beds with the appropriate granulometry, installation of pipes and assembly of the metallic structure to support the beds. For Sperling (2014), the drying beds can be circular or rectangular units, consisting of gravel;

Third stage - installation of a portable meteorological station for measurement and recording of meteorological parameters (temperature, solar radiation, air humidity and air speed);

Fourth stage - sampling of raw sludge (RS) at the Sequele WWTP (Luanda - Angola) on the solid phase treatment line, at the sludge equalization tank discharge valve;

Fifth stage – Beginning of the operation, monitoring of the system in the two sludge dehydration cycles and a series of experimental tests for a period of two months. The samples of dehydrated sludge (DS) and the percolated liquid during the process were analyzed at the Separation Engineering, Chemical Reaction and Environment Laboratory (LESRA), according to the analytical procedures defined in the Standard Methods for the Examination of Water and Wastewater (APHA, 1998).

Table 1 shows the type and granulometry of the filling material chosen for the drying beds and Figures 1 and 2 illustrate the construction and operation steps of the pilot installation.

Table 1. *Filling material for the drying beds.*

Quantity (kg)	Layer	Height (cm)	Tanks
240	Bottom	20	All
120	Intermediate	10	All
80	Higher	10	Tk1, Tk2, Tk3 and Tk4
81	Higher	20	Tk5 and Tk6

Figure 1. *Waterproofing the site, starting the pilot experiment and collecting the samples: dehydrated sludge and water drained from the system.*

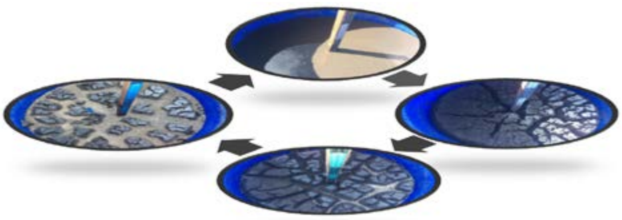

Figure 2. *Reduction in the thickness of the sludge layer inside the bed and concentration of TS, (I and II cycles) .*

RESULTS AND DISCUSSION

Two cycles of dehydration were carried out: the 1st cycle from October 14 to November 14, 2019 and the 2nd cycle from December 22, 2019 to January 22, 2020. During the monitoring of the process, meteorological parameters were measured for the two cycles of dehydration. The average meteorological values obtained are shown in Table 2.

The meteorological parameters have a great influence on the sludge dehydration in DB. In the 1st cycle, the parameter values are lower than in the 2nd cycle. This fact shows that during the process there was some climatic variation. In this context, it can be inferred that the climatic conditions in Angola are favorable for the treatment of sludge in DB and the sanitation problem can be solved using the available natural means.

Table 2 - Amounts weather average recorded during the experiments (I and II cycles).

Parameters	First cycle	Second cycle
Temperature (°C)	30.2 ± 0.5	31.7 ± 0.5
Solar radiation (w / m ^{2})	457 ± 22 , 1	993 ± 29 , 1
Relative air humidity (%)	65.5 ± 1.3 0	65.6 ± 1.6
Wind speed (m / s)	1.28 ± 0.11	1.58 ± 0.19

During the tests, some control parameters of the pollutant load (pH, ST, COD, Total Nitrogen, Total Phosphorus and Total Coliforms) were measured in RS and DS samples. The average values of these parameters during the I and II dehydration cycles are shown in Tables 3 and 4, respectively.

Table 3 - Control parameters and performance evaluation of the DS for the 1st cycle.

Parameters	Gross mud from the 1st cycle	Dehydrated sludge from the 1st cycle
ST (%)	5.4 ± 0.1	42.4 ± 2.9
pH	7.7 ± 0.1	7.2 ± 0.2
CT (MNP / 100 mL)	> 240	49 ± 14 , 1
COD (mgO $_2$/ L)	33 355 ± 5	104 ± 16
Total Phosphorus (mgP / L)	206 ± 0.1	----
Total Nitrogen (mgN / L)	805 ± 0.1	----

Table 4 - Control parameters and performance evaluation of DS for the II cycle.

Parameters	Gross sludge from the second cycle	Dehydrated sludge from the second cycle
ST (%)	5.6 ± 0.11	84.5 ± 3.3
pH	7.9 ± 0.1	7.4 ± 0.5
CT (MNP / 100 mL)	> 240	47 ± 14
COD (mgO $_2$/ mL)	34 314 , 2 ± 4	94 ± 16
Total Phosphorus (mgP / L)	208.6 ± 0.1	----
Total Nitrogen (mgN / L)	804 , 4 ± 0.1	----

The results obtained from the control and performance evaluation parameters of the LS are satisfactory. The drying beds were efficient in removing the total polluting material and increasing the solids content of the sludge. In the 1st cycle, the TS concentration increased from 5.4% in the raw sludge to 42.4% in the dewatered sludge and in the II cycle, the TS concentration increased from 5.6% in the gross sludge to 84.5% in the dewatered sludge.

During the time of exposure to air, LSs with a lower thickness of the initial mud layer had higher performance in the process compared to LSs with a greater thickness of the initial mud layer. Dehydrated sludge must not remain in LS with TS greater than 35%, because it promotes the growth of vegetation and hinders its removal (Andreoli et al, 2007).

The mean values of the drained water parameters, including the TSS value of the pilot system during the I and II cycles, were compared with the VLE in the discharge of effluents, based on Angolan legislation (Presidential Decree 261/11, 2011), as presented in Table 5.The mean values of drainage water parameters including the value TSS pilot system during the cycles I and II were comp plows with ELV in discharge effluents are based on Angola legislation (Presidential Decree 261/11 2 011) as features of n with Table 5 .

During the experimental process, it was found that the results obtained from the drained liquid satisfy the EVL d of the Angolan legislation in force for discharging effluents into the water environment. This means that dewatering sludge in drying beds can be done without causing disturbance to the ecosystem.

Table 5 - Comparison of the values drained at the outlet of the system with the emission limit values (ELV) in the effluent discharge (Angolan legislation).

Parameters	Drained (cycle I)	Drained (cycle II)	ELV
pH	7.13 ± 0.3	7.12 ± 0.5	6.5 - 8.5
CT (MNP / 100 mL)	9.9 0 ± 3.7	9.7 0 ± 3.4	----
SST (mg / L)	6.6 0 ± 1.1	5.21 ± 1.8	60
COD (mgO $_2$/ mL)	103.73 ± 16	87.28 ± 15.5	150
Total Phosphorus (mgP / L)	3.12 ± 1.1	2.79 ± 0.7	3.0
Total Nitrogen (mgN / L)	13.5 0 ± 2.2	12.92 ± 1.9	15

CONCLUSION

The performance of the pilot system in the treatment of sludge during the two dehydration cycles was quite satisfactory. The total solids content of the sludge went from 5.6% in the raw sludge to 84.5 in the dehydrated sludge in the II dehydration cycle. It was also found that the main pollution parameters analyzed in the drained liquid are in accordance with the discharge limit values defined in Presidential Decree 261/11, of October 6, 2011. The process of sewage sludge dewatering in DB, was efficient in considerably reducing the total polluting matter of the dewatered sludge and the drained liquid, giving them greater quality for recovery. Thus, the use of this technology with low costs for sludge treatment in small towns and rural areas, can be justified in other regions of Angola, with similar climatic conditions to solve the problem of sludge recovery.

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COMPARISON OF THE TECHNICAL EFFICIENCY OF VERMICOMPOSTING WITH RESPECT TO THE MOST WIDELY USED PROCESSES FOR THE STABILIZATION OF SEWAGE SLUDGE IN MEXICO

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Abstract
In Mexico, a limited number of wastewater treatment plants (WTP) stabilize their sludge, generally disposing the sludge in open dumps and drains, constituting a severe environmental problem. The research evaluates the technical efficiency of the vermistabilization of sludge from domestic wastewater, with aerobic and anaerobic methods, with the objective of offering a viable option for designers. The evaluation highlights the operability of the stabilization methods. The methods are compared using indicators estimated through sludge and process characteristics. The method for defining the degree of efficiency of the processes is the Banker Chames and Cooper data envelopment analysis, based on the indicators, stabilization percentage, quality, retention time, energy expenditure, required area and stabilized sludge production. The results indicate that vermicomposting is the most efficient technique, reporting the highest number of efficient indicators, only retention time and required area were not efficient. In order to have an overall view of the processes, their standard deviations with respect to the efficient point were estimated; anaerobic is the second efficient technique with a deviation of 0.7

and lastly aerobic with 0.74. The conclusion is that of the three methods compared, sludge vermistabilization is the most efficient technique for domestic WWTP flows of less than 80L/s, which is relevant since 31% of the WWTPs in Mexico correspond to this scale.

INTRODUCTION

A secondary product resulting from water purification is sewage sludge, of which, in Mexico there are no exact figures regarding its generation (Mantilla et al., 2017). NOM-004 (SEMARNAT, 2003) and Article 148 of the National Water Law (LAN, 2020) indicate that sludge and biosolids, must be treated for final disposal, due to their polluting potential, so they need a stabilization process (Oropeza, 2006). However, the management of these wastes is a little attended aspect, since few treatment plants include this process, frequently their destination is direct discharge into the sewage system (Amador et al., 2015). In this context, a strong environmental problem arises in Mexico, due to the pollution and volume produced. Among the stabilization methods used in our country are composting, alkaline stabilization and the most common aerobic and anaerobic stabilization (CONAGUA, 2014). There are alternative technologies such as vermicomposting, which has been proven to have an acceptable efficiency in sludge stabilization, in addition to generating several valuable by-products (Reyes et al., 2020). The application of vermicomposting is scarce, due to the fact that there is no comparative study of technical efficiency and cost efficiency in relation to the most usual techniques, which would allow WWTP designers to discern its feasibility. The importance of the efficiency analysis lies in having a systemic view of the capacity of the different methodologies to obtain better results, depending on the resources and productive processes of each technique.

METHODS

This research applies the data envelopment analysis (DEA) method that has been used in obtaining technical efficiency of treatment plants (De la Vega, 2012) and that allows evaluating each of the efficiency indicators separately. The BBC (Banker Chames and Cooper, 1984) model (Emrouznejad and Cabanda, 2014) is used which assumes yields in input and output orientation. The relative efficiency of an evaluated unit is measured relative to the remaining units, which operate on a similar scale. In the DEA-BBC, technical efficiency highlights the capacity of a production unit (PU) to obtain the maximum output from a given set of inputs, which is obtained by comparing the observed value of each PU with the optimal value defined by the estimated production frontier (Coll and Blasco, 2006).

The main objective of the DEA-BBC model is to determine the efficient frontier of the indicators in order to analyze the stabilizations and determine their technical efficiency. To run the DEA-BBC model, the following steps are followed:

1. The inputs and outputs of all the established indicators are quantitatively determined.
2. The inputs (inputs) are plotted on the "x" axis and the outputs (outputs) on the "y" axis. The empirical production frontier (efficient frontier) is proposed, which determines the indicators with the best performance.
4. If the technique has no other technique above or to the left on the graph, it is considered efficient, otherwise it is determined to be inefficient.
5. Efficiency index calculations and increases or decreases in output are made. To corroborate the result, the Excel Solver optimization mathematical tool is used to obtain the efficiency index, which, if it is different from one, means that it is not efficient with respect to this indicator. This is done with each of the indicators and the results are verified.

It is essential to emphasize that the technical efficiency thus defined is only considered efficient with a value of one. A score close to zero or much higher than one should be understood as meaning that the unit being evaluated is far from being technically efficient. In other words, the dispersion of results with respect to one reflects inefficiency considering the maximization or minimization of the indicator analyzed (Coll and Blasco, 2006).

RESULTS AND DISCUSSION

Three stabilization trains are proposed for a domestic wastewater flow of 20L/s, equivalent to 1730m^3/day, with a minimum flow of 864m^3/day and a maximum of 2590m^3/day. The WWTP used as a model has a pre-treatment process, a primary settling tank, a conventional activated sludge system, a secondary settling tank and a chlorination disinfection process. The characteristics of the sludge are shown in Table 1. The aerobic stabilization train includes aerobic stabilization, gravity thickening without chemical conditioning with sludge recirculation to the stabilization reactor and dewatering by drying beds. In the anaerobic stabilization train, gravity thickening, anaerobic stabilization and dewatering by drying beds were considered. For the vermistabilization train, gravity thickening and vermistabilization were established. Sixty-five beds of 18 meters long by 1 meter high and 1 meter wide are required.

Table 1. *Characteristics of the sludge generated in the model WTP (CONAGUA, 2014).*

Parameter	Unit	Secondary Sludge
Average flow rate	m^3/d	26.1
Suspended solids	g m^3	10 000
Volatile solids	%	55.5
BOD	g m^3	1810
Soluble BOD	g m^3	1.86
COD	g m^3	8320
Soluble COD	g m^3	2.79
Soluble NTK	gN m^3	0.885
NTK	gN m^3	556
Ammonia nitrogen	gN m^3	0.885
Nitrites	gN m^3	0
Nitrates	gN m^3	29.9
Total phosphorus	gN m^3	6.43
pH		7.6
Cations	g m^3	160
Anions	g m^3	160
Settleable solids	mL/L	0
Fats and oils	g m^3	0
Summer temperature	°C	23
Winter temperature	°C	10.2

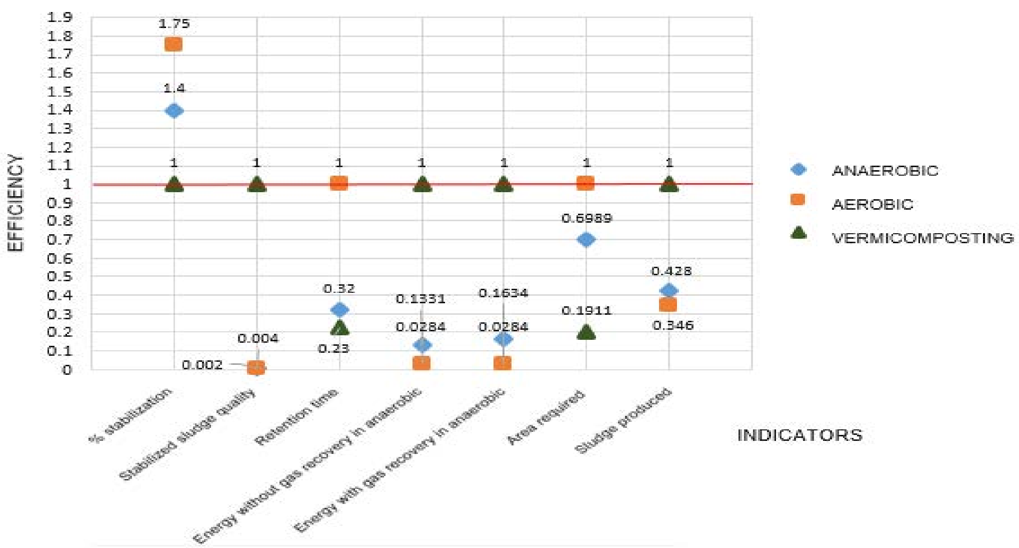
Table 2 shows the technical efficiency of the indicators evaluated for each stabilization. Figures other than one indicate that the methodology is not efficient with respect to the other techniques examined. That is, the efficiency values show the minimum combination of inputs to fulfill the maximum function. Being different from one, they describe inefficiency. The optimal analysis would be found in the methodology that obtained all the indicators with an efficiency value of 1, but this did not occur. Graph 1 shows the indicators with respect to each of the stabilizations. In this case, vermistabilization is the technology closest to efficiency with five efficient indicators out of the seven indicators evaluated; it is not efficient in relation to area and retention time, both indicators very far from efficiency. In second place is aerobic stabilization, which is efficient in terms of retention time and the area necessary for the process. The latter shows to be far from technical efficiency in the indicators of stabilization percentage, sludge quality, energy expenditure and quantity of stabilized sludge.

Table 2. *Technical efficiency of the analyzed indicators obtained by DEA method evaluated by optimization (linear programming) (Own elaboration).*

INDICATOR ANALYZED	TECHNICAL EFFICIENCY OF STABILIZATION		
	ANAEROBIC	AEROBIC	VERMICOMPOSTING
Sludge stabilization	1.400	1.750	1
Stabilized sludge quality	0.002	0.004	1
Retention time	0.320	1	0.2300
Energy expenditure without gas recovery in anaerobic stabilization	0.1331	0.0284	1
Energy expenditure with gas recovery in anaerobic stabilization	0.1634	0.0284	1
Area required for the process	0.6989	1	0.1911
Sludge produced in stabilization	0.4280	0.3460	1

Values equal to 1 indicate efficiency

Graph 1. *Overall technical efficiency (Own elaboration)*



Lastly, we have anaerobic stabilization, which is not efficient in any indicator. However, it is important to mention that in all indicators it is not the farthest from efficiency, as in the case of aerobic stabilization in the indicators of stabilization percentage, sludge quality, energy expenditure and amount of stabilized sludge. Neither in the necessary area and retention time, where vermicomposting is the farthest away. Given this behavior, a measure of dispersion is used for the efficient technique, i.e. the one that has all the indicators with an efficiency of 1. To do this, the standard deviation is estimated with respect to the value of one, the results indicate the dispersion of each technique from the optimum efficiency, which would register a standard deviation of zero. The results indicate that vermicomposting has a deviation of 0.41, anaerobic composting 0.70 and aerobic composting 0.74. Therefore, the second recommended technique is anaerobic, followed by aerobic. From the perspective of this method, vermistabilization is recommended to be applied in sludge stabilization processes, because it obtains the highest amount of efficient indicators and has a lower standard deviation.

CONCLUSIONS

The overall analysis of the seven indicators corresponding to each technique shows that sludge stabilization by vermicomposting is the most efficient technique, followed by anaerobic stabilization and finally aerobic stabilization. The two indicators where vermicomposting was not efficient are the retention time and the area necessary for the process. The retention time considered is the minimum necessary for the sludge to comply with NOM 004; however, if a better quality product is required and a stabilization percentage of 99.98% is reached, the time must be increased and consequently it will have to be considered for the design. The area required for the process is a disadvantage for this technique, especially if the WTP does not have the available space. As the flow rate and organic matter content increases, the area required will increase. It is concluded that sludge vermicomposting is an efficient and simple technique, recommended for domestic wastewater flows of less than 80L/s. In Mexico, of all existing WTPs, 31% have similar flows, so sludge vermicomposting is highly recommended for these treatment plants. For flows greater than 80L/s, it is necessary to take into account that the increase in the amount of sludge to be stabilized will require a larger surface area, as well as technological advances to make the process more productive. These improvements include automation in the processes of feeding, irrigation, control, biomass recirculation and beds with other characteristics. Finally, due to the various characteristics mentioned above that govern the vermicomposting process, it is essential to perform a cost-efficiency analysis, in relation to its marketable products, in order to discern whether vermicomposting is a technically effective methodology at low cost, in relation to anaerobic and aerobic stabilization.

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EFFECT OF OPERATING TEMPERATURE ON YIELD AND METHANE PRODUCTION IN THE ANAEROBIC CODIGESTION PROCESS OF SLUDGE: VINASSE: POULTRY MANURE

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Abstract

The anaerobic digestion of wates and agri-food by-products allows the energy recovery by obtaining biogas with a high calorific value. This assay studies the effect of temperature in the anaerobic co-digestion of sludge from wastewater treatment plant together with wine vinasse and poultry manure, to improve the methane production in the sewage sludge anaerobic digestion process. Two temperatures, 35°C and 55°C usingab stirred lab reactors are tested and methane production and purification performance are analyzed. The experimental results show that the highest purifying yields are reached in the thermophilic range, with values above 70% total solids removal (TS) and a maximum production of 0.7 L / day.

INTRODUCTION

The increase in treatment plants entails the constant growth of sludge that requires new alternatives forsustainable management and with direct applicability in small and large plants. The anaerobic digestion processallows managing the high amounts of organic by-products such as sludge currently generated in wastewater treatment plants (WWTPs), generating energy as well as a final waste that could be used for agronomic purposes. It has been shown that the anaerobic digestion of sludge individually does not lead to good results, requiring the addition of other by-products that supply nutrients that complement the waste characteristics andmake it more optimal for the anaerobic degradation process (Lee et al. 2009). The productivity of anaerobic digesters can be enhanced and improved by adding readily biodegradable co-substrates.

In the present work, the co-digestion of sludge (L) with agri-food by-products such as wine vinasse (V) andpoultry manure (G) is analyzed. The vinasse comes from wine alcohol distillation and is generated in large quantities in our country due to the large number of existing wineries. Poultry manure, turkey droppings, present high generation rates with high total and volatile solids content and high nitrogen concentration. Both wastes require viable management alternatives since they generate serious environmental problems due to their high content of organic matter.

The C/N ratio is a very important factor in anaerobic digestion. Co-digestion of different substrates allowsnutrients to be balanced for use by microorganisms (Khalid et al. 2011). The optimal C/N rate varies according to the authors in the range between 20 and 35 (Lee et al. 2009; Khalid et al. 2011). Therefore, the appropriate mixture of sludge with vinasse and poultry manure could balance the initial characteristics of each waste and achieve a more stable mixture in terms of the most suitable carbon:nitrogen ratio, allowing to improve the performance and production results of biogas.

Temperature is a decisive operational parameter that influences the anaerobic reaction speed of the biological process as well as the composition of the biogas (due to the solubility of gases at different temperatures) and the microbial population depend on it. Thus, thermophilic anaerobic digestion at 55° C has numerous advantages as well as disadvantages (Lee et al. 2009). Among the main advantages is the greater speed of the process as well as greater efficiency in organic load removals. In addition, a greater number of pathogens can be inactivated and removal at high temperatures so the sludge generated at the end of the process can be used directly as biofertilizer or fertilizer free of toxicity (De la Rubia et al. 2006). Additionally, greater process stability, kinetics and performance are achieved in the production of methane (Riau et al. 2010),obtaining much greater quantities than in mesophilic conditions as well as a lower viscosity of the effluent (Zhouet al. 2008).

However, thermophilic temperatures mean an increase in the energy cost of operation as well as an increase in the cost of the system used and its maintenance, since the materials deteriorate more easily at hightemperatures. Another notable disadvantage is related to the stability of the process since thermophilic microorganisms are more sensitive to variations in residual concentration and fluctuations in temperature. On the other hand, high concentrations of nitrogen under thermophilic conditions can cause the appearance of ammonia which, at certain concentrations, is toxic to microorganisms, inhibiting the process when the population is damaged (Acosta et al. 2005).

Once the viability of the mixture Sewage Sludge (S): Vinasse (V): Poultry manure (P)has been verified in relation to its C/N ratio content, the main aim of this study was to analyze the removal TS efficiency (%) and theproduction of methane in anaerobic co-digestion processes at different operational temperatures. mesophilic at 35 ° and thermophilic at 55 ° C.

MATERIALS AND METHODS

Two stirred tank reactors of 3 litres total capacity were used, of which 2 litres are useful. One reactor operated at 35 ° C and the other 55 ° C. Each reactor had an independent stirred system set at 24 rpm. The biogas was collected in 3L capacity polyvinyl fluoride Tedlar bags (Zahedi et al. 2016).

The reactors were initially loaded with 1200 mL of S: V: P mixture (10g / L) and 800 mL of inoculum from a 50:50 single-stage sludge and vinasse reactor. The proportion of 40% inoculum is recommended by several authors, considering it optimal for the production of biogas and acclimatization of the substrate (Zahedi et al. 2016; Montañes et al. 2014). The established THR was 20 days, therefore the daily feeding was 100 ml of the S:V: P mixture (10g / L).

The initial characterization of each by-product and of the feed is shown in Table 1. Total solids (TS), volatilesolids (VS), total and soluble COD, VFA, C/N ratio, and pH were determined according to standard standard methods. APHA-AWWA-WPFC (Standard Methods, 2012). For the determination of pH, a HACH sensION + pH meter was used. The composition and volume of biogas was measured with a Geotech biogas 5000 methanimeter, a gas flow meter (Ritter TG1) and a gas suction pump (KNF Laboport), respectively.

RESULTS AND DISCUSSION

In the first stage, an initial characterization of each by-product was carried out separately and also of themixture of by-products used as feed, as can be seen in Table 1.

Table 1. *Initial characterization of sludge, by-products and feed (L: V: G)*

Parameters	SLUDGE (S)	VINASSE (V)	POULTRY MANURE (P)	S:V:P(10g/L)
TCOD (g/L)	46.88 ± 0.05	26.05 ± 0.01	59.38± 0.02	46.02 ± 0.00
SCOD (g/L)	25.61±0.01	26.04±0.02	51.61± 0.02	30.53± 0.001
TS (g/L)	34.05±0.04	12.90±0.07	704.78±0.03	24.98 ± 0.04
VD (g/L)	27.30± 0.03	10.73±0.02	452.80±0.01	20.78± 0.06
pH C/N	7.19 22.25	3.32 182.82	9.72 6.20	5.55 32.73
Total VFA(mg/L)	6310.11	1640.41	10583.58	6164.71

As can be seen, the C/N ratio of the mixture is within the recommended range, with a C/N ratio of 32.73 (Sreela-Or et al., 2011).

Figure 1 shows the evolution of the purifying efficiency, determined as a percentage of Total Solid removal, throughout the test for both reactors. It is observed that the TS removal values are increasing during all the studied period, following a similar evolution in both mesophilic and thermophilic conditions. The highest values were reaching in the thermophilic reactor with a maximum of 70% TS removal while the mesophilic reactor only reached 60% TS removal. It can also be observed in Figure 1 how thermophilic was most efficient than mesophilic condition, reaching levels above 50% in less time.

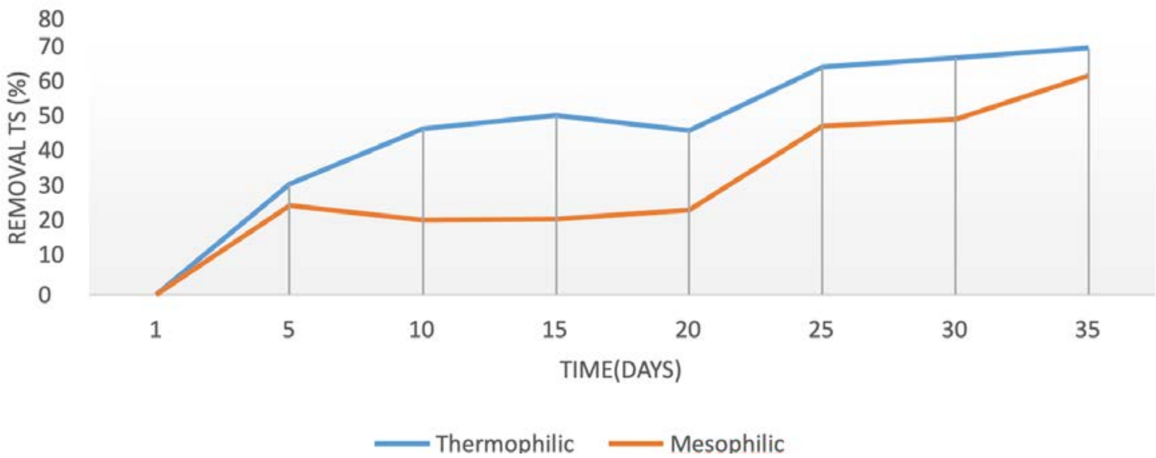

Figure 1. *Evolution of the purifying efficacy determined in percentage of reduction of TS.*

Evolution of daily methane production is shown in Figure 2. By analyzing the CH4 volume data the most striking data can be seen. Under thermophilic conditions significantly higher volume values can be obtained than at mesophilic conditions. The time required for the acclimatization of the substrate and the start of methane production were similar in both reactors as we can see in figure 2. However, after 14 days of the process, mesophilic reactor stoped producing CH4 (or very small amounts), while the thermophilic reactor registered higher methane productions, reaching more than 0.7 litres per day of volume produced. Referring to the % of composition, the % of CH_4 in thermophilic conditions reached 62.3% while, at mesophilic conditions, the maximum value reached was around 30% of CH_4 in the biogas composition.

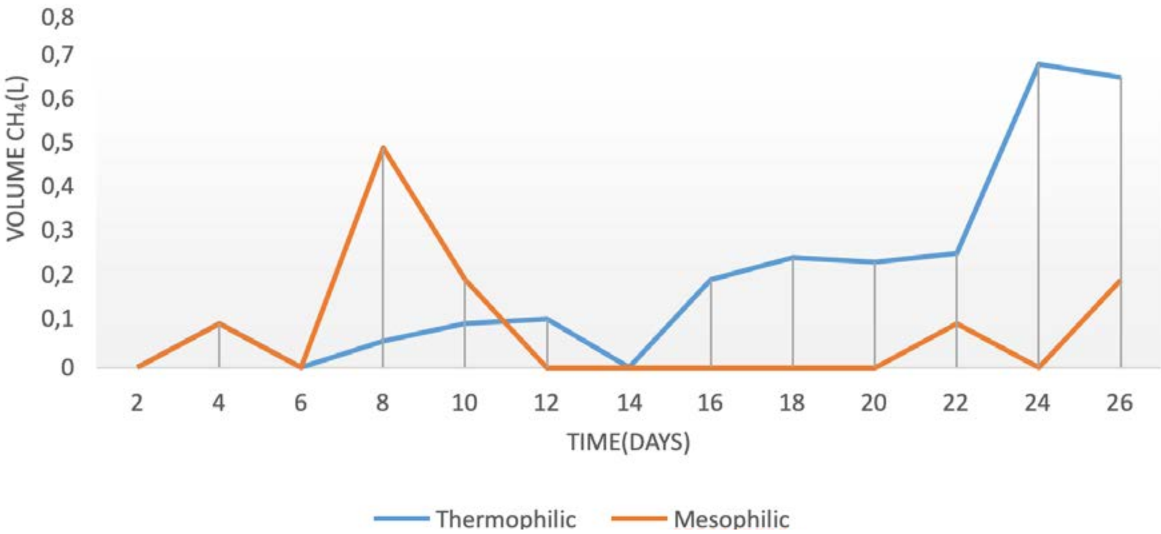

Figure 2. *Evolution of daily methane production (L).*

A notable decrease in total and soluble COD was jointly observed, in addition to volatile solids. On the other hand, there has also been a decrease in the total VFA that lowered its concentration by values close to 40%.

CONCLUSIONS

The ratio of the substrates S: V: P proposed for anaerobic co-digestion under the test conditions at THR: 20 days is adequate, allowing a stable and effective process. In relation to temperature, thermophilic operational conditions allow better removal efficiency in terms of TS (%) in less time, as well as a greater volume of methane during the process, up to 0.6 litres difference per day. Taking into account these results, the 3 by-products can be used as raw material in anaerobic digestion to obtain methane in wastewater treatment plants as a sustainable way to solve an emerging problem and, at the same time, to obtain a renewable and clean energy product.

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06

INNOVATIVE TECHNOLOGIES

LOW COST WASTEWATER TREATMENT IN ANAEROBIC PHOTOBIOREACTORS ENRICHED IN PURPLE PHOTOTROPHIC BACTERIA

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Juan Antonio Melero[2] • Daniel Puyol[2] • Frank Rogalla[1] • Victor Monsalvo[1]

Abstract

Traditional wastewater treatment plants (WWTPs) are increasingly regarded as water and resource recovery facilities (WRRFs), reflecting the value of water, nutrients, energy and other resources, besides ensuring the required effluent quality. However, effective resource recovery remains as one of the biggest challenges in wastewater management systems due to the heterogeneity of the wastewater streams and high operational costs. DEEP PURPLE project's concept relies on a disruptive and low-cost wastewater treatment based on the use of purple phototrophic bacteria in anaerobic carrousel-type photobioreactors. The largest photobioreactors built so far are located at the wastewater treatment plant Estiviel (Toledo, Spain). Nitrogen and phosphorus removal up to 60% and COD and TSS removal close to 90% have been achieved treating domestic wastewater in a single-step process. Thus dissipation of both carbon and nutrients are avoided, while obtaining an effluent compliant with discharge limits in COD, BDO5 and TSS and a nutrient-enriched biomass readily to be used as a fertilizer. Scale up of the technology from TRL 6 to TRL 7 is underway with the construction of two demonstrative photobioreactors with a maximum treatment capacity of 600 m^3/d.

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INTRODUCTION

Emerging concepts on wastewater treatment are related to circular economy in water usage, with two main objectives: (i) energy self-sufficiency or even surplus energy of existing and future wastewater treatment plants, and (ii) full recovery of the resources contained in the wastewater, mainly organics and nutrients (Lema et al. 2017). The next generation of wastewater treatment plants is targeting energy neutrality and complete recovery of nutrients, particularly N and P. However, the activated sludge-nitrogen removal process remains as the prevailing technology to treat domestic wastewater, which presents high operative costs, high C-footprint, large volume of waste disposal and nutrients dissipation through oxidative or reductive processes. The development and validation of alternative and scalable technologies are therefore necessary towards the transition from a WWTP to WWRF in the framework of circular economy.

In this sense, the core technology of DEEP PURPLE project consisting on anaerobic carrousel-type raceways enriched in PPB states as a real alternative to conventional wastewater treatment. The main features of this technology rely on the high flexibility and metabolic versatility of PPB, allowing their adaptation to a variety of environments, including the most complex waste streams. Their successful adaptation to such demanding conditions is partly the result of internal polymers accumulation which can be stored for electron/energy balance or as carbon and nutrients reserves for deprivation periods. Previous studies on wastewater have shown that PPB are able to recover: i) organics: 95% of soluble COD; ii) nitrogen: 99.6% of NH_4^+; iii) phosphorous: more that 89% of PO_4^{-3} (Hülsen et al. 2016).

In this work, we present the preliminary results of two pilot-scale photo-bioreactors located at WWTP Estiviel (Toledo) treating domestic wastewater after primary settling tanks. The aim of this study was to promote the growth and accumulation of a well-stablished PPB culture before switching to continuous operation.

METHODS

The pilot plant consisted of two lines consisting each one on one carrousel-.type raceway of 32.3 m^3 covered by infrared polyamide filters and one 2.7 m^3-decanter. Each anaerobic raceways had two submersible pumps to allow for a proper agitation and access of the infrared light to the PPB in the whole water column.

The pilot plant has been operated in batch mode for 6 months treating domestic wastewater after primary settling tanks. The average composition of the domestic wastewater after primary settling tanks is shown in Table 1. End criteria of each batch was set at 50% soluble COD removal to promote exponential growth of PPB culture in the photobioreactors. Twenty four batches were performed within the period of study (6 months) with the aim of stablishing a PPB-enriched biomass within the anaerobic photobioreactors prior to continuous operation. The duration of each batch was dependant on the removal of soluble COD. Once reached at least 50%

removal, the photobioreactors were emptied while keeping the PPB biomass (approximately 10% of the photobioreactor volume). Each batch lasted from 4 to 8 days. The performance of the pilot plant was assessed in terms of nutrients and soluble COD removal and increase of the ratio of PPB biomass to total biomass. Total and volatile suspended solids (TSS, VSS) were also monitored at the beginning and end of each batch. Ammonium, phosphate, total and soluble COD, total nitrogen and total phosphorus concentrations were determined by commercial colorimetric kits (Hach Langue, Spain). TSS and VSS were measured according to standard methods (Standard Methods, 2005). The ratio PPB biomass to total biomass was measured as the ratio of absorbance at 850 and 660 nm. Total COD, TSS, total nitrogen (TN) and total phosphorus (TP) and BDO5 were measured in the effluent at the end of each batch.

Table 1. *Average composition of domestic wastewater after primary settling tanks at WWTP Estiviel (Toledo).*

Parameter	Units	Value
BDO_5	mg/L	249±56
Total COD	mg/L	452±77
Soluble COD	mg/L	220±80
TSS	mg/L	102±17
TN	mg/L	57±13
TP	mg/L	7.1±1.3
NH_4^+	mg/L	37±7
PO_3^{4-}	mg/L	9.1±1.3
pH		7.1±0.3

RESULTS AND DISCUSSION

Figure 1 shows the main performance indicators of the anaerobic photobioreactors operation during each batch. In general, COD removal efficiencies were promising, with values averaging 50-60%. Ammonium removal reached 40%, although the average removal is only 14%. This is due to the imbalance of the ratio COD/N/P ratio in wastewater (100/12.6/1.6), which may hamper complete removal of nitrogen by assimilative uptake by PPB. Conversely, removal efficiency for orthophosphate was greater than that of the ammonium with values close to 100% in the longest batches. This is likely due to the accumulation of phosphorus as poly-phosphate during sun hours, which is a characteristic mechanism of PPB to save energy for operation in dark periods (Fradinho et al. 2021). The development of the phototrophic bacteria was confirmed by analysis of the ratio of Bacteriochlorophyll *a* absorbance (at 805 or 865 nm) to particulates absorbance (generally given at 660 nm). The highest the ratio is, the best phototrophic enrichment (de Las Heras et al. 2020). This ratio increased from 0.69 to 0.95 at the end of the operation in batch mode, which indicates the proliferation of PPB into the biomass.

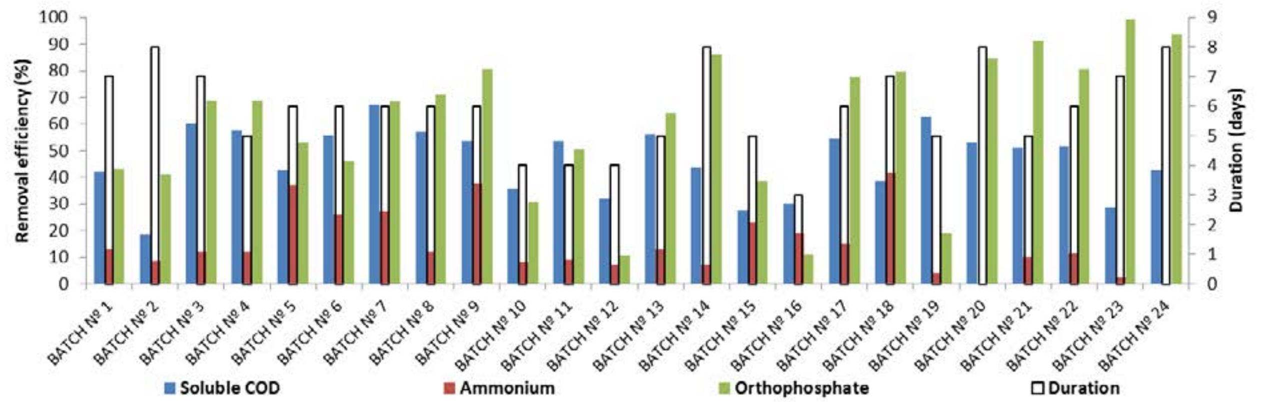

Figure 1. *Removal efficiency of soluble COD and nutrients of pilot anaerobic photobioreactors at WWTP Estiviel (Toledo).*

The quality of the effluent of the photobioreactors complied with discharge limits for total COD, TSS and DBO5, as depicted in Figure 2. Low N removal due to COD/N imbalance caused the N values to be above discharge limits. It is necessary to provide an additional source of COD to the domestic wastewater to attain N discharge limits. This is being tackling in the DEEP PURPLE project by using an organic source coming from the hydrolysis of the organic fraction of municipal solid waste, having very high COD/N ratios. This stream has been proved biodegradable by a PPB mixed culture (Allegue et al. 2020). Thus, next step will be to check the feasibility of the co-treatment of both residual flows in a single line for achieving full discharge limits.

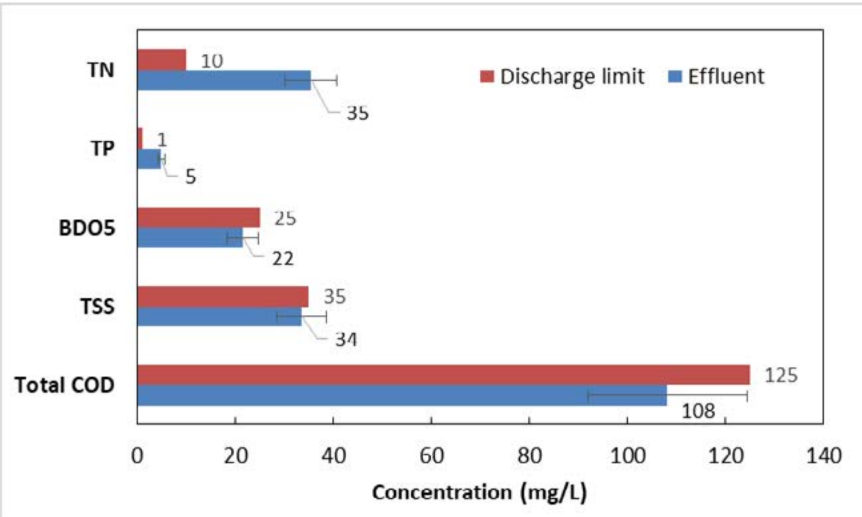

Figure 2. *Discharge parameters for the effluent of the pilot photobioreactors vs discharge limits.*

Following these promising results, within DEEP PURPLE project the PPB-based anaerobic pilot photo-bioreactors will be scaled up to a capacity of 600 m^3/d, which will be the biggest PPB photobioreactors ever built with a biomass production capacity of almost 20 ton/year (TRL 7).

CONCLUSIONS

PPB-based anaerobic photobioreactors state as an emerging technology that enables the recovery of organics, nitrogen and phosphorus from wastewater streams in a single treatment step by biomass concentration. This work shows the feasibility of this technology to treat domestic wastewater while achieving discharge limits for total COD, TSS and BDO5. Optimization of the operational strategy in terms of organic loading rate and optimal COD/N/P ratio is key to increase nutrients removal towards complete compliance with discharge limits. Feeding during light/dark periods and changing infrared irradiance light are parameters to be furtherly assessed.

ACKNOWLEDGEMENTS

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EVALUATION OF SMALL DOSES OF H2O2FOR SOLAR WATER DISINFECTION ENHANCEMENT

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Abstract

Water is essential to human beings yet more than 2 billion people lack access to safely managed water sources. Since ancient times, harvesting rainwater has been one of the main solutions to this problem at isolated areas even if nowadays it is mainly only used at low-income countries. To treat this stored water making it potable,solar disinfection (SODIS) with solar photo-reactors is an efficient and widely-proven water disinfection procedure, which enhancement by adding very low H2O2 concentrations (< 10 mg/L) is the main focus of this work. The effect of this oxidant is tested against several pathogens commonly found in harvested rainwater (E. coli, E. faecalis, S. enteritidis, P. aeruginosa and MS2 bacteriophage). Results have shown that H2O2 addition enhanced the inactivation of all pathogens being the most noteworthy MS2 inactivation. For this pathogen, at the highest H2O2dose tested, a reduction higher than 4 log wasobtained after 1 min of treatment time (1.70 kJm-2) increasing its inactivation kinetic more than 400 times. The four bacteria inactivation kinetics were also enhanced with the combination H2O2/SODIS being inactivated (> 5 log reduction) with 170 kJm-2 (105 min of solar exposure) and so showing an enhancement of 1.5 times.

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INTRODUCTION

Rainwater collection is common practice at isolated areas in low income countries to mitigate water scarcity during dry season. Nevertheless, chemical and microbiological quality of harvested rainwater (HRW) does not fit drinking water standards, requiring a disinfection process prior to its consumption. Solar water disinfection (SODIS) is a very low-cost recognized household water technique for the disinfection of drinking water which is grounded on the lethal effect of solar UV radiation and temperature to inactivate faecal bacteria and other microorganisms in water. Known since the seventies, its use has been increasing worldwide since then, mainly due to its low cost, easy procedure and effectiveness(McGuigan *et al*, 2012).Traditionally, its application involved the exposure of 1.5-2 L polyethylene terephthalate (PET) bottles to direct sunlight for around 6 hours or during 48 hours under cloudy conditions. However, the main drawbacks of this technique are the limited amount of water that can be treated per day and the relatively high time required for reaching safe pathogens' inactivation. In this line, many researchers have focused their efforts in overpassing these mentioned limitations by the use of solar reflectors, transparent containers with higher volume, alternative UV-transmitting materials, the use of chemical oxidants or reduction of water turbidity, among others.

Commonly, the reactors used for SODIS are based in Compound Parabolic Collectors (CPCs) which have been widely investigated for the last two decades, and whose disinfection performance has beengreatly demonstrated. However, a new, low-cost solar reactor design grounded in the use of V-trough mirrors has been developed. The suitability of this reactor for SODIS application has been tested and compared to the CPC photoreactor finding no important differences between the performance of both designs (Martínez-García *et al.* 2020). On the other hand, the enhancement of SODIS with chemical additives such as H_2O_2 at concentrations ranged between 10-50 mgL^{-1} has been also investigated showing very promising results for the inactivation of several microorganisms in different water matrices in CPC photo-reactors (Polo-López et al., 2011; García *et al.* 2012).

In view of such previous investigations, the present work sets its main goal in the evaluation of the influence of adding very low concentrations of H_2O_2 (<10 mgL^{-1}) on simulated harvested rainwater solar disinfection process. Experiments have been carried out in a V-trough photo-reactor of 25 L under controlled conditions by using synthetic rainwater (SRW) as water matrix. The inactivation rate of a consortium of pathogens commonly found in HRW has been assessed under natural sunlight.

METHODS

Experiments have been performed at Plataforma Solar de Almeria (Almeria, Spain) in completely sunny days between July-September 2020 with up to 5 hours of treatment time (11:00 to 16:00 local time). Batches of SRW spiked with five pathogens were tested in a 25 L V-trough photo-reactors (see Fig 1) in static mode (no water recirculation). This reactor is made of a tube of modified polymethyl methacrylate (PMMA) which allows the transmission of solar UV radiation: 40-85 % of UVB and 85-95% of UVA. The tube has a diameter of 200 mm and is placed in the lineal focus of anodized aluminium flat V-shaped reflector built on a 37° tilted aluminium structure which resist outdoors conditions.

Figure 1. *25 L V-trough solar photo-reactor at PSA facilities*

Four culture-type bacterial strains: *E. coli* K-12 (CECT 4624), *Enterococcus faecalis* (CECT 5143), *Salmonella* sub enteritidis (CECT 4155) and *Pseudomonas aeruginosa* (CECT 110), and MS2 bacteriophage (ATCC 15597-B1) and its host *E. coli* C300 (ATCC 15597) were used. Suspensions containing each bacterium and the virus were simultaneously diluted directly into the SRW to an initial concentration of 10^6 $CFUmL^{-1}$. Bacteria samples were enumerated by using plate counting method in different selective media. Adouble-layer agar method with Tryptone Glucose Yeast extract was used for MS2 enumeration. Pathogens' quantification was done by diluting samples 10-fold. When lower concentrations were expected, 500 µL were used to reduce the detection limit (DL) till 2 $CFUmL^{-1}$. Colonies were counted after incubation of 24-48h at 37±2 °C. Reagents and concentrations used for SRW weredescribed in detail elsewhere (Martínez-García *et al.* 2020). In order to compare pathogens inactivation, results were adjusted with some variations of Chick-Watson's Law (first order adjustment) explained elsewhere (Marugán *et al.* 2008). With these fitting, kinetic inactivation constants for each pathogen were obtained.

UV radiation was measured with a pyranometer (Kipp & Zonen CUV-5 (280-400 nm)) tilted 37°. Pyranometer provides data in terms of incident solar radiant energy rate on a surface per unit of area (Wm^{-2}). Microbial concentration reductions (logarithmic reduction value (LRV)) are fitted as function of cumulated solar UV dose (kJm^{-2}). Water temperature (°C) was measured by using a portable digital thermometer Checktemp 1 (Hanna Instruments). H_2O_2 (Sigma-Aldrich®, Germany) 34.5-36.5% (W/v) was employed as received and added directly to the reactor. Its concentration was measured with a spectrophotometer EvolutionTM 220 (Thermo Scientific, Spain) by using a colorimetric method following the standard procedureDIN 38 402 H15 explained elsewhere (García *et al.* 2012). Catalase (Sigma-Aldrich®, Germany) was added to all the samples previously to microorganisms' analysis, to eliminate remaining H_2O_2 and so preventing any further effect over the pathogens.

During the experimental set-up, every condition was tested at least three times. The ANOVA analysis of the results permitted to discard any data with a confidence level below 95%. The error bar was calculated as thestandard deviation of all confident results.

RESULTS AND DISCUSSION

To study the enhancement effect of solar radiation combined with H_2O_2on pathogens inactivation, the mere effect of this oxidant at very low concentrations over all the microbial pathogens was firstly tested in the dark. Results demonstrated that H_2O_2had no inactivating effect over the four different bacteria as the initial concentration remained constant during 5 hours (data not shown). However, in the case of the MS2 bacteriophage,at the lowest concentration tested (1 mgL^{-1}), the initial concentration was reduced more than 3 LRV after 60 minutes, and with the highest dose tested (10 mgL^{-1}) it reached DL after only 20 minutes (Fig. 2. a)).

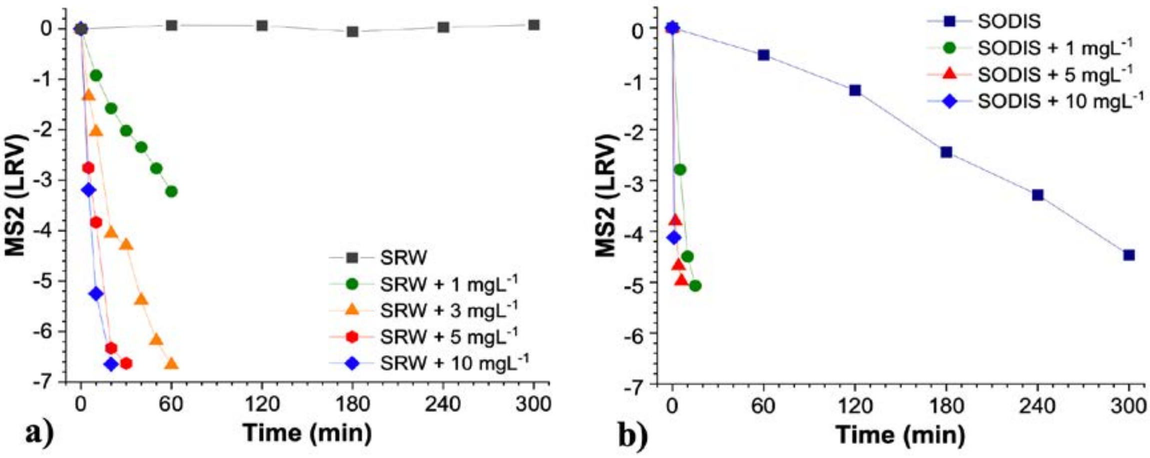

Figure 2. *Effect of H2O2 addition over the inactivation kinetics of MS2 bacteriophage in a) dark, and b) with solar light.*

Afterwards, the combination of three low concentrations (1, 5, and 10 mgL^{-1}) of $H_2O_2$and SODIS were tested and compared with only SODIS.In all cases, H_2O_2 concentration remained almost constant during the experimental time (3 hours). As water matrix did not contain any organic matter which could react with it, the slight differencesof concentrations were considered to be due to normal autodecomposition.

Results highlighted that the combination of SODIS/H_2O_2has a great effect on pathogens' inactivation performance even if not all of them are affected in the same way. Table 1 shows inactivation kinetic constants calculated for all the pathogens in all the conditions tested. As expected the addition of this oxidant proved to be very effective in the inactivation of MS2 and its effect was potentially improved with sunlight (Fig. 2b). The inactivation kinetic of MS2 increased from -0.006±0.001 m^2kJ^{-1}at SODIS (784 kJm^{-2} and 5 hours of solar exposure) to -2.41±0.01 m^2kJ^{-1} when 10 $mgL^{-1}$of H_2O_2 was added (1.70 kJm^{-2} and 1 minute), what means an increase higher than 400 times. In the case of the bacteria, the most affected by the addition of $H_2O_2$are *E. coli* and *P. aeruginosa*. With the highest dose tested these bacteria inactivation kinetic constants increased from-0.03±0.01 and -0.04±0.01 m^2kJ^{-1} to -0.10±0.01 m^2kJ^{-1},which represent anupgrading of 3.4 and 2.4 times for *E. coli* and *P. aeruginosa*,respectively. Contrarily, the less affected by $H_2O_2$is *E. faecalis* for which at the maximum dose of $H_2O_2$the cumulated dose needed to inactivate this bacterium was 170 kJm^{-2} instead of 260 kJm^{-2} needed when only SODIS is applied.This lower effect may be probably due to the fact that this is a Gram-positive bacterium whose cell wall is thicker and prevent damaging oxidative effect of both solar UV radiation and H_2O_2. The mechanisms of inactivation by solar/H_2O_2 is widely described in literature and is grounded in the fact that both, solar UV radiation and H_2O_2generate different reactive oxygen species (ROS) within the cell. When H_2O_2permeates through cell wall it reacts with inner iron-containing complexes which damagethe internal cell structures up to a point where it natural ability of recovery from oxidative damage is overcome leading to their final inactivation(Giannakis *et al*. 2016). In all the experiments, for each pathogen regrowth after 24 and 48 of solar exposure was tested for the samples where DL was achieved and the following one. *P. aeruginosa*was the only pathogen found able to repair the oxidative damage received in all the cases.

Table 1. *Inactivation kinetic constants (m2kJ-1) of all five pathogens for each operational condition tested.*

	E. coli	*E. faecalis*	*S. enteritidis*	*P. aeruginosa*	MS2
SODIS	-0.03±0.01 (R^2 0.98)	-0.03±0.01 (R^2 0.98)	-0.04±0.01 (R^2 0.97)	-0.04±0.01 (R^2 0.96)	-0.006±0.001 (R^2 0.99)
1 mgL^{-1}	-0.03±0.01 (R^2 0.98)	-0.03±0.01 (R^2 0.97)	-0.03±0.01 (R^2 0.96)	-0.06±0.01 (R^2 0.96)	-0.20±0.02(R^2 0.95)
5 mgL^{-1}	-0.04±0.01 (R^2 0.96)	-0.05±0.01 (R^2 0.97)	-0.05±0.01 (R^2 0.97)	-0.07±0.01 (R^2 0.97)	-0.99±0.27 (R^2 0.81)
10 mgL^{-1}	-0.10±0.01 (R^2 0.94)	-0.06±0.01 (R^2 0.88)	-0.07±0.01 (R^2 0.96)	-0.10±0.01 (R^2 0.96)	-2.41±0.01 (R^2 1.00)

CONCLUSIONS

The comination of very low doses of $H_2O_2$with SODIS has been proven to enhance the inactivation of waterborne pathogens even if not all pathogen are equally strongly affected. The three dose tested (1, 5, and 10 mgL^{-1})reduced both the exposure time and the needed cumulated solar UV radiation to inactivate all five pathogens (> 5 LRV) which resulted in an increment of their inactivation kinetics. The order of inactivation found for all the tested H_2O_2 concentrationswas: MS2 >*E. coli*>*P. aeruginosa* >*S. enteritidis*>*E. faecalis*. At the best concentration of H_2O_2 used in this study, 10 mgL^{-1}, DL of *E. faecalis* (the higher resistant bacteria) was attained after a dose of 170kJm^{-2}(105 min) of solar exposure. If an average of 8 hours of solar light per day is estimated, these data prove that between 4 and 5 batches of water could be safely treated-per day.Finally, taking into account the remaining dose of H_2O_2 and that the end use of the treated water is its final human consumption, additional toxicity tests must be carried out in order to ensure its safety.

ACKNOWLEDGMENTS

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NOVEL DIRECT ULTRAFILTRATION SYSTEM ASSISTED BY COAGULATION-FLOCCULATION FOR SEWAGE TREATMENT

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Abstract
Direct ultrafiltration membrane (DUF) is a promising technology for municipal wastewater treatment in locations with space limitations. An innovative rotating hollow fibre module (RHFM) for DUF was used for the rapid treatment of domestic sewage. Coagulation-flocculation was applied in combination with DUF as possible way for mitigate membrane fouling. This process combination can significantly reduce the overall fouling reported and concentrate organics and nutrients into sludge to facilitate resource recovery. Using a commercial coagulant based on Poly-Aluminium Chloride Hydroxide Sulphate, approximately 95.0% of total suspended solids (TSS) and 80% of the chemical oxygen demand (COD) were removed from the sewage influent and retained in the concentrated sludge. The novel membrane module showed its successful removing and dispersing the previous mass attached on the membrane surface, but *in-situ* coagulation resulted more interesting for long-term operation since floccules breakage can contribute to worsen the membrane performance and final effluent quality. DUF with RHFM can be readily modularized and installed as units to upgrade the current scenario of marine outfalls and wastewater treatment plants spillways.

INTRODUCTION

Domestic wastewaters without adequate treatment are discharged to seaside leading to a common problem supported by space availability, financial and management deficiencies. In many cases, these events require alternative solutions to conventional ones.The Canary Islands is the Spanish autonomous region with the longest coastline, and reported394discharging points in 2017, of which an important percentage are marine outfalls for wastewater treatment plants spillways.The current situation should be enhanced by compact technologies capable of decreasing the organic load, as well as removing turbidity and pathogens. In this sense, membrane direct filtration (MDF) without biological reactors and high space and energetic requirements, could be an interesting option. This technology is usually located at the end of conventional wastewater treatment plants by activated sludge, as advanced treatment. Nevertheless, some previous studies have proved its possible and optimistic application to concentrate organic matter from wastewater, without biological degradation, and to produce a filtrate less environmentally harmful. The only handicap of MDF is the membrane fouling which causes permeability loss and productivity limitations. The current work shows a preliminary study of direct ultrafiltration carried out witha novel rotating hollow fibre module designed by ULL research group. Previous works have showed its successful application on anaerobic membrane bioreactors and on direct ultrafiltration (DUF) of raw wastewater, but the present study focuses on a combination of DUF with coagulation-flocculation for preventing to mitigate membrane fouling.

METHODS

Domestic wastewater collected from the municipal wastewater treatment plant of Noreste (Tenerife- Canary Islands, Spain)was pre-treated applying the following protocol: sedimentation of large particles (2 hours) and subsequent coagulation-flocculation-sedimentation process. The pre-treatment was carried out at ambient temperature with DEFLOCAR (Proquimia SA, Barcelona, Spain) using a dose of 0.4 mM Al^{3+}. The selection of the commercial flocculant-coagulant, whose active principle is Poly-Aluminium Chloride Hydroxide Sulphate, and its concentration is based on previous studies (Ruigómez et al, 2019). pH of DWW was not adjusted, remaining between 7–8. In addition, two different methods of pre-coagulation were assayed: *ex-situ* and *in-situ*. In *ex-situ* pre-coagulation trials, the coagulant was added to the wastewater supernatant of the first sedimentation stage using an external mixing chamber. The effluent was then fed to the experimental unit. On the other hand, *in-situ* pre-coagulation involved direct coagulant addition into the filtration tank, keeping the flocs in the feed suspension. Bench filtration trials were performed with a ZeeWeed ZW-1® hollow-fiber ultrafiltration module (SUEZ Water Technologies and Solutions, Ontario, ON, Canada). The bundle consisted in 97 fibreswith an average pore size of 0.04 μm and a nominal surface area of 0.047 m^2. Filtration was carried out at constant flux from outside to inside of the fibres. Permeate was extracted from the upper header of the module through the vacuum generated by a magnetic drive gear pump (Micropump-GA Series, AxFlow, Stockholm, Sweden) and membrane rotation was applied during the physical cleanings (i.e. backwashing or relaxation stages) in order to improve foulants back-transport effectiveness.Membrane fouling was assessed using a pressure sensor (Sensotech, Barcelona, Spain) that measure and register the transmembrane pressure (*TMP*) of the unit (Fig. 1).

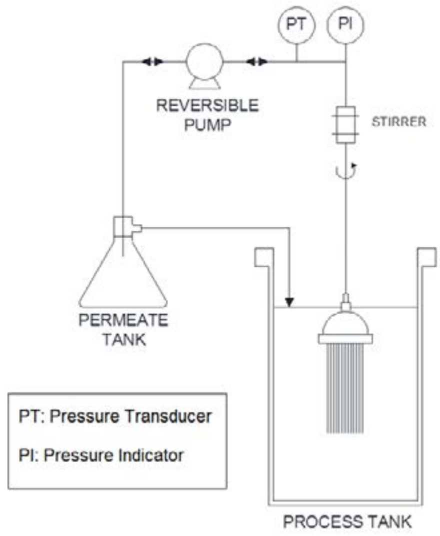

Figure 1. *Experimental set-up*

RESULTS AND DISCUSSION

Wastewater DUF performance was evaluated in terms of the hydraulic resistances (*R*) to mass transfer through the membrane. Preliminary trials with and without pre-treatment (sedimentation of large particles and subsequent pre-coagulation, *ex-situ* and *in-situ*) were conducted in order to determine the best operational mode. All trials were conducted at J of 24 L/h·m^2 and periodical backwashing was applied each 450 s during 30 s at 60 L/h·m^2 accompanied of rotation module at 260 rpm.

As in previous studies, membrane fouling was quantified using the membrane resistance-in-series model (Vera et al, 2015; Ruigómez et al, 2016). According to this approach, the overall fouling resistance can be determined as the sum of different contributions due to specific fouling mechanisms. It is assumed that each resistance is additive and independent of each other. Figure 2a shows the internal residual fouling (R_{if}) evolution throughout wastewater *DUF* with and without pre-coagulation. Results showed an abrupt rise of global resistance after each backwashing period (R_m+R_{if}) at the beginning of the trials. This behaviour could be due to a fast mass consolidation that cannot be detached by physical cleanings. This phenomenon has been related to the large content of colloidal particles in the *DWW* which may cause a rapid blocking/narrowing of the membrane pores (Zhao et al., 2019;Lin et al, 2014). Despite the fact that all trials exhibit the same trend, with an initial phase where the system reached a pseudo-steady state and R_{if} increased slowly and gradually, it also seems evident that wastewater pre-treatment plays a significant role on membrane fouling development. Without chemical coagulation, the large content of particulate matter (see Table 1) caused a marked increase of the transmembrane pressure during the first hours, up to approximately 40 kPa.By contrast, membrane fouling was effectively mitigated in the trials performed with pre-coagulation. In agreement with prior works, this process is related to a sweep mechanism where particles and colloids are entrapped within a mesh of aluminium hydroxide forming a precipitate. Therefore, *SS* and *COD* removals close to 95% and 80% were achieved in the supernatant of *ex-situ* and *in-situ* pre-coagulation trials.

Table 1. *Resistance evolution with the elapsed time*

Parameters	*Units*	*Raw wastewater*	*Settled wastewater*	*Supernatant from Coagulation-flocculation*
		Average value ± standard deviation		
TSS	*mg/L*	*580.7±204.9*	*175.5±54.3*	*33.0±19.3*
VSS	*mg/L*	*456.9±147.3*	*157.8±47.8*	*27.1±19.1*
COD	*mg/L*	*994.9±156.9*	*434.7±81.2*	*211.0±99.2*
COD_s	*mg/L*	*242.6±113.3*	*197.9±54.0*	*175.8±127.5*
DOC	*mg/L*	*56.8±26.5*	*54.2±6.0*	*41.8±28.5*
Turbidity	*NTU*	*495.8±165.7*	*159.8±51.0*	*20.3±15.9*
pH	*-*	*7.5*	*7.9±0.1*	*7.6±0.3*
Conductivity	*mS/cm*	*1,978.2*	*1,890.4±88.4*	*1,968.8±201.9*

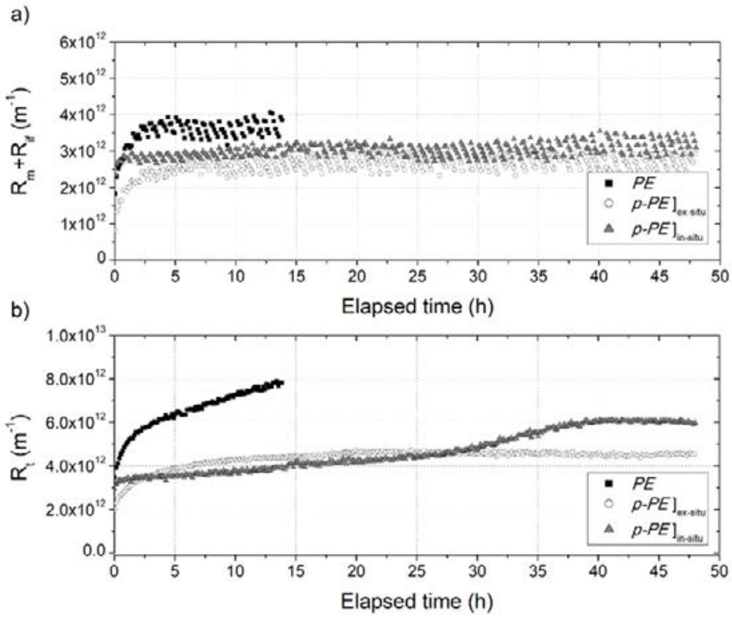

Figure 2. ***Resistance evolution with the elapsed time***

As expected, this reduction of foulants decreased the external fouling propensity of the *DWW*, but two different behaviours of total or also called, final fouling as R_t were observed depending on the dosage mode at long-term. In *ex-situ* pre-coagulation, after the initial phase, R_t remained stable at values close to $4.5{\cdot}10^{12}$ m^{-1}. In addition, the resistances were a 38-44% lower than those obtained without chemical pre-treatment. On the other hand, *in-situ* dosage led to pseudo-steady state followed by a mark increase of R_t. Therefore, although membrane rotation re-suspended the settled aggregates during the physical cleanings, the presence of foulants changes the fouling tendency after the first 25-30 hours. This behaviour may be associated with the high shear rates and the strike of the flocs with the fibres that can result in their breakage throughout *in-situ* pre-coagulation. In fact, it is consistent with the variation of the turbidity observed in the feed suspension during the first 24 hours, which increased from 39.2 to 130 NTU. By contrast, the turbidity of the *ex-situ* pre-coagulation trial remained constant at 39.1+1.5 NTU during the same experimental period.

CONCLUSIONS

Pre-treatment of sewage by *ex-situ* coagulation-flocculation has shown to be a promising way to enhance direct ultrafiltration membrane performance in long term. In addition, the combination of both technologies succeeded in increasing the effluent quality, although it did not reach the requirements of the Directive in relation to discharges from urban wastewater treatment plants. Anyway, these preliminary results from lab-scale units must be validated at pilot scale fed with real and continuous sewage, since hydraulic conditions and the effect of rotation movement and clarification can completely differ. Nevertheless, it may be necessary to incorporate a complementary treatment to adjust the final effluent quality to legal requirements.

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INTEXT PROJECT: HYBRID INTENSIVE-EXTENSIVE RESOURCE RECOVERY FROM WASTEWATER IN SMALL COMMUNITIES

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Abstract

Aqualia leads LIFE INTEXT project: **INT**ensive-**EXT**ensive resource recovery from wastewater in small communities. INTEXT consortium is creating a technological platform located in Talavera de la Reina wastewater treatment plant (WWTP), where innovative hybrid technologies for wastewater treatment and resource recovery in small communities are developed, with the next objectives: (1) Wastewater treatment system robustness against environmental (winter-summer) and pollutants/industrial loads variations, (2) Reduction of investment and maintenance Costs -Reduction of the required area < 1 m^2/PE, (3) Quantification and assessment of greenhouse gases emissions reduction, (4) Assessment of emergent pollutants removal, (4) Disinfection and water reuse, (5) Decision Support System (DSS) based on Life Cycle Analysis and finally validation of technologies broadly used in the north and centre of Europe.

Keywords: Small communities, hybrid, resource recovery, wastewater

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INTRODUCTION

Waste Water Treatment (WWT) is fundamental to ensure public health and environmental protection and has improved in all parts of Europe over recent decades, with a growing population connected to treatment facilities to meet more demanding standards. Avoiding water pollution reduces major global risk factors for illness, diseases and mortality, and contributes to preserving available drinkable water worldwide.

The provision of economically and environmentally sustainable WWT for Small Communities (SC) remains a great challenge all over Europe, especially in countries with large numbers of small and scattered settlings. Thus, water demand and WWT must be tackled efficiently to avoid water scarcity (WS) and high operation and maintenance (O&M) costs.

In Spain, the sanitation and treatment coverage in populations with less than 2,000 population equivalent (PE) is, in general low. In 2010, only around 40–50% of the population that lived in SC had adequate sanitation coverage. It was estimated that the pollution load in that range of population that still did not have adequate treatment was between 3 and 4 M PE, spread over more than 6,000 agglomerations, many of them smaller than 500 PE.

Experience has shown that, when treating wastewater (WW) in SC, the solutions used in larger cities, INTensive technologies (such as conventional activated sludge system) are not sustainable and affordable. In this context, simple nature-based technologies with affordable O&M costs, that is, EXTensive systems based on solar energy and photosynthesis both by plants and more recently microalgae constitute sustainable alternatives to allow the transition towards this new model. However, low flexibility and adaptability to changing conditions, and large footprint are the main challenges faced by EXTensive technologies. Moreover, the removal of Emerging Pollutants (EP), the reduction of greenhouse gases (GHG) and the recovery of nutrients, are major challenges key to consider in the circular economy approach to enable safe WR, avoid waste of energy and scarce elements such as phosphorus (P), and prevent climate change.

In this context, **LIFE INTEXT aims to develop innovative solutions based on a combination of INTensive and EXTensive technologies** to deal with WWT and reuse to tackle water scarcity in SC, benefiting from the advantages that both types of technology provide. It will also provide a smart monitoring and evaluation system for consistent operation of the technologies and criteria for supporting the decision-making process of operators real-time.

PROJECT OBJECTIVES

The project aims to demonstrate innovative resource recovery technologies from WW of SC in Mediterranean and continental climate at full-scale (below 2,000 PE) and environmental conditions (TRL 7-8) with the following main objectives:

1. **To validate improved EXTensive technologies, two innovation spaces will be prepared under 2 different climate conditions (Seville and Toledo, Southern and Central Spain) to host different INTEXT technologies**:
   a) Low Carbon WWT: solar-based EXTensive systems with an improved and reliable treatment performance and footprint below 1 m²/PE.
   b) Produce valuable products in a resource-oriented circular economy
   c) Removal of EP through aerated wetlands, High Rate Algae Ponds (HRAP) and solar driven chemical free anodic oxidation based on inline electrolytic and UV radiation
2. **To provide criteria to support the decision-making process for the identification of the most appropriate INTEXT technologies** based on life cycle analysis (LCA) and socio-economic and technical factors, minimizing emissions and maximizing recovery of resources (GHG emissions reduction of 80- 90 %).

INTEXT TECHNOLOGIES & EXPECTED IMPACTS

INTEXT consortium is creating a technological platform located in Talavera de la Reina WWTP, where innovative hybrid technologies for WWT and resource recovery in SC are developed. Figure 1 shows a detail diagram of all selected technologies to be implemented, More than 16 different technologies will be implemented. Each technology will be sized to treat 125 PE and could be classified in the next different groups according to their technology and companies involved:

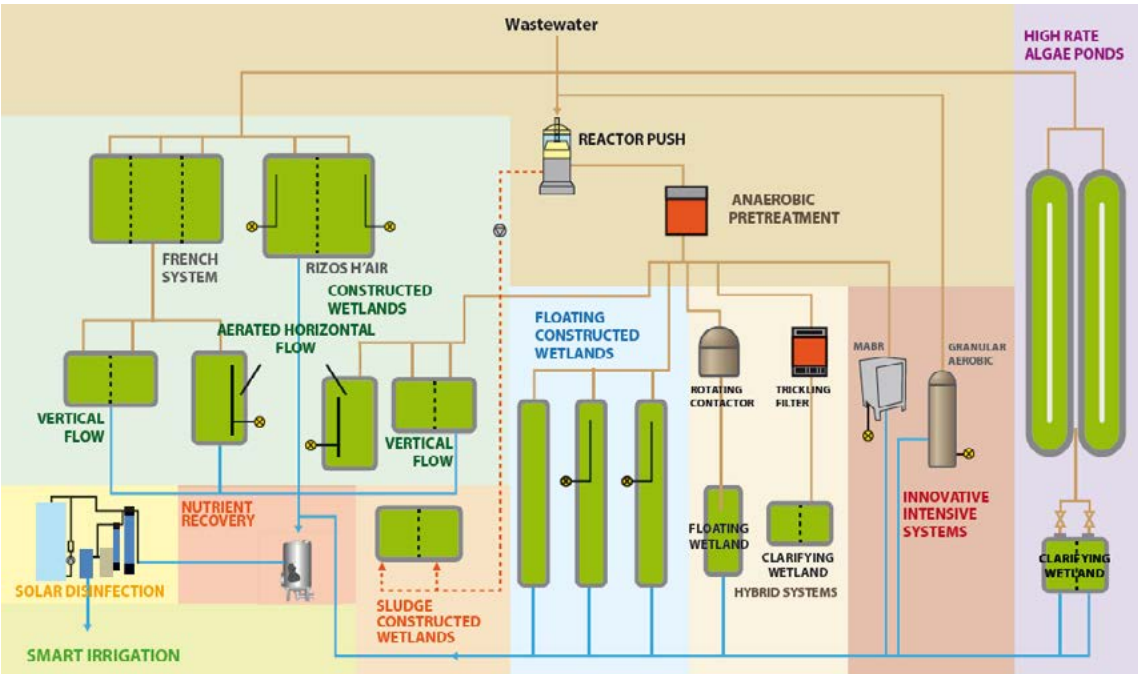

Figure 1. *Flow diagram of INTEXT Plant.*

· ANAEROBIC SYSTEMS FOR WW PRE-TREATMENT

Upflow anaerobic sludge blanket (UASB) reactor for reducing the load of solids (TSS) feeding constructed wetlands (CW) by 30–50% vs. other traditional pretreatment technologies (settler, Imhoff tank), thus avoiding media clogging in CW. LIFE INTEXT will use the PUSH® reactor (EP 3 009 408 B1) to apply the technology at demo scale and sub-mesophilic T (<20°C).

Expected impacts:

· Suspended solids removal >80% or TSS <75 mg/L.
· Chemical Oxygen Demand removal >75%.
· E recovery from biogas with a biogas yield of 0.5 m^3 CH_4/kg VS removed.
· Adapt the operation mode to load variation.
· Dissolved methane and sulfide recovery by stripping tank from effluent >70% and >95%, respectively.

· INNOVATIVE HRAP

Microalgae bioreactor with raceway configuration that supply free O2 by microalgae and new low E mixing system (0.08 kWh/m3), will include air injection automatically during winter conditions.

Expected impacts:

· Reduce E consumption, to a max. of 0.08 kWh/m^3.
· Low cost harvesting by settlement with TSS <35 mg/L.
· P removal of 80% or P <1 mg/L & N removal of 70-80% or N <10 mg/L.
· Reduce footprint to less than 1 m^2/PE
· Adapt operation mode to load variation.

· AERATED CW (CONSTRUCTED WETLAND)

Mixing of aerated wetlands (AW) and improved Rhizosph'air® system to obtain reusable treated WW, advanced denitrification (<5 mg N/L) and high pathogen removal (E. coli <1,000 MPN/100mL).

Expected impacts:

· E consumption <0.2 kWh/m^3.
· High pathogen removal: E. coli <1,000 MPN/100mL.
· Optimized denitrification (<5 mg N/L)
· Implement a compact CW tech. <1 m^2/PE.
· Adapt operation mode to load variation.
· Produce a valuable treated WW for reclamation with an integrated driven disinfection.

· SOLAR-DRIVEN AERATED HELOPHYTES IN FLOTATION SYSTEMS

Equipped with photovoltaic power-based aeration system to improve performance (E <0.2 kWh/m^3, P and N removal 70-85%).

Expected impacts:

· PV power-based aeration, Lower E consumption than 0.2 kWh/m^3.

· Removal of P to 85% & N to 80%.

· P AND N RECOVERY BY SUSTAINABLE ADSORPTION TECHNOLOGIES

Using innovative adsorbent materials, such as sol-gel coatings, nano tech engineered fibers and natural materials as apatite or opoka. It is expected a P and N recovery of 70-80 % from WW for agricultural use.

· SLUDGE TREATMENT WETLANDS

Excess sludge mineralization and dewatering from 1% dry solids content to 30%, thus reducing 30 times transportation needs.

· SOLAR ANODIC OXIDATION FOR WATER DISINFECTION

Treated water after clarification will be disinfected with an electrochemical chlorination system in order to produce high quality water for irrigation.

· SMART IRRIGATION SYSTEM in order to save water for irrigation

PROJECT IMPLEMENTATION AND VALIDATION

LIFE INTEXT works in Talavera de la Reina started on June 2020, with no delay despite the pandemic situation. Civil works have suffered certain delay during the last months of 2020 due to the heavy rainfalls suffered. However, works are progressing as scheduled and start-up is expected to take place at mid-2021. The experimental plant will be then in operation during at least two years, in order to test and validate all the presented technologies and combine them in order to optimize each configuration to reach the optimal WW quality and at the same time minimizing OPEX and footprint.

In parallel, the CENTA facilities in Carrión de los Céspedes (Sevilla) will be used for the implementation of INTEXT technologies and the evaluation of their potential to improve the existing treatment plants. Two constructed wetlands will be revamped and optimized to test INTEXT technologies in a Mediterranean climate.

Next, INTEXT experimental plant infographics and pictures of the works are presented (Figure 2). In June 2021, the works will be accomplished and further updated information and pictures could be presented.

Figure 2. *Infographics and picture of INTEXT Plant in Talavera WWTP*

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DECENTRALISED WASTEWATER TREATMENT IN THE FRAME OF CIRCULAR ECONOMY IN URBAN AND RURAL AREAS IN INDIA: SARASWATI 2.0

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Abstract

Almost three-quarters of wastewater in India is discharged untreated and most of decentralised wastewater treatment plants in small agglomerations do not reach sufficient effluent quality. In that context, the Saraswati 2.0 project, jointly funded by the European Union and the Government of India, aims to identify the best available and affordable technologies (BAT) for decentralised wastewater treatment (WWT) with the scope of energy efficiency, resource recovery and reuse in urban and rural areas in India. With that aim, ten pilot technologies will be implemented across India, demonstrating enhanced removal of organic pollution (BOD, TSS), nutrients, organic micro-pollutants and pathogens.Recommendations on how to implement basic as well as advanced and fully automated control strategies for all piloted technologies based on decision support systems for plant operation will be elaborated. In order to identify BAT the test results of the monitoring will be complemented with a comprehensive sustainability assessment including LCA, LCC& affordability,

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that will include support to the implementation of the pilots with planning concepts to facilitate involvement of local stakeholder.

INTRODUCTION

A report by Central Pollution Control Board (CPCB, 2005) states that around 73 % of the sewage generated in India is discharged untreated because of limited treatment capacity of the existing treatment plants, and the situation has hardly improved since then (Chatterjee *et al.*, 2016). Furthermore, many centralized wastewater treatment plants (WWTP) across India do not reach sufficient effluent quality (Starkl *et al.*, 2018; Chatterjee *et al.*, 2016). In addition, centralized treatment plants may not be a sustainable solution for a developing country like India, where the power supply is rarely continuous, and the operation and maintenance cannot be ensured (Chatterjee *et al.*, 2016). On the other hand, decentralized WWT systems are widely used, specifically in South India but most of these WWTP are denounced owing to failures on several fronts including design, operation and maintenance, installation and monitoring (Suneethi, 2015). Depending on the site conditions and availability of a wide range of technologies, decentralized treatment units give the option of choosing a technology sustainable for local conditions (Chatterjee et al., 2016). Today innovative wastewater treatment plants are needed in India, which not only aim treating the wastewater, but also provide other benefits such as reuse of treated water, energy conservation/recovery, and nutrient recovery, depending on the local context (Chatterjee et al., 2016).

In that context, SARASWATI 2.0 was build. SARASWATI 2.0 is a four-year Horizon 2020 project jointly funded by the European Union's Horizon 2020 Research and Innovation programme under Grant Agreement n° 821427 and by the Department of Science and Technology (DST)/Department of Biotechnology (DBT), Government of India. The project finds its framework under the call topic: SC5-12-2018 EU-India water co-operation. It builds on previous EU-India projects, in particular on the SARASWATI project that was implemented from 2012 to 2017. The consortium of the project is comprised of a well-balanced EU-Indian team of 17 partners and is leaded by BOKU.

The aim of SARASWATI 2.0 project is to identify best available and affordable technologies (BAT) for decentralized wastewater treatment (WWT) with scope of resource/energy recovery and reuse in urban and rural areas in India. Further, it addresses the challenge of real time monitoring and automation. The previous SARASWATI project has shown that a number of decentralized wastewater treatment plants (WWTP) in India do not perform properly and that there are few plants that would meet the more stringent standards proposed by the Indian Government in 2015 (Singh *et al.*, 2015, Suneethi *et al.*, 2015, Chatterjee *et al.*, 2016, Singh and Kazmi, 2017; Kamble *et al.*, 2017, Singh *et al.*, 2018).

The SARASWATI 2.0 project adapt the definition of BAT to the local context, based on complementing the treatment efficiency with the costs of the technology and affordability, and local context in the location of application (Starkl *et al.*, 2018). This will allow to identify BATs with more stringent standards if required and suitable for the location. Thereby, ten pilot technologies in 7 Indian States demonstrating enhanced removal of organic pollution (BOD, TSS), nutrients (particularly Nitrogen), organic micro-pollutants and pathogens have been proposed. Further, all pilots permit resource recovery contributing to the principles of a circular economy and will undergo a comprehensive performance assessment complemented by a sustainability assessment informed by ISO standards. This will allow identification of BATs for the Indian context. In addition, suitable automation and control strategies will be tested and recommended, taking into account the presence of operators and their level of knowledge and expertise.

METHODS

Ten innovative technologies for WWT will be piloted across India. The locations chosen for implementing the 10 pilot technologies show a variety of India's water challenges and hence will contribute to find solutions for them. The identified candidates for BATs belong to the following four groups: a) Decentralized (smaller-scale) municipal WWT technologies (combined black- and greywater from domestic and communal sources), b) Black-water treatment technologies (domestic wastewater not including greywater), c) Sludge treatment technologies and d) Post treatment technologies for (conventionally) treated effluent (Tables 1, 2, 3 and 4).

Table 1. *Decentralized (smaller-scale) municipal WWT technologies (combined black- and greywater from domestic and communal sources)*

Pilot	Technology	Treatment/ use	Location	Pilot leaders EU/ India
1	UASB +Deammonification technologies	Carbon + nutrient removal/ Biogas production Water reuse	IIT Bhubaneswar (OD)	UT/ IITBBS
2	C-TECH SBR small package type plant	Carbon + nutrient removal Automatic control with SCADA/ Water reuse	Haridwar (UK)	IITR
3	BiokubeMars packaged decentralized WWTP	Carbon + nutrients removal + Disinfection Medical and domestic WW/ Water reuse	Jaipur (RJ) Kishangarh (RJ)	BioK/ MNIT BioK/CURAJ

4a	RMBR coupled with pressure sand filter and 4 disinfection units in parallel (electrochemical, UV, chlorination and ultrasound)	Carbon removal + Disinfection/ Water reuse (irrigation)	Bhandup-Mumbai Pumping station	CENTA/ NITIE

UT – Tartu University (Estonia); IITBBS - Indian Institute of Technology Bhubaneswar; IITR - Indian Institute of Technology Roorke; BioK - Biokube AS (Denmark); MNIT - Malaviya National Institute of Technology Jaipur (India); CENTA – Andalusian Public Foundation Centre of New Water technologies (Spain); NITIE - National Institute of Industrial Engineering (India)

Table 2. *Black-water treatment technologies (domestic wastewater not including greywater)*

Pilot	Technology	Treatment/use	Location	Partners
5	Anaerobic digestion (AD) coupled to photoheterotrophic bioreactor (PHBR)	Carbon + nutrients removal/ Resource recovery (nutrients, feed, food additives) Water reuse	IIT Kharagpur (WB)	TU Delft/ IIT KGP
6	Anaerobic digestion coupled with electrically conductive biofilter	Secondary + Tertiary WWT + Disinfection/ Water reuse (irrigation)	IIT Kharagpur (WB)	CENTA/ IIT KGP

TU Delft - Technische Universiteit Delft (Netherlands); IIT KGP - Indian Institute of Technology Kharagpur (India)

Table 3. *Sludge treatment technologies*

Pilot	Technology	Treatment/Use	Location	Partners
7	Cambi sludge digestion	Sludge digestion/ Resource recovery (fertilizer, biogas)	IIT Roorkee (UK)	Cambi/ IITR
8	Ultrasonic treatment of sludge	Sludge dewatering and disinfection/ Soil conditioning, composting	IIT Kharagpur (WB)	TU Delft/ IIT KGP

Cambi - Cambi Solutions AS (Norway); IITR – Indian Institute of Technology Rorkee (India)

Table 4. *Post treatment technologies for (conventionally) treated effluent*

Pilot	Technology	Treatment/Use	Location	Partners
4b	Disinfection of WWTP effluent by means of sand pressure filter plus chlorination, UV and ultrasound units	Water Disinfection/ Water reuse (irrigation)	Burhanpur (MP)	CENTA/ CEMDS/ NITIE
10	Nitrate removal using Ion exchange membrane bioreactor (IEMB) reactor	Nitrate removal/ Water reuse (environmental)	Chennai (TN)	BGU/ IITM

CEMDS - Zentrum fur Umwelt management und Entscheidungs theorie (Austria); BGU - Ben--Gurion University of the Negev (Israel); IITM – Indian Institute of Technology Madras (India)

A comprehensive monitoring regime will be conducted for 6-15 months for all piloted technologies, focusing on physical, chemical and microbial parameters relevant for performance assessment in the Indian context and will further encompass experimental work such as varying operational parameters for optimization of pilot. Depending on the presence of operators (or their absence, in which case we will pursue a fully automated control approach) and their level of knowledge and expertise, recommendations on how to implement basic as well as advanced and fully automated control strategies for all piloted technologies based on decision support systems for plant operation will be elaborated. In order to identify BATs the test results of the monitoring will be complemented with a comprehensive sustainability assessment including LCA, LCC& affordability, informed by recent ISO product standards in the sanitation sector (ISO 305 about non-sewered sanitation systems and ISO13065 about sustainability criteria for bioenergy). Further this objective will include support to the implementation of the pilots with planning concepts to facilitate involvement of local stakeholder.

RESULTS AND DISCUSSION

The first three groups are the most pressing problems for wastewater management in India, as these are major causes of pollution in India, and there are hardly any well-functioning decentralized technologies available. Group four has great potential to upgrade the many WWTP across India which do not reach sufficient effluent quality.

Pilot 1 has a huge potential as an energy efficient nitrogen removal process and hence also suitable for rural areas with unstable energy supply in India. The technology readiness level (TRL) is 4 for Indian context. Pilot 2 is up to 50 % savings of space and costs. C-TECH plant is successfully applied in India for plants larger than 1.0 MLD capacity. However, there are no experiences yet in India with smaller scale C-TECH plants. Hence, the TRL is 6 for Indian context. It has a huge scope of replication and upscaling as manufacturing can be easily set up in India with the help of SFC Environmental Technologies. Further, the plant is designed to operate fully automatically and is controlled by a programmable logic controller (PLC) connected with a SCADA. Pilot 3 provides for reliable operation without blockage of the filter media and without breakdowns in service. It can recover well from adverse events such as input flow surges and accidental inflow of substances harmful to the bacteria living on the filter media. It is already operating at 25 plants in India so it would be TRL 9. However, this pilot will test the plant with a mix of domestic and medical wastewater. It can be quickly replicated an upscaled across India. Regarding Pilot 4a, it is a reliable and robust WWT system. Although the technologies proposed in this pilot action are implemented worldwide, there are no experiences yet of their joint implementation in the same treatment process for water reuse in irrigation, which gives it a high innovative character globally. Hence, the TRL is 6 for Indian context.

In pilot 5, the TRL is 8 for the acidifying anaerobic digester, and 4 for PHBR. This pilot uses minimum energy input and also focuses on resource recovery through a very basic innovative design, which can be constructed easily using local resources. The ability to implement this technology at a decentralised household level, without much cost implication and lesser footprint on land, makes it more adoptable for upscaling in India, both at household community level as well as for centralised treatment facility. The electroconductive biofilter of pilot 6 (called METland®) is considered to be TRL 5 in India. Kharagpur and nearby states are facing water challenges where people are mostly relying on groundwater source for meeting the water demand and, in many cases, groundwater is polluted. This system can either be used as secondary or tertiary treatment as an add-on to existing treatment plants, or in combination with e.g. an UASB reactor as in this pilot for complete treatment of black water. It is a robust WWT system and therefore has a high potential for application in India.

In Pilot 7 the THP reduces sludge viscosity and increases its biodegradability and shortens hydraulic retention time of anaerobic digesters. TRL is 6. The process shall be applicable for pre-treatment of all kind of biological sludge generated from WWTP. It can be primary sludge, secondary sludge (waste activated sludge) or mixed sludge (primary+ secondary). Pilot 8 will reduce the residual organic matter in the sludge, decreasing the pathogen content. This disinfected sludge can be applied as soil conditioner, assuming limited heavy metal contamination and can serve for composting. Further, it is also expected to reduce refractory organics and organic micropollutants. The TRL of pilot 8 is 5. Bio-sludge management is increasingly in demand in India. The emergence of persistent pollutants in the sewage stream is a serious concern in the Indian context. Therefore, this technology has a great potential to help solving those water challenges across India.

Although the treatment units of pilot 4b are widely implemented worldwide, there are no experiences yet of the joint implementation of such treatments in the same flow sheet, mainly ultrasound, UV and chlorination, for water reclamation and reuse in agricultural irrigation, which gives it a high innovative character globally. Hence, the TRL is 6 for Indian context. There is a high potential for low cost post treatment systems to enhance effluents of existing treatment plants to be safe for reuse. In particular, there is a high demand for agriculture to use safe treated wastewater as more than 60 percent of India's irrigated agriculture is dependent on groundwater, which is depleting fast in many areas. In pilot 10, the treated effluent will be suitable for reuse by returning the purified water to surface water bodies, as a result of removing nitrogen forms with minimal energy input. The pilot unit has been applied to synthetic groundwaters; thus, the TRL is 4. It is simple to operate and flexible, since multiple modules in parallel allow increase in capacity, and in series allow to reach the nitrate removal that is required. The system can be used as add on units to any existing WWTP.

CONCLUSIONS

The project SARASWATI 2.0 has proposed to adopt the principle of best available technologies (BAT) in a more flexible way, adapting the definition of BAT to the local context, based on complementing the treatment efficiency with the costs of the treatment technology and affordability and local context in the location of application. This BAT concept has enormous significance when deciding what type of technology to choose in small towns and rural areas in India, as it ensures compliance with treatment standards at an affordable cost. As a general requirement, to be considered a BAT, the technology has to achieve the minimum existing WWT standard under the condition of average pollutant concentration.

However, this flexible definition will allow to identify BATs with more stringent standards if required and suitable for the location. Thereby, apart from enhanced removal of organic pollution (BOD, TSS), specifically technologies able to remove nutrients (in particular Nitrogen), organic micropollutants and pathogens have been proposed. More stringent standards related to pathogens to reduce health risks will also help to build uptrust among stakeholders for reuse of treated wastewater and other products and thereby encourage waterreuse and recycling.

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TWIST: INVOLVING STAKEHOLDERS IN THE INNOVATION PROCESSES AS THE ENGINE OF THE NEW KNOWLEDGE ECONOMY

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Abstract

The close relationship that exists between the development of a region and its level of technological innovation is unquestionable, the strengthening of which is essential as an engine for the economy and social challenges. Undoubtedly, in a new economy that is defined as the knowledge economy, R & D & I activities acquire an increasing weight within the framework of the same, acquiring a role as or more relevant than production processes.

Within this framework, the TWIST project (Transnational Water Innovation STrategy) is proposed, financed by Interreg Sudoe Program, seeks to promote innovation capacities for smart and sustainable growth through an innovative approach: a model of transnational and transregional organization and collaboration for the co-creation, experimentation, evaluation and market launch of innovative technologies and products in the water sector in which the participation of the quadruple helix represents an added value.

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INTRODUCTION

In the water sector, the vulnerability of water resources to climate change, the circular economy as a transition to the new economic development model, the duality of natural resource-productive asset, among others, are challenges that require new and better management approaches, efficient technologies and solutions adapted to the current context.

The lack of a common supra-regional framework that encourages investment in R&D and its transfer in the form of marketable solutions, makes it difficult to attract and retain qualified human capital and economic-social development based on knowledge and the circular economy.

In this context, innovation becomes an indispensable tool to respond to the new challenges that today's society demands. Thus, innovation is essential and decisive, requiring a good understanding of the needs of society and the market, the ability to explore new ideas as well as the skills of companies and administrations to implement new solutions.

Nowadays, any innovation process has to involve all stakeholders: technological agents (universities, public or private research centers, research companies ...), economic agents (the market, companies ...) and social agents (users, governments, civil society ...).

METHODS

The development of the TWIST project aims to address these challenges through the development of the following actions:

· Analyzing of the interest groups involved in R & D & i processes in the water sector, creating the basis that ensures that the stakeholders of the Quadruple Helix are integrated into the experimentation and capacity building processes, strengthening the territorial impact of the project.

· Characterizing of regional innovation processes for the capitalization of Research and Innovation strategy for Smart Specialization Strategies (RIS3)

· Developing of a common methodology for transnational experimentation for the creation of three Living Labs and training workshops to engage stakeholders.

· Promoting of cooperation with innovation platforms in water sector.

· Analyzing of opportunities for the development of new consortium and new R & D & i projects.

· Developing of three Living Labs in the field of wastewater treatment, regeneration, reuse, resource recovery and infrastructure management.

· Analyzing of market opportunities, feasibility studies, market barriers, trends, customer segments for a selection of research carried out in Living Labs.

· Developing of market plans for technologies / services developed in the Living Labs.

· Developing of pilot actions for Innovative Public Procurement (IP) in the field of water technologies that respond to common needs and procedures detected.

· Holding training sessions aimed at regional public administrations and companies.

· Approaching, structure and methodology for the creation of a Transnational Business School aimed at training SMEs, start-ups and entrepreneurs in eco-efficiency and circular economy.

· Analyzing of the complementarity of Smart Specialization Strategies, capitalization of TWIST good practices and support for the development of market plans in the SUDOE regions.

RESULTS

Table 1 shows the products resulting from the project as well as the stakeholders/beneficiaries to whom they are addressed, promoting the creation of new transnational and transregional partnerships between agents in the water sector capable of mobilizing and creating processes of R & D & i.

Table 1. *Stakeholders involved of the products obtained*

Results obtained	Stakeholders involved
TWIST mutual learning and capitalization strategy of RIS3	Administrations and companies
Specialized and complementary Living Labs in the field of wastewater treatment, regeneration, reuse, resource recovery and infrastructure management	Research groups, regional water administration, regional innovation administration and SMEs
CPI pilot cases in water sector	Administrations and SMEs
Transnational Business School	SMEs, Start-ups and entrepreneurs
TWIST Market Place	Research groups, SMEs and entrepreneurs

The inclusion of users, administrations, knowledge generators and companies is a complex process given the lack of mechanisms for the coordination and transfer of knowledge and R & D & I capacities.

In a global problem such as that of water, the transnational approach instead of the national, regional or local ones makes it possible to improve the efficiency of the methodological and technological responses to the challenges identified in the participating regions. Furthermore, these regions are at different levels of innovation and networking will help foster regional convergence. Transnational cooperation will generate new perspectives on their own regional strategies, which will promote the development of coordinated common strategies and methods.

CONCLUSIONS

Innovation is more than the technological factor: it is an essential part of the set of actions aimed at incorporating development strategies.

Stakeholders in the value chain, interested parties and their visions and interests must participate in it: technology providers, service providers, researchers and academics, relevant institutional actors (including administration) and end users (professionals or individuals), these one evolving from being a passive object at the end of the value chain to being an agent involved in the different phases of innovation.

It is an open innovation model that allows us to face new challenges with a new approach through the promotion of teamwork, collaboration and the exchange of ideas of the quadruple helix, fostering change through an increasingly participatory innovation ecosystem.

Involvement of the quadruple helix in the innovation processes implies a paradigm shift with respect to the R & D & I process, which is no longer a linear and sequenced concept in which the added value is provided by the final product, to be configured as an interactive, multidisciplinary and collaborative process, which favors the development of innovative solutions for real needs. Thus, both the final result and the process itself acquire value.

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LIFE CYCLE ASSESSMENT OF A WASTEWATER TREATMENT PLANT UNDER THE SCOPE OF CIRCRURAL 4.0 (SMALLWAT21)

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Abstract

The maintenance costs of aeration flow are one of the most significant economic factors in a wastewater treatment plant's operation (WWTP) (Liu, Li et al. 2011). The development of an off-gas analysis system capable of coupling the techniques for N_2O measurements and OTE evaluation was investigated and successfully applied in field measurements in the Charneca de Óbidos WWTP. The obtained results for OTE ranged between a minimum value of 3.2 and a maximum of 34.1, with an average value of17.97%5.38. The N_2O emitted during the field measurements was in the lower range with an average of 0.012 mgN-N2O/m3, even though the conformation type of the biological reactor at Charnecade Óbidos WWTP results in low emissions of N2O during the treatment by activated sludge there are other direct/indirect GHG emissions. Those emissions result in a Global Warming Potential (GWP) 100 years of 9.63 Kg CO_2 equivalent with a Eutrophication Potential (EP) of 6.75 Kg phosphate equivalent and an Acidification Potential of 6.82 Kg SO2 equivalent. The same approach will be applied in the Fregenal de la Sierra WWTP, operated by Promedio.

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INTRODUCTION

Wastewater treatment plants contribute to climate change by releasing greenhouse gas (GHG) into the atmosphere. Moreover, nitrous oxide (N_2O) has been reported as one of the most important sources of GHG in the wastewater treatment process with a Global Warming Potential (GWP) of 265 times higher than carbon dioxide (CO_2) being produced in nitrification/denitrification processes(Libra, Schuchardt et al. 2002, Liu, Li et al. 2011, Delre, ten Hoeve et al. 2019). Furthermore, the aeration tank, one of the key structures in wastewater treatment processes,is among thehighest energy-consuming steps of the WWTPs operationand represents between 40-60% of the total energy consumption(Redmon, Boyle et al. 1983, Liu, Li et al. 2011, Raghuvanshi, Bhakar et al. 2017). As a result, an evaluation of the environmental impact caused by water treatment and how it can be mitigated is of the utmost importance.

Life Cycle Assessment (LCA) is a valuable tool that can be used to evaluate the environmental impacts associated withWWTPs(Guest, Skerlos et al. 2009). LCA investigates the environmental impacts of systems or products from cradle-to-grave throughout the full life cycle, from the withdrawal, refining, and supply of materials and fuels, through the production and operation of the investigated objects, to their final disposal or recycling. Also, LCA provides a comprehensive set of environmental indicators that can be interpreted as impact indicators at a global and local level(Hospido, Moreira et al. 2004, Pasqualino, Meneses et al. 2009).

Our approach is to collect in-field data to perform an LCA analysis by developing an off-gas analysis system capable of coupling the techniques for N_2O measurements and oxygen transfer efficiency (OTE) calculation.

METHODS

To perform oxygen off-gas assessment and N_2O measurements, our sensorwasinstalledin Charneca de Óbidos WWTP (Águas do TejoAtlântico, SA). This WWTP treats 6.322 m^3/ of water per day (average value), coming essentially from industrialand from the population. After the treatment, water is discharged to Lagoa de Óbidos.Figure 1 shows the sensors in the aeration tank in one of the oxidation ditches.

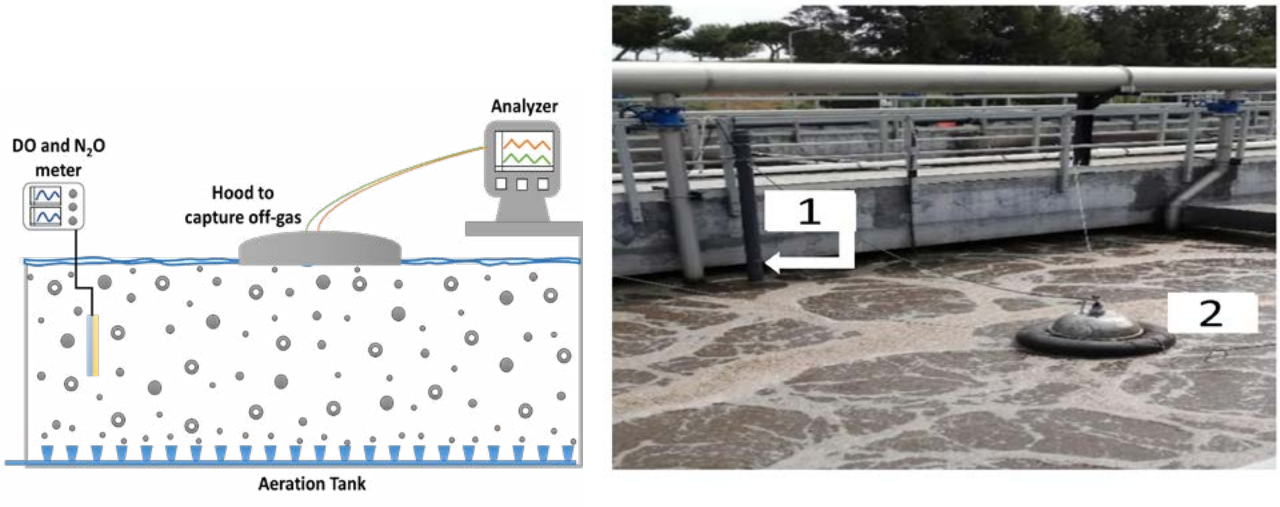

Figure1: *A –Sensors scheme assemble; B –Sensor installation (1 –submerged sensors; 2 –sensors in off-gas hood) The off-gas tool was installed in several different positions, inside the oxidation ditch, to determine if there were significant variations between locations. Figure 2 shows seven different locations where the off-gas tool was installed.*

The off-gas tool was installed in several different positions, inside the oxidation ditch, to determine if there were significant variations between locations. Figure 2 shows seven different locations where the off-gas tool was installed.

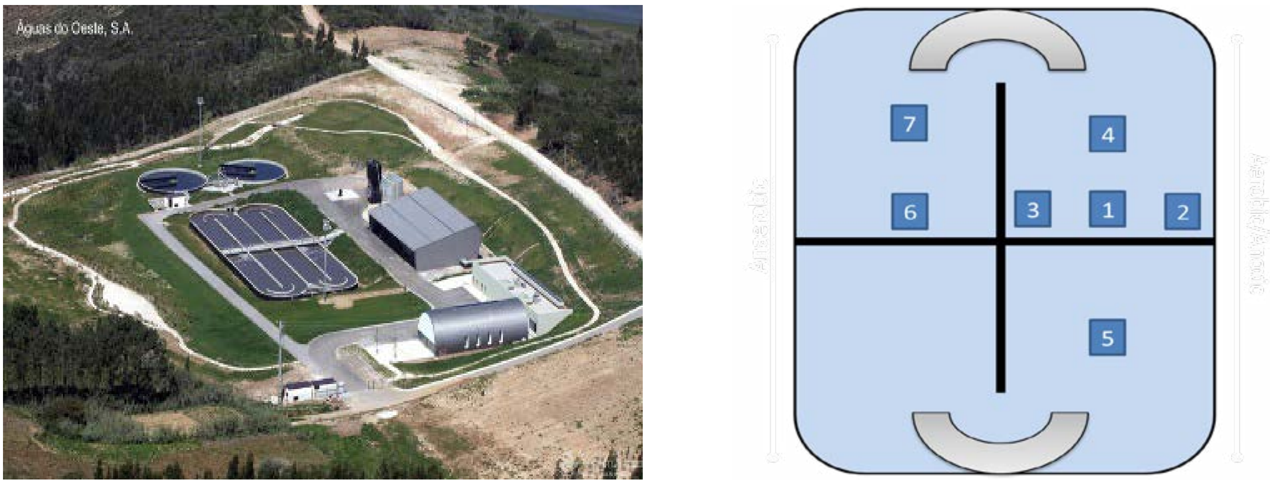

Figure 2: *Aerial view of Charneca de Óbidos WWTP and off-gas tool installation in the oxidation ditch*

The fowling equations were used to calculate N_2Oemissions andOTE (%) during the campaign.

$$EmittedN_2O_{(aeration)} = [\sum (C_{N_2O} \times Q_{gas(aeration)} \times \Delta t)] \quad \text{Equation 1}$$

$$EmittedN_2O_{(anoxic)} = [\sum (C_{N_2O} \times Q_{in(anoxic)} \times \Delta t)] \times (\frac{A_T}{A_{hood}}) \quad \text{Equation 2}$$

CN_2O (mgN-N_2O m^3) = CN_2O(ppmv N_2O)x(1/0.08205) (atm L mol^{-1} K^{-1})x[28/T(K)]; Qgas(aeration) - gas flow exiting the reactor in aerated zones (m^3 d^{-1}); Δt - time interval of operational periods; Qin (anoxic) - gas flow entering the sensor (0.5 mL/min); A_T - area of

$$OTE\ off-gas = \frac{Y_{in} - Y_{out}}{Y_{in}} \quad \text{Equation 3}$$

reaction tank (m^2); A_{hood} - area of analysis hood (m^2)
Y_{in} – molar fraction of oxygen gas in inlet gas; Y_{out} – molar fraction of oxygen gas in outlet gas
kLa – volumetric oxygen transfer coefficient (h^{-1}); Cs – O_2 saturation concentration (temperature dependable) (g/L); C – measured dissolved oxygen (g/L); V – tank volume (m^3); Q – tank aeration (m^3/h); p – O2 densitiy (g/L)

$$OTE\ OUR = \frac{kLa \times (Cs - C) \times V}{Q \times p} \quad \text{Equation 4}$$

RESULTS AND DISCUSSION

The data collected during the campaign (30 days), pointed to position 5 (Fig. 2) as the highest OTE zone. An average of 15.04% for OTE was established ranging between a minimum value of 6.5% and a maximum of 18.1%.To prove the reliability of the register values by the sensors the previously OTE percentage calculated was compared with other values described in the literature (Table 1).

Table1: *Results for off-gas analysis and comparison with literature*

OTE %	**Range**	6.5-18.1	8.0-16.7	12-20	-	-
	Avg.	15.4	11.85	-	12.25	14.57
	Ref.	Thisstudy	(Redmon, Boyle et al. 1983)	(Libra, Schuchardt et al. 2002)	(Iranpour, Magallanes et al. 2000)	(Iranpour, Magallanes et al. 2000)

The external factors that influence OTE like temperature, saturation concentration of oxygen in the ditch and other condition factors such as solids retention time or the presence of surfactants could explain the variability of the obtained results. Even though the values achieved by this study are higher than the ones reported in the literature the results were satisfactory and comparable for OTE estimation using the off-gas method.

For validation of this technique, the in-situ Oxygen Uptake Rate(OUR) method was used to calculate OTE (table 2), this allows to compare the variance between the two methods. The low variance (5.05%) existent between the two methodologies validates the application of this off-gas tool in diffused air systems analysis.

Table 2: *OTE values using off-gas method versus OUR method*

OTE (%)				
Off-gas method		**OUR method**		
Range	Average	Range	Average	Difference from off-gas (%)
6.5-18.1	15.04	4.5-25.2	14.27	5.05

Regarding N_2O emissions, it was observed that the dissolved N_2O was always approximately zero, proving the absence of N_2O accumulation. The average value of emitted N_2O was in the lower range (0.012 mgN-N_2O/m^3). Since N_2O emissions can be influenced by dissolved oxygen (DO) concentration in the aeration tank, insignificant variations in the DO (aeration flow average value - 1235.5 m^3/h) correlate to the low N_2O emissions during the campaign and the absence of N_2O emission peaks.

The results obtained during the field campaign and other data concerning the quality of the water, sludge management and the WWTP operation were provide by AdTA. This information was inserted into a specific LCA program and used for the impact calculation. The Life Cycle Impact Assessment (LCIA) results (Table 3) reveal that N_2O release during the water treatment process is not the main source of the pollution regarding the GHG effect. For this study only the volume of untreated water regarding de 30 days campaign were used which means the LCIA was determined for 4787 m^3 of untreated water.

Table 3: *LCIA results for the impact categories*

Impact Category	Excluding Energy	Including Energy	Reference Unit
Global Warming Potential 100 years	5.707E03	9.63E03	CO_2 eq.
Acidification Potential	-	6.82	SO_2 eq.
Eutrophication Potential	5.819	6.75	Phosphate eq.
Ozone Layer Depletion	-	1.18E-10	R11 eq.
Abiotic Depletion Elements	-	1.39E-3	Sb eq.
Freshwater Aquatic Ecotoxicity	-	7.93	DCB eq.
Marine Aquatic Ecotoxicity	-	4.51E05	DCB eq.
Photochemical Ozone Creation	-	0.523	Ethene eq.
Human Toxicity	-	175	DCB eq.

“eq.”- equivalent

Analyzing the LCIA results only two impacts results directly from the process of treating water all the others are indirectly cause from the consumption of electricity. Global warming potential (GWP) normally calculated over a particular period, normally 100 years is the most commonly chose, this impact is influenced by the production of CH_4, N_2O and other GHG as well as from the sludge management. Eutrophication Potential (EP) impact is influence by the efficacy on nutrients removal which is influenced by the DO rate on the oxidation ditch.

A new type of dissolved oxygen (DO) controllers wasdeveloped for this project and will be installed in this WWTP, these controllers will be responsible to control in real life the DO concentration on the oxidation ditch by turning on-off the aeration flow to always achieve the perfect DO that allows a correct denitrification process without a release of N2O. Through more efficient use of electricity in this process the overall cost can decrease as well asthe release of direct/indirect GHG for the atmosphere will be reduced. This process will be replicated and applied in the Fregenal de la Sierra WWTP, operated by Promedio in Spain which is a smaller WWTP but receives mainly untreated water from the population.

CONCLUSIONS

The off-gas tool is suitable to use in future studies for the measurement of N_2O level as well as to evaluate the OTE of the WWTPs. The use of both techniques results in an improvement of operational costs because the optimal OTE for the process can be obtained by adjusting aeration flow daily without compromising the compliance for recommended N_2O emission values and the global impact of these adjustments can be evaluated through a LCA.

ACKNOWLEDGMENTS

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CAN PERMEABLE PAVEMENTS ACT AS WATER STORAGE SYSTEMS? EVALUATION OF WATER TREATMENT CAPACITY AND REUSE

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Abstract

As a nature based solution, permeable pavements offer multiple benefits and ecosystem services when the water resource needs to be managed. Nevertheless, the urgency in transforming urban areas under a more sustainable model, has stimulated the need of the development of this technology. This work fouses on the study of the functioning ability of permeable pavements, as a water storage system. Several filling and emptying cycles of a permeable pavement pilot plant, at a laboratory scale, are simulated, analysing whether its capacity of treatment allows the reuse of managed water to satisfy an irrigation demand from Mediterranean areas.

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INTRODUCTION

Urban areas concentrate human activities, consuming a large amount of environmental resources (Grimm et al. 2008). Of particular concern are the challenges faced by these areas, such as climate change, population aging, or consumption of natural resources. In this context, a transformation of urban ecosystems is necessary to face these challenges (Elmqvist et al. 2008). The implementation of nature based solutions (NBS) must be addressed as a pillar of the paradigm of new sustainables cities. For example, permeable pavements as a sustainable urban drainage systems (SuDS) technique, offer multiple benefits and ecosystem benefits in terms of water quality and quantity. Several researches focus their studies on the capacity of pollutant removal in permeable sections, such as suspended solids, pathogens and even heavy metals. Additionally, the installation of permeable pavements in small urban agglomerations, provides operational protection of water treatment plants, based on artificial wetlands or lagooning. Permeable pavements reduce peak flow during rain events, and control suspended solids from entering the drainage system. Within the HOFIDRAIN Project framework, this study aims to investigate the suitability of giving permeable pavements a new function: underground storage of urban runoff and its reuse for a typical irrigation demand of the Mediterranean área. Despite the fact that urban runoff can contain high concentration of pollutants (Andrés-Doménech et al. 2018), the starting hypothesis states that stored runoff in the system, will be suitable for different urban uses, due to the treatment capacity of permeable pavements.

METHODOLOGY AND MATERIALS

The subterranean storage system provides collection and preservation of rain water, which can be used for different urban purposes, such as street washing and gardening. Since there is no legal framework that regulates its reuse in Spain, the water quality requeriments set out in R.D. 1620/2007, which establishes the Legal Regime for Treated Water Reuse for urban use, will be taken as reference (MINISTERIO DE MEDIO AMBIENTE Y MEDIO RURAL Y MARINO, 2010).

The simulation of the operation of the permeable pavement storage tank (PPST) during a rain event, considers that it receives both direct precipitation on the pavement, and runoff that reaches it. The ratio between the impervious surface that drains towards the permeable pavement is 2 to 1, that is, 2 m^2 of impervious surface for every m^2 of permeable pavement. Once the storage capacity of the PPST is completed, water levels inside it are managed according to a typical irrigation demand in Mediterranean areas. Direct precipitation is simulated by adding deionized water into the PPST, while runoff is simulated from real samples, taken in a small impervious catchment of 79 m^2 within the facilities of the Universitat Politècnica de València (UPV) (see figure 1).

Figure 1. *Photographs of the impervious catchment and the sampling chamber (Andrés-Doménech et al. 2018).*

To study the treatment capacity of the permeable pavement system, two samples are analysed: PPST input (PPSTi) and PPST output (PPSTo). The PPSTi sample consists of a representative mixture of runoff collected by an automatic sampler (ISCO3270) equipped with a liquid detector, and a set of 24 1-litre bottles. The Monitoring System is complemented with a 0.2 mm precision Detectronic rain gauge, with a Bühler Montec datalogger. This allows to record the hietogram of the rain and to determine the corresponding volume of deionized water to be added into the PPST. The water quality parameters analyzed are: organic matter, nutrients, total suspended solids (TSS), turbidity, electrical conductivity (E.C.), pH, dissolved oxygen, temperature and *Escherichia coli*.

The pilot plant is located in the Sanitary Engineering Laboratory, at UPV. It consists of a 0.25 m^2 permeable pavement system, designed to evaluate the hydraulic capacity of a permeable surface and quality of infiltrated water. The PPST is configured, from bottom to top, by 25 cm of washed limestone gravel + geotextile layer + 5 cm of fine washed limestone gravel + permeable block paving which joints are topped with small size gravel. Likewise, a drain pipe is placed on the bottom of the system, and connects to a flexible pipe that allows regulation of the water level inside the PPST. Periodically, samples from the PPST are taken to monitor the water quality inside the system. The extracted volume corresponds to the existing irrigation demand. In case of the hydraulic capacity of the PPST is exceeded (29 litres), water flows through the flexible pipe. Then, another sample is also taken (see figure 2).

Figure 2. *Different views of the PPST.*

RESULTS

At the time of writing this abstract, three rainfall events have been replicated in the pilot plant, which rainfall volumes were 8.36 mm, 15.22 mm and 28.45 mm respectively. The water quality parameters of the PPST input and ouput are shown in table 1. The results indicate a significant improvement in the water quality, confirming the capacity of treatment of the permeable pavement. It is noted that part of the improvement is due to the effect of dilution with rainwater. Therefore, other processes such as biodegradation and filtration help to improve water quality.

Tabla 1. *Aljibe's input and output water quality parameters (average values of the analysed events).*

Water quality parameter	PPSTi	PPSTo
COD (mg/l)	49	7.5
TN (mg/l)	3.00	0.9
TP (mg/l)	0.278	0.015
TSS (mg/l)	111	5.5
Turbidity (NTU)	108	2.7
E.C.$_{25°C}$ (μS/cm)	222	200
E. coli (UFC/100 ml)	4.3	1

For water reuse purposes, it is remarkable the reduction of suspended solids and organic matter, since they could cause clogging problems in irrigation systems. It is also worth noting the low hydraulic conductivity of the stored water, which is beneficial to avoid soil salinization. In this case, it is not observed a decrease between PPSTi and PPSTo. The effect of dilution by rainwater is balanced by the contribution of salts, contained in the materials that configure the pilot plant. Finally, in relation to microbiological quality, the stored water meets the requirements established in RD 1620/2007 for urban uses in Spain (E. coli <200 CFU / 100 ml, TSS <20 mg / l, turbidity <10 NTU).

CONCLUSIONS

The main conclusion that can be drawn from the first tests, is that permeable pavements hold high treatment capacity. This fact means that permeable pavements can act as water storage systems for later reuse for urban uses, complying with legal requirements on the reuse of treated water. The availability of this high-quality water resource at a city scale, can lead to significant energy savings in drinking water supply and well pumpings. Therefore, a new functionality of the technology is presented, which is recommended to be incorporated in the transformation path towards the more sustainable city models. The study reveals a new ecosystem service, provided by this type of nature based solutions, in addition to those already known, such as heat island effect reduction, infiltration improvement or flood risk reduction.

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SOLAR PROCESSES AND OZONE AS ALTERNATIVE TREATMENTSFOR WASTEWATER REUSE IN AGRO-FOOD INDUSTRIES: WATER, CROPS AND RISK ASSESSMENT

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Abstract

Water-food challenges to current and future sustainability agriculture are increasing as a consequence of water stress. In this regard, agro-food industrial wastewater reuse in agriculture represents an unconventional water supply to improve the water use efficiency and ameliorate this environmental problem. Among the agro-food industries, the fresh-cut industry stands out for its large market and its high-water consumption. Nevertheless, the application of the chlorination treatment during vegetable washing and the European concern about the generation of unhealthy disinfection by-products (DBPs) has resulted in the forbiddance of chlorination practice and in a notable increase in the evaluationof alternative water treatments to apply in this industry.In this regard, a full

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cycle of agricultural reuse (treatment, storage and in-vivo irrigation assays) of synthetic fresh-cut wastewater (SFCWW) reclaimed by two solar processes (H_2O_2/solar and Fe^{3+}-EDDHA/H_2O_2/solar) and ozoneat pilot plant scalehas been investigated in this study. The results showed a high capabilityof the three processes studied to significantly reduce the microbiological (*E. coli* O157:H7 and *Salmonella enteritidis*) and chemical (5 organic microcontaminants, OMCs) wastewater contamination, their translocation to the irrigated vegetables and the human risk (chemical and microbiological) associated with their consumption.

INTRODUCTION

The continuous global population growth along with water scarcity as a consequence of climate change are generating new water-food challenges. Among the different sectors, the agrifood industries represent one of the main economic activities where the implementation of a circular economy through a competitive and environmentally sustainable wastewater treatment and reuse strategy can provide important benefits. Specifically, the high water footprint of the fresh-cut industry (water consumption of ca. 40 m^3 per ton) and the common environmental discharge of the chlorinated wastewaterconverts this industry into a clear example of the potential benefits derived from the water reuse strategy.

Due to the low cost and high disinfection efficiency of the water chlorination process, the addition of chlorinated compounds during industrial processingis the most commonly applied strategy to decrease the microbiological risk associated with the consumption of fresh-cut products. Nevertheless, the presence of high content of dissolved organic matter in fresh-cut water lead to the generation of harmful chlorinated disinfection by-products (DBPs) and hence, to theforbiddance of the chlorination practice in some European countries as well as to increase to concern about chlorinated wastewater reuse in other activities such as agriculture. Consequently, the implementation of alternatives technologies that ensure microbiological safety without DBPs generation and allowing the agricultural reuse of the wastewater generated represents a viable alternative for the simultaneous reduction of the water footprint in these industries and water scarcity (Inyinbor*et al.* 2019).

In this regard, advanced oxidation processes (AOPs) have shown a high treatment capability to disinfect a wide range of complex water matrixes included fresh-cut wastewater (Huang *et al.* 2018; Selma *et al.* 2008a-b). However, and although it is not yet regulated and studies on their removal are lacking, the fresh-cut wastewater is also contaminated by OMCs (usually pesticides) which accumulation control during the industrial processing stage is also a key issue to reduce the further potential health and environmental impact (Campos-Mañas*et al.* 2019). On the other hand, the use of solar radiation as the source of photons, so-called solar-driven AOPs and solar photochemical processes have proven to disinfect and decontaminate fresh-cut wastewater without DBPs generation being also competitive in implementation costs (Nahim-Granados et al., 2020a-b).

In line with this, this study aims to assess the feasibility of a conventional process (ozone) and two solar processes (H_2O_2/solar and Fe^{3+}-EDDHA/H_2O_2/solar) as alternative treatments to chlorination studying their suitability in a full cycle of agricultural reuse: to improve the microbiological (*E. coli* O157:H7 and *Salmonella enteritidis*) and chemical quality (5 OMCs: atrazine (ATZ), azoxystrobin (AZT), buprofezin (BPF), procymidone (PCM) and terbutryn (TBT)) of synthetic fresh-cut wastewater and to reusethe treated wastewater for in-vivo irrigation of two raw-eaten crops (radish and lettuce) studying subsequently, contaminants translocation to the irrigated vegetables and the potential human risk (chemical and microbiological) associated with their consumption.

METHODS

The experimental study has been carried out entirely in the facilities of the Plataforma Solar de Almería, using lab-made synthetic water (synthetic fresh-cut wastewater, SFCWW)as a water matrix, thus avoiding composition variation of the industrial wastewater water (Nahim-Granados *et al.* 2018). This water matrix was spiked with a cocktail of 5 OMCs (100 µg/L each) and two foodborne pathogens (*E. coli* O157: H7 and *S. enteritidis*, 10^6 CFU/mL each). Liquid chromatography with ultraviolet detection (HPLC UV-DAD, Agilent 1260) andmembrane filtration method using selective agar media (detection limit of 1 CFU/100 mL) was used for OMCs and microbial quantification, respectively (Nahim-Granados *et al.* 2020a).

For solar tests (H_2O_2/solar with 20 mg/L and Fe^{3+}- EDDHA/H_2O_2/solar with 2.5+20 mg/L), a Compound Parabolic Collector (CPC) photo-reactor consisting of 2 modules of CPC mirrors placed on an anodized-aluminium platformwith 10 borosilicate-glass tubes each with 60 L of total volume was used. On the other hand,ozonation tests (10 L) were performed in a column type ozone reactor with constant production of 0.9 gO_3/ h(Nahim-Granados *et al.* 2020a-b). Reagents concentration (Fe^{3+} -EDDHA, Sequestrene 138 Fe G100, Syngenta, Spain; and H_2O_2, 30% w/v, Merck, Germany) and lettuce chlorophyll content were determined by spectrophotometric methods (Nahim-Granados *et al.* 2019; Standard Methods, 1998).

The reuse irrigation assays were performed under controlled conditions using a 30 m^2-experimental greenhouse divided into 4 individual areas of 7.5 m^2 (Suministros D.R., Spain). Lettuce(*Lactucasativa var. longifolia,* cultivation period of 12 weeks) and radish (*Raphanussativus L.*,cultivation period of 6 weeks) irrigation tests were done simultaneously: 100 pots per each type of crop and irrigation condition (controls and SFCWW treated by ozone, H_2O_2/solar and Fe^{3+}-EDDHA/H_2O_2/solar) were placed in an individual area. The microbiological and chemical analysis of the harvested crops (33 out of 100 samples were analyzed) was carried out based on a previous study (Aguaset *al*. 2020) and the real data obtained from this analysis was used to perform chemical and microbiological quantitative risk assessmentsbased on the guidelines of theEuropean Food Safety Authority (EFSA) and with dose-response models and Monte Carlo simulations (software FDA-iRISK®), respectively.

RESULTS AND DISCUSSION

The treatment performance of the three processes understudy (ozone, H_2O_2/solar and Fe^{3+}-EDDHA/H_2O_2/solar) for bacterial inactivation and OMCs removal was assessed at 240 and 300 min of treatment time for ozone and solar processes, respectively (Table 1). The experimental results obtained showed high efficiency of all the processes studied to reduce the microbiological contamination until levels that satisfied the microbiological quality of the new European Regulation(EU 2020/741) even for the more restrictive quality (*E.coli*≤ 10 CFU/ 100 mL).On the other hand, concerning OMCs, their concentration was reduced by all the processes studied showing the ozonation process a high efficiency (OMCs reduction of ca. 90%) and a moderate efficiency was shown by the solar processes: ca. 20 and 40 % for H_2O_2/solar and Fe^{3+}- EDDHA/H_2O_2/solar, respectively.

Table 1. *Average OMCs concentration and bacterial load in untreated and treated SFCWW employed to crops cultivation.*

Treatment	OMCs, Total load (µg/L)	*E. coli* O157:H7 (CFU/100 mL)	*S. enteritidis* (CFU/100 mL)
Untreated	500	10^8	10^8
Ozone	52±24	< 1	< 1
H_2O_2/solar (20 mg/L)	397±96	7±2	1±1
Fe^{3+}-EDDHA/H_2O_2/solar (2.5+ 20 mg/L)	299±71	1±1	1±1

Once a suitable treatment performance that allows subsequent agricultural reuse was confirmed, in-vivo irrigation essaysusing non-treated SFCWW (lab recipe spiked with the target contaminants) and treated SFCWW for lettuces and radishes cultivation were performed. The microbiological analysis of the harvested crops irrigated by non-treated SFCWW showed a high concentration (> 60 CFU/g) of both bacteria in all the samples analysed, whereas the crop samples irrigated with treated SFCWW by the three processes studied revealed a complete absence of microbial contamination (<LOD). For OMCs, in comparison with the results obtained for crops irrigated by untreated SFCWW, significant reductions of OMCs uptake were observed: > 90% in both crops irrigated by ozonated SFCWW and reductions of ca. 70 and 50 % for lettuces and radish irrigated by solar treated SFCWW. Moreover, the chlorophyll content of the harvested lettuce leaves was analysed to establish a potential physiologic plant benefit derived from the use of the commercial iron micronutrient as a catalyst in the solar photo-Fenton process. The results obtained from these analyses showed twice chlorophyll content in the lettuce irrigated by the solar photo-Fenton process with respect to those irrigated by the other two processes that do not incorporate the iron fertilizer.

Table 2. *Mean risk of illness obtained from quantitative microbial risk assessment for the consumption of the harvested crops irrigated by untreated and treated SFCWW*

	Untreated SFCWW	Treated *SFCWW*
***E.coli* O157:H7**		
Radish	0.62	5.05×10^{-5}
Lettuce	0.70	5.13×10^{-5}
S.enteritidis		
Radish	0.29	1.09×10^{-6}
Lettuce	0.33	1.11×10^{-6}

Finally, the potential chemical and microbiological health risk associated with the consumption of the harvested crops was assessed through a dietary risk assessment for the combined exposure of the OMC residues and a quantitative microbial risk assessment (QMRA), respectively. The results obtained from the chemical risk assessment showed the absence of a significant chemical risk for the consumption of the harvested crops irrigated by both, untreated and treated SFCWW being significantly lower the risk associated with crops irrigated by reclaimed wastewater. However,the QMRA results obtained (Table 2) showed that the crops irrigated by untreated SFCWW represent asignificant source of infection for the consumer (both crops and both bacteria) whereas the infection risk associated with the consumption of crops irrigated by treated SFCWW showed a reduction of at least 4 orders of magnitude.

CONCLUSIONS

This study showed the capability of ozone and two solar processes to efficiently reclaim agro-food wastewater until comply with the more restrictive values of the European Regulation 2020/741. The reuse of the reclaimed wastewater to irrigate two different raw-eaten crops (radish and lettuce) has demonstrated to significantly reduce pathogen load and OMCs uptake in both crops. Moreover, a physiologic plant benefit derived from the use of a commercial iron micronutrient (Fe^{3+}-EDDHA) as the iron source to reclaim wastewater by solar photo-Fenton was confirmed. Finally, the agricultural reuse of SFCWW treated by the proposed treatmentssignificantly reduces the human risk (chemical and microbiological) associated with crops consumption.

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COMPARISON OF UVC/H2O2 AND UVC/S2O82– PROCESSES FOR SIMULTANEOUS REMOVAL OF MICROCONTAMINANTS AND BACTERIA IN SIMULATED MUNICIPAL WASTEWATER AT PILOT-SCALE

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Abstract

The world is undergoing socio-economic changes that involve important environmental problems, and one of them is freshwater scarcity. Thus, the effectiveness of UVC-based advanced oxidation processes (UVC AOPs) for the simultaneous elimination of organics microcontaminants (OMCs) and pathogens in a simulated municipal wastewater (SMWW) at pilot-scale has been successfully investigated.UVC/H_2O_2 and UVC/$S_2O_8^{2-}$ processes have been compared in terms of the required treatment time to remove at least 80% of the sum of OMCs, bacterial inactivation, regrowth assessment and energy consumption. Despite the UVC/H_2O_2 and UVC/$S_2O_8^{2-}$ processes showing similar results to simultaneously eliminate OMC and bacteria, UVC/H_2O_2 process did not exhibit bacterial regrowth at dark conditions. According to the results and taking into account the required microbiological detection limit (DL) of 1 CFU 100 mL^{-1} for reclaimed

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water, the UVC/H_2O_2 process is a feasible alternative for wastewater reuse purposes.

INTRODUCTION

Natural water contamination by contaminants of emerging concern (including OMCs and pathogens) has been widely pointed out as one of the main challenges for safe reclamation of conventionally treated municipal wastewater.Unfortunately, consolidated tertiary treatments, such as UVC radiation, ozonation and chlorination are not effective enough or present serious drawbacks in their application to remove these contaminants. On the other hand, AOPs have been widely proposed as suitable technologies due to the wide variety of highly reactive free radicals they can produce, *e.g.*, hydroxyl ($HO^{\cdot}$), sulfate ($SO_4^{\cdot-}$), chlorine ($Cl^{\cdot}$), carbonate ($CO_3^{\cdot-}$), etc., which might, in turn, lead to high rates of OMCs and pathogens elimination (Babaei and Ghanbari 2016, Sánchez-Montes *et al*., 2020, Xiao *et al*., 2019).

UVC/hydrogen peroxide (H_2O_2) is one of the most disseminated AOPs used for organic compound degradation and water disinfection (Guan *et al*., 2018, Miklos *et al*., 2018). In this process, $HO^{\cdot}$ species can be produced from the homolytic cleavage of H_2O_2 by absorption of UVC radiation (λ<280 nm). AOPs based on $SO_4^{\cdot-}$ species have been reported in the literature due to their property to remove recalcitrant OMCs, and more recently, for inactivation of microorganisms (Guerra-Rodríguez *et al*., 2018, Liu *et al*., 2020). This radical has an oxidation potential comparable to the $HO^{\cdot}$and can be also generated by the activation of persulfate ($S_2O_8^{2-}$) under UVC irradiation (Yang *et al*., 2010). However, although previous studies have investigated the efficiency of different UVC AOPs (even UVC/$H_2O_2$and UVC/$S_2O_8^{2-}$) to eliminate OMCsor pathogens from water, decontamination and disinfection processes have been studied independently and only few works reported on the concomitant achievement of OMC removal and elimination of microorganisms.

Within this framework, this work aimed to investigate the effectiveness of UVC/H_2O_2 and UVC/$S_2O_8^{2-}$ processes for the simultaneous decontamination and disinfection of a simulated municipal wastewater (SMWW) effluent at pilot scale.UVC AOPs were compared in terms of total OMC degradation, bacterial inactivation, regrowth assessment and energy consumption. In addition, the influence of the oxidant residual concentration on the bacterial regrowth has been analyzed for assuring the safe application of such regenerated wastewater in agricultural activities. Six OMCs, acetaminophen (ACT), caffeine, (CAF), carbamazepine (CBZ), trimethoprim (TMP), sulfamethoxazole (SMX) and diclofenac (DCF) were chosen as target molecules since they are usually detected in municipal wastewaters. In addition,*Escherichia coli*(*E.coli*), *Enterococcus faecalis*(*E. faecalis*) and *Salmonella enteritidis* (*S. enteritidis*) were selected as target microorganisms because they are used as pathogen indicators in regulations and guidelines for wastewater disposal and reuse (ISO, Switzerland, 2015, USEPA, Washington, 2012).

METHODS

The concentration of OMCs (100 μg L^{-1} each) was monitored by ultra-performance liquid chromatography with a UV-DAD detector (Agilent Technologies, Infinity Series 1200) by using a Poroshell 120 EC-C18 column as the stationary phase (Agilent Technologies: 50 mm×3.0 mm, 2.7 μm particle) and a mixture of 25 mmol L^{-1} formic acid and ACN as the mobile phase in a gradient elution mode at 1 mL min^{-1}. The DL for each one of the investigated compounds was 5 μg L^{-1}. Considering environmental regulations already established for OMC elimination (FEO, Switzerland, 2017), experiments were performed with the aim of removing 80% of the sum of OMCs($\sum C/\sum C_0$). H_2O_2 and$S_2O_8^{2-}$were determined by titanium (IV) oxysulfate and iodometry method, respectively, by using a UV-Vis Evolution 220 spectrophotometer (Thermo scientific). Ion chromatography was used to measure anion and cation concentrations in previously filtered (0.45 μm nylon filter) samplesby using a Metrohm 850 Professional analyzer.

Selected strains of bacteria were provided by Spanish Culture Collection (CECT): *E. coli* (O157:H7) (CECT 4972), *E. faecalis* (CECT 5143) and *S. enteritidis* (CECT 4155). These strains were used to prepare the microbial suspensions spiked in the SMWW effluent. A suspension of each bacterial strain (10^5 CFU 100 mL^{-1}each) was prepared according to a previously published procedure (Nahim-Granados*et al.*, 2019). Bacterial quantification was performed by the standard plate counting and filtration method by using specific culture media for each bacteria. The DL obtained (1 CFU 100 mL^{-1}) is in accordance with the minimum disinfection level required in some regulations for reusing reclaimed wastewater (Regulation (EU) 2020/741), as well as those established in the regulations for drinking water (<1 CFU 100 mL^{-1}) (Directive EU, 2020/2184). Bacterial regrowth was quantified in predetermined samples stored at room temperature for 24, 48 and 144 h (6 days).Other studies report assessment of bacterial regrowth but only after 24 and 48 h.

UVC, UVC/H_2O_2 and UVC/$S_2O_8^{2-}$ experiments were carried out by using a UVC pilot plant. Briefly, the pilot plant consists of three medium pressure UVC lamps (230 W; mainly □ = 254 nm) protected by quartz tubes and axially located in a stainless steel cylindrical photo-reactor. In this study, a single lamp was used in batch mode (recirculation flow rate 36 L min^{-1}) with a total working volume of 80 L.More details about the UVC pilot plant can be found in Cerreta*et al.*, 2020. Finally, the accumulative UVC energy per liter (Q_{UVC};kJ L^{-1}) required in these processes was calculated according to Eqn.1:

$$Q_{UVC}(\mathrm{kJ\ L^{-1}}) = \mathrm{Dose}\,(\mathrm{kJ\ m^{-2}})\,\frac{S_p(\mathrm{m^2})}{V_T(\mathrm{L})} \quad (1)$$

whereDoseis the product of emitted irradiance by the UVC lamp (84.8W m^{-2}) multiplied by the illumination time fraction (s). S_p is the total irradiated surface of the photo-reactor (0.34 m^2) and V_T is the total working volume (80 L).

RESULTS AND DISCUSSION

The effect of H_2O_2 and $S_2O_8^{2-}$ concentration on total OMC degradation is shown in Fig. 1a. The use of the oxidants in combination with UVC irradiation increased the removal of all target compounds compared with UVC alone, resulting in a degradation rate of over 80% for the sum of OMCs under all investigated conditions. This effect was mainly caused by the effect of $HO^{\cdot}$ and $SO_4^{\cdot-}$ generated by the homolysis reaction of H_2O_2 and $S_2O_8^{2-}$, respectively. Illumination time (and accumulative UVC energy) required to achieve 80% of total OMC degradation decreased with the increase in oxidant concentration, *e.g.,* from 180 min (3.8 kJ L^{-1}; only 60%of OMC elimination was attained)without oxidant to 24 min (0.4 kJ L^{-1}) and 90 min (1.8 kJ L^{-1}) with 25 and 20 mg L^{-1} of H_2O_2 and $S_2O_8^{2-}$, respectively (see Table 1). In addition, higher values of pseudo-first order kinetic constants for total degradation were observed in the UVC/H_2O_2process compared to UVC/$S_2O_8^{2-}$,probably due to the higher quantity (in terms of mol) used of this oxidant in comparison with $S_2O_8^{2-}$.Other parameters, such as pH, conductivity, turbidity, dissolved organic and inorganic carbonand ion concentration were also monitored but had an insignificant variation throughout experiments.

Table 1. ***Pseudo-first order kinetic constants (k) fortotal OMC degradation in a SMWW effluent by UVC/H2O2 and UVC/S2O82– processes.***

Process	***k*(10⁻² min⁻¹)**	time[a] (min)	Q_{UVC}[b] (kJ L^{-1})
UVC alone	0.5 (0.82) only 60%	180	3.8
UVC/H_2O_2 (25 mg L^{-1})	5.7 (0.99)	24	0.4
UVC/H_2O_2 (50 mg L^{-1})	10.0 (0.99)	17	0.3
UVC/$S_2O_8^{2-}$ (20 mg L^{-1})	1.6 (0.99)	90	1.8
UVC/$S_2O_8^{2-}$ (40 mg L^{-1})	4.2 (0.98)	45	0.9

[a] values refer to 80% of total OMC removal, except for the UVC alone experiment (only 60% was attained)

[b] accumulative UVC energy required to attain 80% of total OMC removal

values in parentheses refers to coefficient of determination (R^2)

Regarding bacterial inactivation, the DL (removal of 5-log) for *E. coli*, *E. faecalis* and *S. enteritidis* was achieved in both processes in the concentrations of oxidants investigated, as it can be observed in Fig. 1b. Similar to the UVC results (data not showed), a double log-linear kinetics was observed for all bacterial inactivation. In particular, inactivation kinetics of *E. faecalis* and *S. enteritidis* were slower in the UVC/$S_2O_8^{2-}$ process than UVC/H_2O_2 system (45 min and 0.2 kJ L^{-1} for all bacteria). In contrast, DL for *E. coli*inactivation was achieved only after 15 min (0.2 kJ L^{-1} accumulated UVC energy) for 20 mg $L^{-1}$of $S_2O_8^{2-}$, which is possibly associated with the highly selective reactivity of $SO_4^{\cdot-}$ species towards electron-rich moieties on the surface of *E. coli*. Wordofa *et al*.,also showed that exposure to $SO_4^{\cdot-}$promoted the loss of cell viability of *E. coli*O157:H7 5 times faster than when $HO^{\cdot}$ was generated.

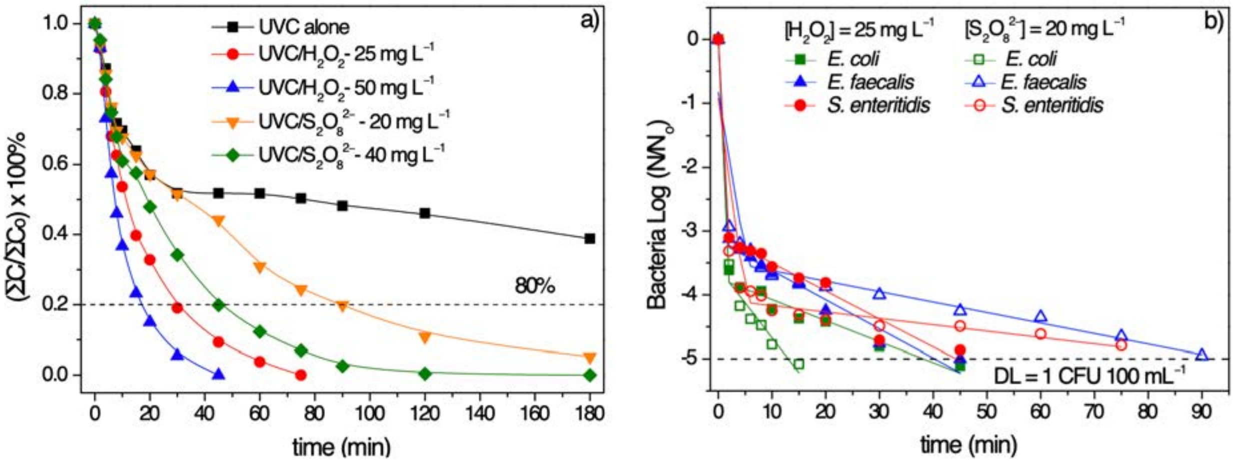

Figure 1. *a) Total OMC degradation and b) bacterial inactivation in SMWW effluent by UVC/H2O2 and UVC/S2O82– processes as a function of treatment time. Dashed lines refer to 80% removal of total OMC (∑C/∑C0) and DL = 1 CFU 100 mL–1.*

Finally, bacterial regrowth assessment was carried out after 24, 48 and 144 h(6 days) after the treatment was finished. Regrowth for all bacteria was detected for UVC/$S_2O_8^{2-}$, but much lower compared with that using only the UVC treatment (data not showed). *E. coli* regrowth was significant, increasing the concentration of viable bacteria after 24 and 48 h at dark conditions, until 1.3 and 1.9-log, respectively, remaining almost constant after 144 h. This means that the residual concentration of $S_2O_8^{2-}$ (18 mg L^{-1}) after treatment did not prevent bacterial regrowth since $S_2O_8^{2-}$ has no bactericidal effect by itself. However, in the presence of H_2O_2, regrowth was not observed after the treatment for all times analyzed. In this sense, residual H_2O_2(23 mg L^{-1}) had a potential further bacteriostatic effect, preventing bacterial repair/reproduction during the storage or through the distribution system.

CONCLUSIONS

UVC wastewater treatment was not suitable by itself due to a very slow and incomplete removal of OMCsaccompanied by a subsequent bacteria regrowth. By adding 25 mg L^{-1} H_2O_2 under UVC light, 4-log of *E. coli*, *E. faecalis*and *S. enteritidis* bacterial inactivation (without subsequent regrowth) and total OMC degradation rate higher than 80% were attained in less than

30 min.*E. coli* inactivation was attained in only 15 minby using UVC/$S_2O_8^{2-}$ (20 mg L^{-1}) due to the possible reaction of generated $SO_4^{\cdot-}$ with macromolecules in the cell wall.However, regrowth of bacteria was not prevented, possibly due to the limitation of $S_2O_8^{2-}$ diffusion through the cell membrane. In this sense, UVC/$S_2O_8^{2-}$ process would need a slight post-addition of bactericidal species to avoid bacterial regrowth along water storage or reclaimed water distribution systems. Finally, UVC/H_2O_2 treatment was able to produce an effluent with enough quality to be reused for several purposes, being crops irrigation as one of the most suitable end-uses as it is the highest consumer of freshwater worldwide.

ACKNOWLEDGMENTS

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07

INTENSIVE TECHNOLOGIES AND NUTRIENT REMOVAL

RISKS IN SUSTAINABLE MANAGEMENT OF WASTEWATER TREATMENT PLANTS OF SMALL VILLAGES ASSOCIATED WITH THE OBJECTIVES OF NUTRIENT CONTENT REDUCTION

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Abstract

The number of treatment plants in small villages has undergone a strong increase in the last decade as a result of regulatory changes. Thus, in the case of the province of Badajoz, a total of 90 treatment facilities with less than 2 000 inhabitant equivalent are to be built before 2027, a number that is higher than those already built and currently operating. The most widespread management model for treatment services in this province is a public-private collaborative management model led by PROMEDIO, which is based on economies of scale to ensure the sustainability of the plants. Inthe case of the new infrastructures, 90% of them must reduce the nutrient content oftheir influent waters, as established by authorities responsible for water quality, a fact that will generate a future scenario in which the management of the plants will suffer the risk of unbalancing the economy of scale. Thus, the present work addresses the need for reflection faced by the management models of treatment services, since the boom in the construction of small treatment plantsdemands a necessary evolution of these models to ensure both sustainability and quality of treated water from small wastewater treatment plants.

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INTRODUCTION

After modification of article 325 of the Penal Code imposed by the Organic Law 1/2015 of March 30, in which discharges threateningwater quality are classified as punishable offenses, and after the last extension granted for compliance, in2027, of the achievement of the good state of the water bodies established by the Water Framework Directive, the building of new treatment infrastructures, mainly in small villages, has experienced a strong boost.

The Emission Limit Values (ELV) established in the discharge authorizations for the aforementioned facilities are based on a normative criterion that leaves total freedom to local authorities, since it only establishes the need to provide an "Adequate Treatment" to urban agglomerations with less than 2,000 inhabitants equivalent (IE) [91/271/EEC]. This fact has generated significant differences betweenregions. So, as an instance;in the Guadiana basin, province of Badajoz, ELVs for Total Nitrogen, Total Phosphorus and Ammonium for populations of less than 2,000 IEare more restrictive than those for populations of more than 10,000 IE, while in the Guadalquivir basin only a reduction of the content of organic matter and solids is required.

In this way, wastewater treatment technologies for small villages with nutrient removing requirements present unaffordable implementation and exploitation costs, putting at risk the management of thosefacilities.

Thus, in the present work, the case of small villages in the province of Badajoz is presented, in which the weight of nutrients in the effluent is much higher than that in large villages, since their industries, characterized by small agro-livestock farms connected to the sewage network, generate a high nutrient load. This fact, along with very restrictive ELVs, demand the implementation ofdepuration technologies whose costs are higher than those provided by standardones that allowed reaching the classic limits for organic matter and suspended solids. Therefore, the treatment scenario these villagesfaces shows a complex vicious circle in whichthe need to implement treatment technologies to reduce the impact of pollution on the quality of water bodies avoiding legal problems, gives rise todisproportionate operating and maintenance costs that put at risk the sustainable management of the treatment services, which increases the risks of non-compliance with the discharge authorizations, driving the facilities to the initial situation of unlawfulness. In this way, the present work provides a necessary reflection to address the management of new treatment infrastructures which represents an evolution of current management models in order to minimize the risks to the sustainability of the depuration services in small villages.

METHODS

Emission Limit Values in small villages.

As it has been previously pointed out, the case of the province of Badajoz is studied.In it a total of 90 treatment plants are to be built in villages of less than 2,000 IE. Of these, 90% discharge their effluents to the corresponding Public Hydraulic Domain (PHD) of the Guadiana basin and the rest to the PDH of the Guadalquivir basin.

ELV values from both basins were checked and very different and disparate limit values, mainly related to nutrients, were obtained. They are shown in Table 1 and Table 2.

As can be seen in Table 1, ELVs required for treatment plants discharging into the Guadiana basin are based on their distance to the main water body, requiring, in many cases, restrictions in nutrients and even in organic matter much higher than those established by European regulations [91/271 / CEE]. However, for the Guadalquivir basin, to which only 10% of urban agglomerations of less than 2,000 IE which are to be equipped with a treatment system release their effluents, only organic matter and solids requirements are imposed. Nevertheless, when the discharge is released into a sensitive area, they must achieve nutrient removal values equal to those imposed to facilities for more than 10,000 IE (Table 2). It should be noted that, in the case of the province of Badajoz, none of their urban agglomerations releases into sensitive areas.

Affluent characterization

Regarding affluent characterization, a total of 87 sampling points have been monitorized, of which 27 correspond to inletsof operating Waste Water Treatment Plants (WWTP) of villages with more than 2000 IE, while the rest correspond to the discharge points of the sewage network of 60 villageswith less than 2000 IE.

Each sampling pointof the 27 WWTPs have provided 4 annual samples quarterly recorded by taking hourly samples throughout 24 hours, while the remaining 60 points from the sewage networks have been half-yearly recorded, again with samples hourly taken throughout a whole day.

The samples have been analyzed according to standard methods defined in the ISO17025 norm [UNE-EN ISO17025: 2017]. Conductivity, pH, COD, BOD5, TSS, TN, TP, NH_4 and oils and fats have been measured.

Table 1. *LEVsprovided by the Guadiana Hydrographic Confederation for small villages releasing effluents to the HPD managed by this institution.*

IE	[1.000-2000)				[600-1000)				[300-600)				[100-300)				< 100			
ELV	Distance to the main water body (Km)				Distance to the main water body (Km)				Distance to the main water body (Km)				Distance to the main water body (Km)				Distance to the main water body (Km)			
	0-1	1-5	5-10	>10	0-1	1-5	5-10	>10	0-1	1-5	5-10	>10	0-1	1-5	5-10	>10	0-1	1-5	5-10	>10
BOD_5 (mg/l)	20	25	25	25	25	25	25	25	25	25	25	25	25	25	25	25	25	30	35	40
COD (mg/l)	100	125	125	125	125	125	125	125	125	125	125	125	125	125	125	125	125	135	145	155

TSS (mg/l)	**25**	**35**	**35**	**35**	**35**	**35**	**35**	**35**	**35**	**35**	**35**	**35**	**35**	**35**	**35**	**35**	**35**	**45**	**55**	**65**
NH_4^+ (mg/l)	**4**	**6**	**8**	**10**	**5**	**7**	**9**	**11**	**6**	**8**	**10**	**12**	**8**	**10**	**12**	**14**	**10**	**12**	**14**	**16**
TN (mg/l)	**15**	**20**	**25**	**30**	**18**	**23**	**28**	**-**	**20**	**25**	**30**	**-**	**25**	**30**	**-**	**-**	**30**	**-**	**-**	**-**
TP – Daily Max. Value (mg/l)	**5**	**6**	**7**	**8**	**5,5**	**6,5**	**7,5**	**-**	**6**	**7**	**8**	**-**	**7**	**8**	**-**	**-**	**8**	**-**	**-**	**-**
TP– Daily Mean Value (mg/l)	**3**	**4**	**5**	**6**	**3,5**	**4,5**	**5,5**	**-**	**4**	**5**	**6**	**-**	**5**	**6**	**-**	**-**	**6**	**-**	**-**	**-**
TP- Yearly Mean Value (mg/l)	**1,5**	**2**	**3**	**4**	**2**	**2,5**	**3,5**	**-**	**2,5**	**3**	**4**	**-**	**3,5**	**4**	**-**	**-**	**4**	**-**	**-**	**-**

**BOD_5 (Biological Oxygen Demand, 5 days); COD (Chemical Oxygen Demand), TSS (Total Suspended Solids); NH_4^+ (Ammonium); TN (Total Nitrogen), TP (Total Phosphorous).*

Table 2. *LEVs provided by the Guadiana Hydrographic Confederation for small villages releasing effluents to the HPD managed by this institution.*

LEV	Concentration
BOD_5 (mg/l)	**20**
COD (mg/l)	**100**
TSS (mg/l)	**25**
If the discharge is released to a SENSITIVE AREA	
TN (mg/l)	15
TP (mg/l)	**2**

RESULTS AND DISCUSSION

As shown in Fig. 1, the mean value of nutrients in small villages is much higher than in large ones, being 50 % higher in ammonium concentration and practically 45% for phosphorous. This is so because, in small villages, agricultural and livestock farms have a large weight on waste waters, in which, in the dry season, nitrogen values above 100 mg/l may be recorded.

Thus, the fact that regulatory authorities impose on effluents from those small villages quality criteria with threshold values even lower than those required by regulation 91/271/CEE for populations of more than 10,000 IEdischargingwaste waters into sensitive areas, demand from the treatment technologies to be implemented higher technological performances than those associated with an adequate removal of organic matter, what causes management and service costs which cannot be assumed by small municipalities.

The most widespread technologies used to deal with water treatment problems in small villages are Rotating Biological Contactors and Artificial Wetlands. Thus, based on the results shown in Figure 1, and taking into account the design criteria of these technologies, imposing the aforementioned treatment removal performances to the design criteria of both technologies will require a surface to support the biomass 40-60% higher than that required by conventional technologies, which generates a significant increase in implementation cost of 30-50% and 25-40%, respectively,for management.

In this way, reaching the established ELVs and ensuring the sustainability of the treatment service in small villagesof the province of Badajoz discharginginto the Guadiana basin, will demand an estimated extra cost of 40% when compared with thosedischarginginto the Guadalquivir basin. In addition, these last ones only represent 10% of villages with less than 2000 IEto be equipped with treatmentplants. Therefore, the management models for thosefacilitieshave tode designed to deal with these extra cost ensuring that the service will not decrease its performanceand without generating a deficit that could jeopardize the future of water treatment in small villages.

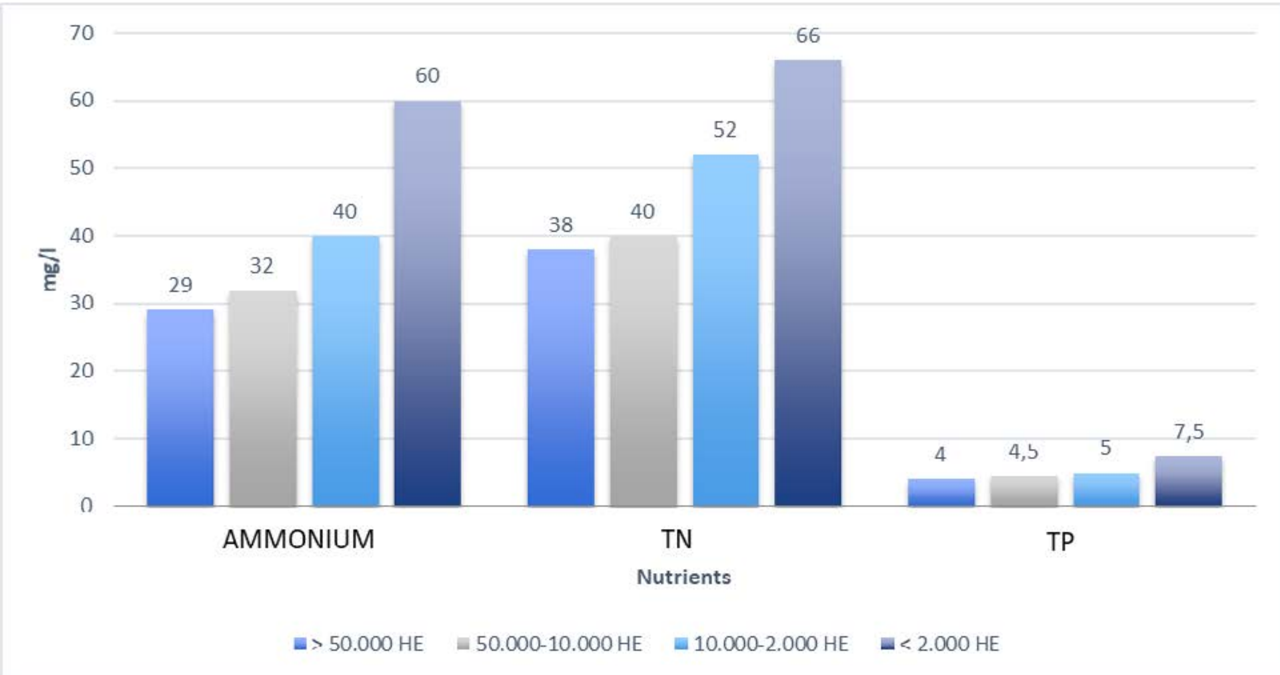

Figure 1. *Characterization of affluent waste water from villages in the province of Badajoz related to IE. These data were recorded in 2villages> 50.000 IE; 5 villages with 10.000-50.000 IE; 20 villages with 2.000 a 10.000 IE and 60 villages with< 2.000 IE. They are yearly mean values, hourly recorded throughout 24 hours.*

CONCLUSIONS

The modification of the Penal Code has promoted the need to provide treatment systems to small villages.Nevertheless the imposition of stricter ELVs related to nutrients will lead to significant increases inimplementation, operation and maintenance costsin the case of the province of Badajoz. Thus, the risk to the sustainability of those facilities has practically doubled, associated with a management model withoutenough strength to support the treatment process, since classical models of direct management are not feasible for villageswith minimal economic and technicalresources and indirect management models are not attractive for private companies. So the supra-municipal management model based on economy of scale appears as the only solution. Nevertheless, this management model presents some limits, since the economy of scale does not have the capacity to manage all small villagesdemanding their services. Therefore, thiswaste water treatment scenedemands further investigations not only in more efficient treatment technologies but also in more efficient management models which could assume and guarantee the sustainability of the treatment systems.

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MEDITERRANEAN LIVING LABS FOR NON-CONVENTIONAL WATER REUSE AT LOCAL SCALE: MENAWARA PROJECT

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Abstract

Mediterranean countries are torn between old and new water policies, and water shortage experience is not related only to increasing demand and/or climate change, but also to poor infrastructure and management practices. This situation is aggravated in those rural areas where irrigated agriculture represents the backbone of the social growth and driving force of the economic activity.In this sense, the joint challenges of MENAWARA project consist of providing additional resources by recycling non-conventional water (drainage and wastewater), rationalizing water use practices and setting operational governance models in line with national and international plans. The project is designed to enhance access to water through the treatment of wastewater to be re-used as complementary irrigation and to strengthen the capacity of stakeholders, including local farmers. In the specific interventions in Spain, Italy, Tunisia, Palestine and Jordan, the actions are foreseen to turn into open living

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labs (for the co-creation, experimentation and evaluation of innovative products in the non-conventional water treatment and reuse sector); whereat small-scale innovative and technological solutions to encourage the end-use application of treated wastewater for agricultural irrigation and increase water efficiency and availability are being implemented, strengthening the promotion of synergies and the transfer of knowledge and innovation between the different stakeholders involved in the water governance.

INTRODUCTION

The Mediterranean region is considered as one of the world's most water-stressed areas where some countries have less than 1000 m^3 /capita/year. A number of reasons are behind this situation, which includes, but are no restricted to, the relatively uneven distribution of precipitation, high temperatures, or increased demands of the population, and climate change is expected to aggravate the situation even more. More than 70 % of total water withdrawals are allocated to irrigated agriculture and water losses and leaks during conveyance and distribution, combined with inefficiency and waste in both irrigation and domestic use, are estimated at 45 % of total water demand for these two sectors.

Additionally, Mediterranean countries are torn between old and new water policies, and specifically, the agriculture sector has to adapt to the new political and institutional framework, both at the national and international level, where the synergy between Agricultural Policies together with Environmental Policies and Conservation of natural resources is taking place. More specifically, the above-mentioned situation is aggravated in those rural areas where irrigated agriculture represents the backbone of the social growth and driving force of the economic activity.

Under the above context, the use of non-conventional water, as drainage and wastewater, is one of the most sustainable alternatives to cope with water shortage. It would have a number of advantages that include closing the gap between supply and demand, stopping the pollution of freshwater resources, providing sound solutions to water scarcity and climate change, and helping to achieve Millennium Development Goals.

The EU funded project "MENAWARA – Non Conventional Water Re-use in Agriculture in Mediterranean countries" (ENI CBC Med program) is designed to enhance access to water through the treatment of wastewater to be reused as complementary irrigation and to strengthen the operational capacity of stakeholders of the quadruple helix (Cavallini et al., 2016), including local farmers. In the specific interventions in Spain, Italy, Tunisia, Palestine, and Jordan, the actions are foreseen to turn into open living labs; whereat small-scale innovative and technological solutions to encourage the end-use application of treated wastewater for agricultural irrigation and increase water efficiency and availability are being implemented, strengthening the promotion of synergies and the transfer of knowledge and innovation between the different stakeholders involved in the water governance.

The actions developed in the MENAWARA project will reduce the stress on freshwater sources from the agriculture sector and will improve the quality of treated wastewater in agriculture. Clean and environmentally friendly technological, managerial, and operational innovation will be applied and results shared among relevant stakeholders. Also, it will play an important role in reducing water insecurity by designing the most suitable post-treatment and MAR systems for each intervention area and by promoting sustainable development in rural areas.

LIVING LABS´S INTERVENTION SITES

Living labs are defined as "user-centered, open innovation ecosystems based on a systematic user co-creation approach in public-private-people partnerships, integrating research and innovation processes in real-life communities and settings" (Evans et al., 2017), (Water Europe, 2019).

Six intervention sites of MENAWARA project, for treating less than 2,000 m^3/d each one, in Tunisia, Palestine, Jordan, Spain and Italy, are foreseen to turn in open living labs, a peer-to-peer learning space where youths, technicians, water users' associations, local farmers and local authorities will be trained on capitalizing on innovative and user-oriented wastewater treatment, reuse and irrigation technologies. The engagement of stakeholders, based on a model of the quadruple helix will facilitate knowledge transfer regarding sustainable use of water resource and circular economy, fostering the dialogue, and developing national planning more responsive to the community's needs. Finally, MENAWARA project is expected to provide a "field lab" to develop, test, and validate a combination of solutions for sustainable wastewater treatment and reuse.

RESULTS AND DISCUSSION

In Tunisia, two living labs are foreseen at the wastewater treatment plants (WWTP) of Choutrana II (extended aeration + secondary decanter + post-treatment train based on pressurised sand filtration followed by UV disinfection) and Borj Touil (2 horizontal flow constructed wetlands followed by maturation pond). Reclaimed water will be used for the irrigation of local plots.

In Palestine, the living lab is foreseen at the WWTP Beit Dajan, where aninnovative pre-treatment compact system will be implemented. For water reclamationis foreseen a post-treatment train based on a filtration process using pressure sand filters and a subsequent disinfection stage by application of hypochlorite as a disinfecting agent.Reclaimed water will be used for the agricultural irrigation in an experimental plot. (*Figure 3*).

In Jordan, the living lab is foreseen at the Ramtha Research Station,the reclamation train will be based on sand filtration followed by UV disinfection. Reclaimed water will be used for the irrigation of different crops in experimental plots.

In Spain, the living lab is foreseen at the Carrión de los Céspedes experimental center. The treatment train will be based on different kind and configurations of constructed wetlands (aerated, floating macrophytes, vertical flow, superficial flow), followed by a storage pond with ultrasound treatment and pressurised sand filtration. The reclaimed water will be used for the irrigation of a local olive grove plot.

In Italy, the Managed Aquifer Recharge (MAR) technique based on Forested Infiltration Areas (FIA) will be tested as a best practice to mitigate the groundwater nitrate contamination for the sandy phreatic aquifer (SHU) in the NVZ of Arborea (central-western Sardinia, Italy). The FIA system will be implemented in an area of around 0.4 ha and supplied with non-conventional water (drainage water), pumped from an existing dewatering pumping station. It will consist of six parallel recharge trenches placed between rows of white poplar trees (*Populus alba*) and equipped with an innovative Passive Treatment System, consisting of a mixture of inert and organic materials to attenuate organic and inorganic contamination and to prevent clogging processes at the infiltrating surface.

Figure 1. *Overview of the location of (6) living labs: (A) WWTP Choutrana II- Tunisia, (B) WWTP Borj Touil- Tunisia, (C) WWTP Beit Dajan- Palestine, (D) Ramtha research Sattion- Jordan, (E) Experimental Center of Carrión de los Céspedes- Spain, (F) Arborea intervention site- Italy.*

Target groups of the living labs will be farmer households living in the different intervention areas using TWW to irrigate olives trees, fodders and ornamental and fruits plants, technicians from local institutions, relevant local and national authorities involved in inter --/regional roundtables and very important, the women, considered the most vulnerable in fragile agricultural systems. In the frame of MENAWARA project, building women resilience through the proper reuse of better quality treated wastewater is one of the challenging objectives.

CONCLUSIONS

The actions developed in the proposed living labs for non-conventional water reuse at small scale will reduce the stress on freshwater sources from agriculture sector and will improve the quality of treated wastewater in agriculture. Clean and environmental friendly technological, managerial and operational innovation will be applied and results shared among relevant stakeholders. Also, it will play an important role in reducing water insecurity by designing the most suitable post-treatment and MAR systems for each intervention area and by promoting a sustainable development in rural areas.

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EC ENVIRONMENTAL TECHNOLOGY VERIFICATION OF RICHWATER® MEMBRANE BIOREACTOR

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Abstract

The implementation of RichWater® Membrane Bioreactor (MBR) for urban wastewater reuse in agriculture irrigation and fertilization will support to reduce pressure in European regions water sources suffering from droughts and water scarcity. In addition, it will support the transition to more sustainable economy as the use of reclaimed water for agriculture fertigation is a powerful climate change adaptation measure and truly aligned with the principles of Circular Economy. RichWater ® contributes not only to water circularity, but also to close nutrient cycle by the valorisation of nutrients embedded in the wastewater. In order to overcome current barriers for the implementation of RichWater® MBR, it has been verified by the European Commission **Environmental Technology Verification (ETV)**, which is a tool that helps innovative technologies to reach the market. The results of the verification process show the performance efficiency of RichWater® MBR in treating and reclaiming urban wastewater to deliver high quality effluent for its reuse in agriculture fulfilling the legal framework. The ETV allows stakeholders such as farmers, public administration to access to verified data which probe the RichWater® MBR claims. RichWater® MBR is one of the 16 verified technologies in the field of Water Treatment & Monitoring.

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INTRODUCTION

Water scarcity is an increasingly frequent phenomenon. Concerns on water efficiency are therefore getting importance in Europe in the last years as reflected in relevant strategic documents such as the new Circular Economy Action Plan. In southern European countries such as Greece, Italy, Portugal, and Spain more than 70% of water consumption goes to agriculture irrigation. Therefore, measures to increase water efficiency in agriculture will have a major impact to cope with water scarcity.

In this context, water reuse has a strong potential to provide an alternative source of water for irrigation as well as for other purposes such as street cleaning, garden irrigation or other industrial uses. Indeed, Water reuse is commonly and successfully practiced in several EU Member States, as well as in, for example, Israel, California, Australia, and Singapore. However, this practice is so far deployed below its potential in the EU.

RichWater® project has demonstrated a wastewater treatment and reclamation technology based on a Membrane Bioreactor (MBR) which has been designed to reuse the effluent in agricultural irrigation. As one of the main barriers for reusing water is public perception, the project has followed a process of Environmental Technology Verification (ETV) in order to obtain an official certificate on the treatment performance. Moreover, one of the innovations of RichWater® is the specific design to preserve and optimize nutrients in the effluent. Thus, ETV verification also validated the nutrient content in the effluent in order to estimate the additional benefits for farmers resulted from reducing the need of using fertilizers for irrigation.

METHODS

RichWater® MBR is the result of the work done by BIOAZUL SL within the RichWater H2020 project and has been verified under ETV. The ETV concept is to **offer a verification procedure to cutting edge environmental technologies that may otherwise find it difficult to establish their environmental added value**. The verification procedure allows for an independent assessment and validation of the manufacturer's claims on the performance and environmental benefits of their technology. The information produced by the verification is public and can be used to compare performance parameters and therefore becomes an extremely useful tool to convince relevant stakeholders of the merits of a technology, potentially enhancing its market value and acceptance.

The information provided, in the form of a **Statement of Verification**, gives the possibility for direct and objective comparison between different technologies reducing the risk on adopting new technologies and encouraging informed and sound investments. ETV results could be used to prove compliance with any relevant legislation, to underpin a bid in public tendering, to convince investors or customers of the reliability of performance claims and to avoid having to repeat demonstrations for different users.

A Verified Technology has been through a Verification Process. This means that a certified entity called Verification Body, has reviewed all the data that supports the technology's performance. The Verification Body has checked that the data is consistent, of good quality and that it covers the important aspects of the technology's performance. After completing the Verification Process, a Statement of Verification is issued and published in the EU ETV website (https://ec.europa.eu/environment/ecoap/etv_en).

RESULTS AND DISCUSSION

Richwater@ MBR is located in an operating municipal wastewater treatment plant (WWTP) located in the municipality of Algarrobo within Malaga province (Spain) which treats wastewater from the population of ca. 6,500 inhabitants. The plant is managed by AXARAGUA, a private--public partnership created by the "Mancomunidad de Municipios de la Axarquía".

Figure 1. *Algarrobo WWTP and RichWater ® MBR location (google maps link: https://goo.gl/JUnQdb)*

RichWater® system is conceived as a single solution for wastewater treatment and reclamation of urban wastewater for its use in agriculture, combining irrigation and fertilization purposes. The overall system allows to treat 150m^3/day and produce high quality effluent (reclaimed water) meeting the regulatory standards for irrigation of crops, to be consumed by humans while preserving the content of nutrients relevant for the fertilization effect. The core part of the RichWater® system is the Membrane Bioreactor (MBR). It is a compact system for decentralized urban wastewater treatment based on membrane technology. The design allows to produce high quality effluent that meets the requirements for reclaimed water used for irrigation of crops for human consumption while maintaining high content level of nutrients. It allows to achieve an optimal, simultaneous irrigation and fertilization effect.

The sampling campaign for the ETV started on the 24th of September 2018 and finalized the 11th of December 2018. A total of 16 samples of the influent entering to RichWater® MBR and 16 samples of the RichWater® MBR effluent were taken. The total duration of each composite sampling was 24h. Grab samples of influent and effluent for microbiological analyses were collected on Tuesdays and Thursdays. These samples were taken within 2 hours after the autosampler completed the 24-hour sampling. Grab samples of the mixed liquor for MLSS, MLVS analysis were taken from the aeration and membrane tanks on the same days as the samples for microbiological analyses i.e. every Tuesday and Thursday, also within 2 hours after the autosampler completed the 24-hour sampling.

In order to carry out the RichWater ® MBR ETV and verify that the technology produces effluent rich in nutrients for irrigation of crops to be consumed by humans, performance parameters and the given limits were assessed, as well as operational parameters as shown in table 1

The quality of the reclaimed wastewater (effluent) and operational parameters in relation to the performance claims for verification and legal requirements (if applicable) are presented below. The declared values were given at the beginning of the verification and the results obtained during the tests performed for assessing the *RichWater® MBR.*

Table 2: *Performance and operational parameters compliance*

Performance parameters	
Declared Values	**ETV results**
Nitrates ≥ **50** mg/l	The average value was **92,31** mg/l, **above the declared level. Effluent quality better than declared**.
Phosphorus ≥ **1.5** mg P/l	The average value was **4.34** mg P/l, **above the declared level. Effluent quality better than declared**.
Potassium ≥ **15** mg/l	The average value was **14,43** mg/l **Slightly below the declared level**.
BOD_5 ≤ **25** mg O_2/l	The average value was **<15** mg O_2/l **Effluent quality better than declared.**
COD ≤ **125** mg O_2/l	The average value was **23,67** mg O_2/l **Effluent quality better than declared**.
Suspended solids ≤ **20** mg/l	Concentration below detection limit for all effluent samples (<**8** mg/l) **Effluent quality better than declared**.
Turbidity ≤ **10** NTU	The average value was **0.69** NTU **Effluent quality better than declared**.
Escherichia Coli max **50** CFU/100 mL	In **94%** of effluent samples, the concentration was **below 50** CFU/100 mL. Only on 09.10.2018 Escherichia Coli content was 90 CFU/100 mL but it was below the maximum admitted values of this parameter defined for TWW use on tree crops at RD1620/2007 (100 CFU/100 mL).

Legionella spp. **max 1000** CFU/L	**94%** of effluent samples did not contain Legionella. Only on 16.10.2018 Legionella content was 1200 CFU/L. The maximum admitted value of this parameter is 1000 CFU/L for at least 90% samples (RD1620/2007). The sample that exceeds the maximum admitted value did not exceed the maximum deviation limit of 1 logarithmic unit.
Nematodes max **1** egg/10 L	**100%** of effluent samples did not contain Nematodes.
Operational parameters	
Declared Values	**ETV results**
Wastewater flow in the range of **0-10** m^3/h	The wastewater flow in the range of **0.41-4.85** m^3/h,
HRT in the biological reactor in the range of **30-10** h	HRT in the biological reactor in the range of **9.86- 115.90** h
MLSS concentration in the biological reactor **≤ 6** kg/m^3	The average concentration of MLSS in the biological reactor **3.476** kg/m^3
OLR **< 0.5** $kgBOD_5/kg \cdot d$	The average value of the parameter **0.08** $kgBOD_5/kg \cdot d$
DO concentration in the biological reactor in the range of **1.5-2.0** mg/l	DO concentration in the range of **0.05- 2.4** mg/l (**DO concentration different than declared**)

RichWater ® MBR allows the production of high quality reclaimed water rich in the most important plants macronutrients- Nitrogen, Phosphorus and Potassium as shown in table 2. Therefore, the reclaimed water has a double effect: irrigation and fertilization. The used of this nutrient rich reclaimed water allows the total or partial replacement of chemical fertilizers from non-renewable resources.

Table 2: Summary of RichWater ® MBR removal performance

	BOD_5 (mg O_2/l)	COD (mg O_2/l)	TS (mg/l)	TN (mg N/l)	Nitrates (mg N/l)	TP (mg/l)	TK (mg/l)	Turbidity (NTU)
Influent								
Min	93,00	239,00	106,00	25,00	<5	3,30	8,80	2,60
Max	232,00	646,00	397,00	67,00	<5	12,00	25,00	349,00
Average	163,25	408,31	210,06	38,81	<5	5,97	14,81	120,48
Efluent								
Min	<15	15,00	< 8	16,00	5,20	0,16	8,10	0,16
Max	<15	32,00	< 8	45,00	40,68	12,00	24,00	2,30
Average	<15	23,67	< 8	26,00	20,85	4,34	14,43	0,69

CONCLUSIONS

ETV verification has recognized the potential of RichWater ® to provide an effluent to be used in irrigation which is safe and optimize the amount of nutrients for the plants.

ETV verification conducted by an independent body helps innovators with the commercialization by providing confidence to the potential clients and guarantees that performance data is credible and accurate. These guarantees are particularly relevant in the context of water reuse since negative public perception has been recognized as one of the main barriers for the use of reclaimed water in agriculture.

The recent approval of EU Regulation 2020/741 implies the application of more stringent standards in terms of water quality for reuse in agriculture. This scenario implies further investments in wastewater treatment and reclamation in order to comply with the new legal requirements. In this sense, membrane technologies as RichWater ® have proven to be a reliable solution to extend water reuse.

Finally, RichWater ® has proven its potential to fertigate and provide nutrients to the plant. This is one of the main benefits of using reclaimed water for irrigation. Nutrients contained in the reclaimed water are directly assimilated by the plants and therefore can partially replace conventional chemical fertilizers. This implies economic savings to farmers and reduce the dependence on chemical fertilizers.

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EVALUATION OF EFFECTIVE MICROORGANISMS (EM) IN THE PROCESS OF TREATMENT OF DOMESTIC WASTEWATER IN HIGH-ANDEAN CONDITIONS

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Abstract

The research had the objective of evaluating the effect of effective microorganisms (EM) in the domestic wastewater treatment process in high Andean conditions. 4 treatments of 110 L water capacity were designed, the first two contain Efficient Microorganisms (220 mL) applied in aerobic and anaerobic systems; and the other two were witnesses; Likewise, plastic solar collectors were incorporated to increase the temperature of the treatments with an operating time of 10 hours during 22 days. The monitored parameters were T°, pH, EC, DO, BOD_5, COD, TSS, $N\text{-}NH_4^+$, $N\text{-}NO_2^-$, $N\text{-}NO_3^-$, P-Total and turbidity. This wastewater under study has a BOD_5/COD ratio that is not very degradable and the BOD, N and P relationship is not optimal for the development of biological processes. The results show that the removal in the first treatment were BOD_5 (80.7%), COD (79.9%), TSS (88.9%) and P-total (81.8%) and the second treatment presents better results for the removal of nitrogen $N\text{-}NH_4^-$ (100%), $N\text{-}NO_2^-$ (100%) and $N\text{-}NO_3^-$ (98.1%). Therefore, EM treatments show better results in reducing the monitored parameters.

INTRODUCTION

The United Nations World Program for the Assessment of Water Resources states that in the world more than 80% of wastewater is discharged into the environment without treatment (UNESCO, 2020). In 2012, eight hundred thousand people died from drinking contaminated water. Likewise, inadequate sanitation services, limited hand washing facilities and the spillage of residual water, increase the dead and deoxygenated zones in the seas and oceans, affecting the ecosystem in an area of 245000 km^2, in addition, it generates negative impacts on industrial and artisanal fisheries, as well as the food chain (WWAP, 2017).

On the other hand, municipal and industrial wastewater treatment is 70, 38, 28 and 8% in high-income, upper-middle-income, lower-middle-income, low-income countries, respectively (UNESCO, 2020). Efficient microorganisms consist of liquid formulated products that contain more than 80 species of microorganisms, some species are aerobic, anaerobic and even photosynthetic species whose main achievement is that they can coexist as microbial communities and can even be completed(Hoyos et al., 2008)

The study on the treatment of domestic wastewater, allowed to propose alternative solutions for the treatment of wastewater using effective microorganisms in high Andean conditions (3850 meters above sea level, atmospheric pressure of 0.634 atmospheres and temperatures <20 °C), using a heating system solar with low-cost flat collectors that can be replicated to existing treatment systems in order to optimize or generate new wastewater treatment projects to comply with the Peruvian regulatory framework on wastewater treatment, the objective of the study was to evaluate the effect of effective microorganisms (EM) in the domestic wastewater treatment process in high Andean conditions.

METHODS

The research was carried out in Villa Chullunquiani, located on the Arequipa Km 6 highway in the Juliaca district, San Román province in the Puno - Peru region, at an altitude of 3850 meters above sea level (mbsl). The treatments are bach type reactors with a capacity of 115 L in volume, the average determined atmospheric pressure was 0.63 atmosphere and average daily ambient temperature of 10.11 ° C and the average daily temperature during the hours of sun and night was 13.36 and 6.57 ° C , respectively, the minimum and maximum peaks during the experimental period (22 days) were 3.22 and 17.32 ° C, which occurred at 5:00 and 13:30 in the months of October and November (data obtained from the meteorological station UPeU, Juliaca).

The monitored parameters (pH, EC, DO and temperature of the treatments) were determined in the field, recorded manually at 12:00 hours for 22 days, and the monitored laboratory analyzes were BOD_5, COD, TSS, N-NH_4^+, N-NO_2^-, N-NO_3^-, P-Total and Turbidity, the analyzes and equipment used were from the UPeU Professional School of Environmental Engineering laboratory - Juliaca campus.

Table 1. *Métodos de análisis de laboratorio de los parámetros monitoreados*

Parameter	Laboratoryanalysismethod
Biochemical Oxygen Demand	Electrometric method. NTP 214.037:2015.
Determination of Chemical Oxygen Demand	Colorimetric method. NTP.360.501:2016.
Determination of total suspended solids..	Gravimetric method. NTP. 214.039:2015.
Ammonia nitrogen	Colorimetric method
Nitrite	Colorimetric method
Nitrate	Colorimetric method
Total phosphorus	Colorimetric method
Determination of turbidity	Nephelometric method. NTP 214.006 1999.

Four solar heating treatments were carried out with flat plastic collectors with an operating time of 10 hours from 7:00 a.m. to 5:00 p.m. in 22 days and effective microorganisms (EM) were applied for the first two treatments. Treatment 1: Aerobic with EM + 220 mL EM for 110 L applied on a sponge; Treatment 2: Anaerobic with EM +220 mL EM for 110 L, applied on a sponge; Treatment 3: Anaerobic without EM, and Treatment 4: Aerobic without EM. Figure 1 shows the methodological design.

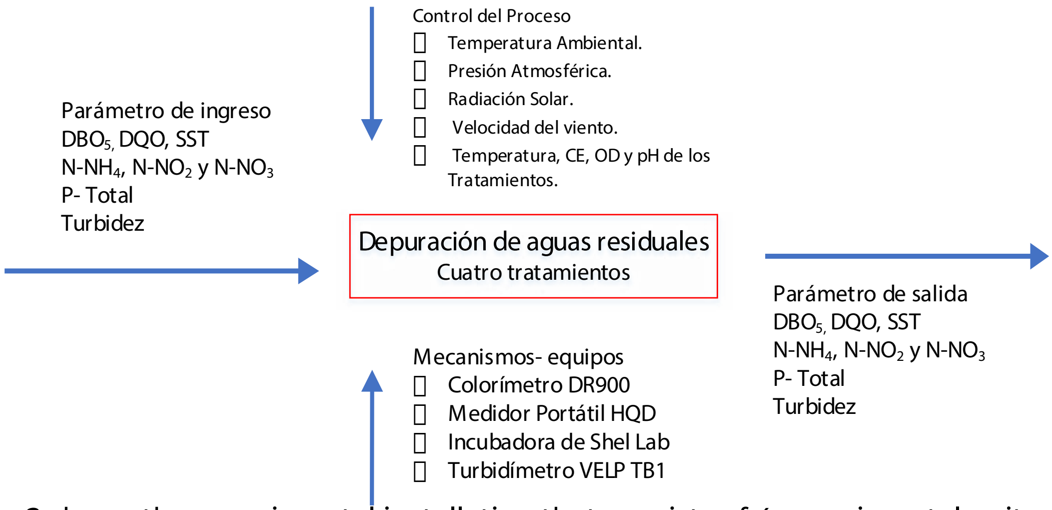
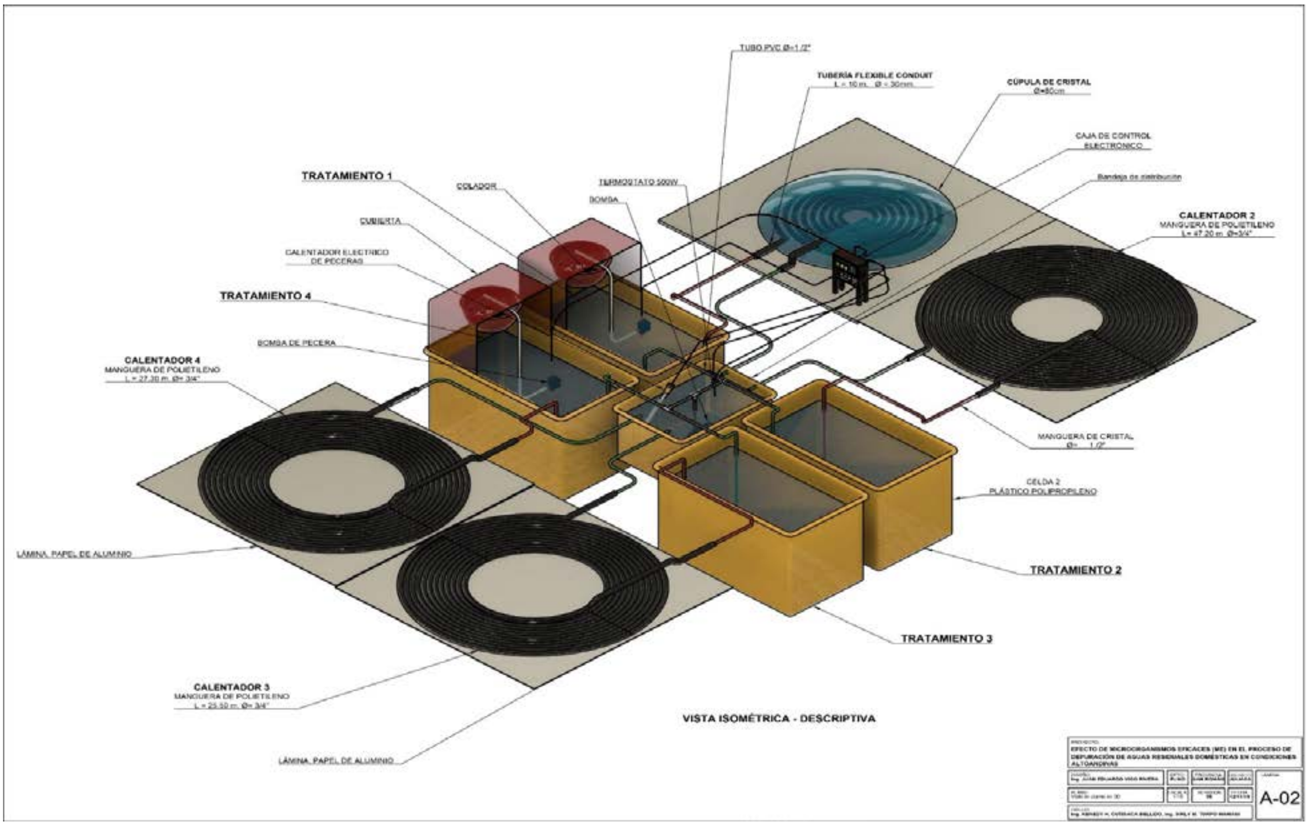


Figure 2 shows the experimental installation that consists of 4 experimental units with a capacity of 120L (2 with aeration system) that act as bach-type complete mixing reactors and are distributed hourly to which are called treatments, around them there are 4 plastic or so-called “low-cost” solar collectors, the first collector has a cover or glass dome to protect it from the cooling effects of the wind and the remaining three do not.



RESULTS AND DISCUSSION

Table 2 shows the initial results of wastewater when treating domestic sources.

Table 2. *Results of the initial values of the monitored parameters of domestic wastewater*

Parameters											
In situ				Laboratory							
Temp.	pH	EC	DO	Turbidity	COD	BOD_5	TSS	N-NH4	N-NO2	N-NO3	P -Total
°C	-	µS/cm	mg/L	NTU	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
22	7.15	1330	0.11	159	434.8	182	90	36	0.23	10.4	56.7

Table 2 shows that the initial DO was 0.11 mg/L; the pH value is 7.15 (medium level) and the EC of 1340 µS/cm (high level) which are typical values of urban wastewater according to Henze (1992). The BOD_5(182 mg/L) and COD (434.8 mg/L) correspond to a medium concentration and the TSS (90 mg/L) to a low concentration according to Metcalf and Eddy (1995); N-NH4 (36 mg/L) correspond to mean values of urban wastewater according to Sánchez-Ramírez et al. (2017) and

$N\text{-}NO_3^-$ (10.4 mg/L) + $N\text{-}NO_2^-$ (0.23 mg/L) and P-Total (56.7 mg/L) correspond to municipal wastewater with high nutrient content according to Henze and Comeau (2011). In addition, theturbidity has a value of 159 NTU as a reference data forthetransparency of thewater.

The BOD_5: N: P ratio is 100: 26: 31, which according to the Manual of operation and maintenance of wastewater treatment plants with the activated sludge process Marín and Osés (2013) is a non-optimal ratio for the development of biological processes.

With the values of the parameters in table 2, the ratios shown in table 3 were determined.

Table 3. *Parameter ratios*

Ratios	Value
BOD_5/COD	0.4
COD/BOD_5	2.4
COD/P-Total	7.67
BOD_5/P-Total	3.21

The BOD_5/COD ratio presents a value of 0.4 which, according to Ardila Arias et al. (2012) is not very biodegradable and according to the Manual of operation and maintenance of wastewater treatment plants with the activated sludge process of Marín and Osés (2013) it has industrial influence (see Table 3).

According to Henze and Comeau (2011), the COD/BOD_5 ratio corresponds to a typical mean municipal wastewater ratio, the relationships is COD/P-Total and BOD/P-Total does not correspond to municipal wastewater relationships. Table 4 shows the results of the parameters determined in the laboratory.

Table 4. *Resultados de los parámetros determinados en laboratorio*

días		Tiempo	BOD $_5$	COD	TSS	N-NH$_4$	N-NO$_2$	N-NO$_3$	P -Total	Turbidity
		mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	UNT	
Tratamientos	1	0	182.00	434.80	90.1	36	0.230	10.4	56.7	159.00
		7	80.00	180.21	41.3	24	0.006	1.5	38.9	35.70
		14	43.60	102.00	21.4	0	0.013	1.1	14.1	19.23
		22	36.60	84.00	10.0	0	0.000	0.4	10.0	5.75
	2	0	182.00	434.80	90.1	36	0.230	10.4	56.7	159.00
		7	102.00	329.00	64.1	26	0.021	3.3	54.9	133.40
		14	58.70	194.00	37.0	7	0.001	0.8	30.5	105.55
		22	44.60	140.25	30.0	0	0.000	0.2	16.7	97.80
	3	0	182.00	434.80	90.1	36	0.230	10.4	56.7	159.00
		7	87.00	265.25	71.0	19	0.011	4.2	55.0	138.20
		14	57.10	179.00	47.0	3	0.010	2.9	31.8	110.50
		22	43.50	129.00	39.0	0	0.000	1.8	21.8	101.00
	4	0	182.00	434.80	90.1	36	0.230	10.4	56.7	159.00
		7	83.50	236.50	51.3	28	0.002	1.8	41.4	59.10
		14	47.20	129.00	29.3	0	0.004	1.5	16.6	27.25
		22	38.20	116.50	19.0	0	0.000	0.6	12.9	20.10

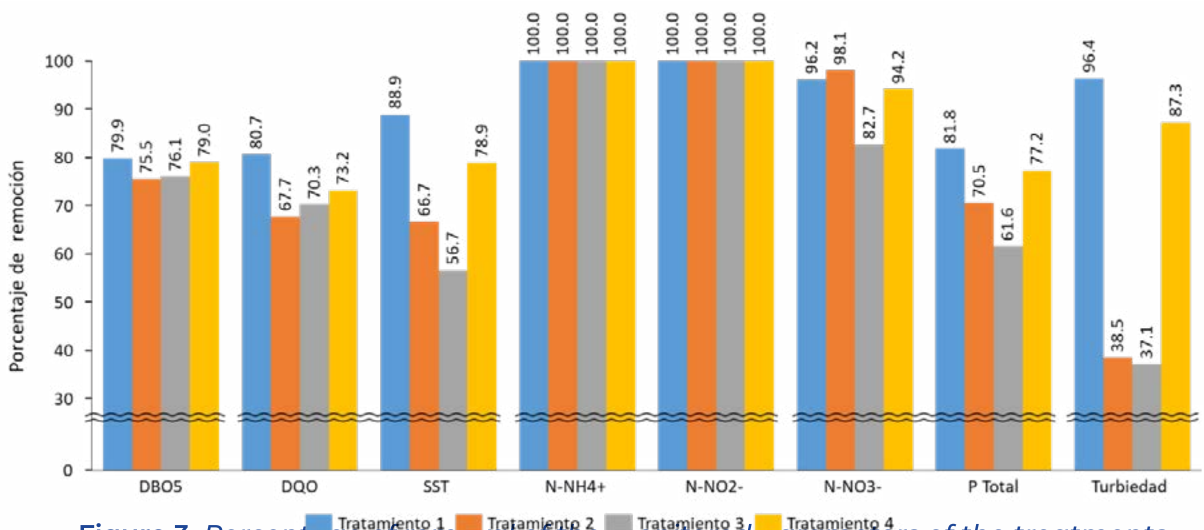
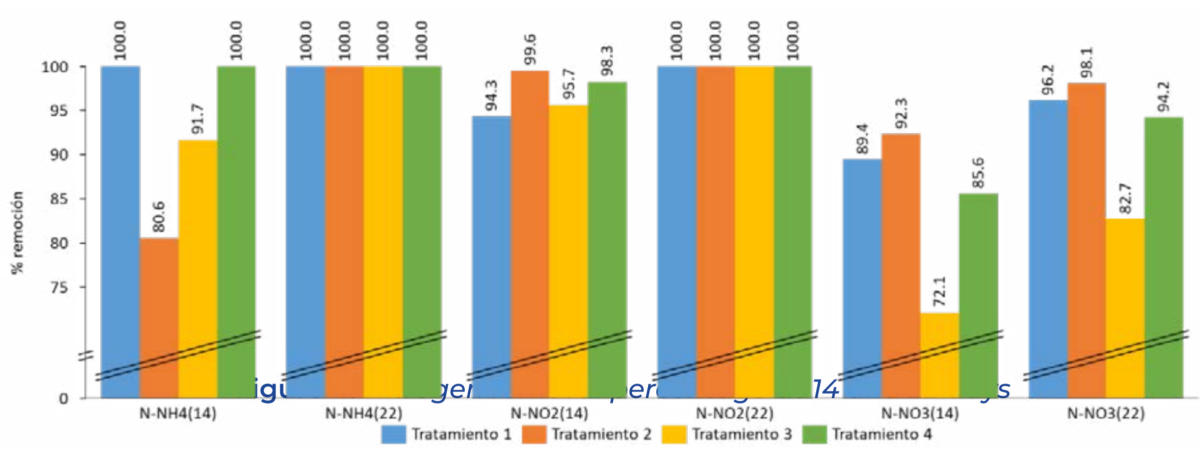


Figure 3. *Percentage of removal of the monitored parameters of the treatments*



Treatment 1 shows better removal results for BOD_5, COD, SST, P-Total and Turbidity (see Fig 3). Treatment 2 shows better results in the removal of nitrogen in the form of nitrite and nitrate (14 days), but all treatments at the end of 22 days remove 100% of ammonia nitrogen and nitrogen in the form of nitrite (see Fig 4). This happens due to the oxidation of ammonia nitrogen to nitrite and then to nitrate.

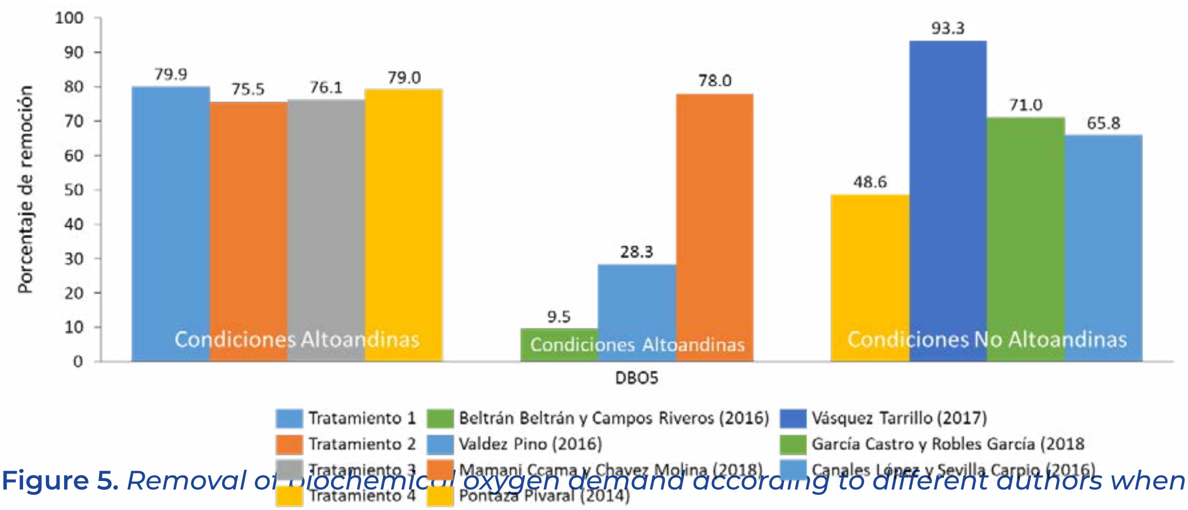


Figure 5. *Removal of biochemical oxygen demand according to different authors when applying Effective Microorganisms to domestic wastewater*

The first, second and third block of grouped columns show the four treatments, investigations in high Andean conditions (≥2500 mbsl) and investigations in non-high Andean conditions, respectively. Beltrán and Campos (2016) had removal of 9.5% of BOD_5 at 3360 meters above sea level at an average environmental temperature of 12 ° C in 90 days; Valdez (2016) had a removal of 28.3% of BOD_5 at 3830 mbsl at ambient temperature that ranged from -5 to 17 ° C in 90 days; Mamani and Chavez (2018) had a 78% removal of BOD_5 at 3850 mbsl at an average environmental temperature of 12 ° C in 15 days. Treatment 1 with ME (mean temperature 22.5 °C) presents better results than the aforementioned investigations under high Andean conditions and treatment 2 with EM (mean temperature of 24.3 ° C) presents better results than the investigations of Beltrán and Valdez (see Fig. 5). Despite being in high Andean conditions, the treatments had energy inputs from low-cost solar collectors that increased the temperature and made a difference compared to the investigations by Beltran, Valdez and Mamani carried out in high Andean conditions.

CONCLUSIONES

Effective microorganisms have a positive effect on the wastewater treatment process in high Andean conditions, as shown in the results of this research, where treatment 1 (Aerobic with EM + 220 mL EM) presents better BOD_5 removal results, COD, SST, total phosphorus and turbidity compared to the other treatments, but treatment 2 (Anaerobic with EM +220 mL EM) presents better results for the removal of nitrogen ($N\text{-}NH_4$, $N\text{-}NO_2$ and $N\text{-}NO_3$).

The treated wastewater presents typical characteristics of domestic origin with high phosphorus content, with a BOD_5/COD ratio that is not very degradable and the COD/P-Total and BOD/P-Total ratios that do not correspond to municipal waters, the BOD, N and P is not optimal for the development of biological processes.

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LIFE PHOENIX: INNOVATIVE COST-EFFECTIVE TREATMENTS FOR REUSING WATER AND NUTRIENTS FOR AGRICULTURAL APPLICATION IN SMALL COMMUNITIES

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Abstract

The possibility of recovering reclaimed water for agriculture and essential nutrients for plants growth from wastewater (WW) in small communities presents an opportunity to foster the development of new technologies. In this context, **LIFE PHOENIX** project develops innovative solutions for urban WW regeneration for agricultural purposes that meets the new European Directive 2020/741, that will come in force in less than 3 years, while eliminating microplastics and contaminants of emerging concern. A novel combination of WW treatment processes using HRAP for microalgae-bacteria cultivation and constructed wetlands has been demonstrated to be a suitable WW treatment able to provide a high quality effluent that can be directly subjected to disinfection for its reuse in

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agriculture. Next, a solar photo-Fenton technology eliminates pathogens and emerging pollutants by the action of solar radiation and AOP. The combination of these technologies presents a novel cost-effective WWT process to produce high quality reclaimed water and recover nutrients in small WWTPs.

Keywords: constructed wetlands; microalgae; photo-Fenton; tertiary treatment; water reuse

INTRODUCTION

The possibility of recovering water and essential nutrients for plants growth from wastewater (WW) presents an opportunity to foster the development of technologies that enable the production of organic fertilisers and help to protect these valuable ressources. When talking about small communities (SC) and rural areas, this topic gains special relevance in order to boost circular economy and recovery of local ressources. In this context, the reuse of wastewater for irrigation results vital, especially when talking about dry and arid regions where water scarcity is present. According to the European Environmental Agency (EEA), around 20 % of the total population of the Mediterranean region live under permanent water stress conditions, and more than half (53 %) of the Mediterranean population is affected by water stress during the summer. In a national context, in Spain the reuse of wastewater is regulated by RD1620/2007, which sets the minimum water quality in terms of contaminants such as pathogen content or turbidity according to its final use. However, although there is no applicable EU standard in force concerning water reuse yet, the *Proposal for a regulation of the European Parliament and of the Council on minimum requirements for water reuse (December, 2019)*, which has been approved by the European Parliament on the 12.05.2020 (European Directive 2020/741), sets the water reuse quality limits in agriculture. This new Directive sets more severe water quality limits than the current national regulations.

In this context, **LIFE PHOENIX** project develops innovative solutions for urban wastewater regeneration for agricultural purposes. The LIFE PHOENIX project's main objective is to obtain reclaimed water in WWTPs that meets the new European Directive 2020/741, while eliminating microplastics (MPs) and contaminants of emerging concern (CECs). The main objectives of the project are:

1. Obtain reclaimed water that meets the strictest requirements of the new European regulations.
2. Minimize the possible environmental and health effects of the use of reclaimed water by reducing toxics, emerging pollutants, antibiotic-resistant bacteria and MPs.
3. Develop a decision support system to ensure the adaptability of tertiary treatment to each specific case.
4. Ensure water quality through online monitoring of certain parameters as pathogens.
5. Recover more than 90% of the nutrients from the wastewater.

6. Test reclaimed water and nutrients in field studies.
7. Reduce the costs of tertiary treatment to 0.10-0.15 €/m^3
8. Study compliance with the new European Directive in existing tertiary plants, and, if necessary, propose upgrading solutions.
9. Promote the replication, transferability and market launch of technologies.
10. Assessment of environmental, social and economic impacts.

To achieve the proposed objectives, two experimental plants will be created, including technologies adapted to the needs of medium-large and small populations, and will be tested in different locations. More than 10 different technologies will be tested throughout the life of the project, with advanced disinfection and oxidation processes as well as pretreatments. The performance of the different possible combinations will be evaluated, to optimize tertiary treatments.

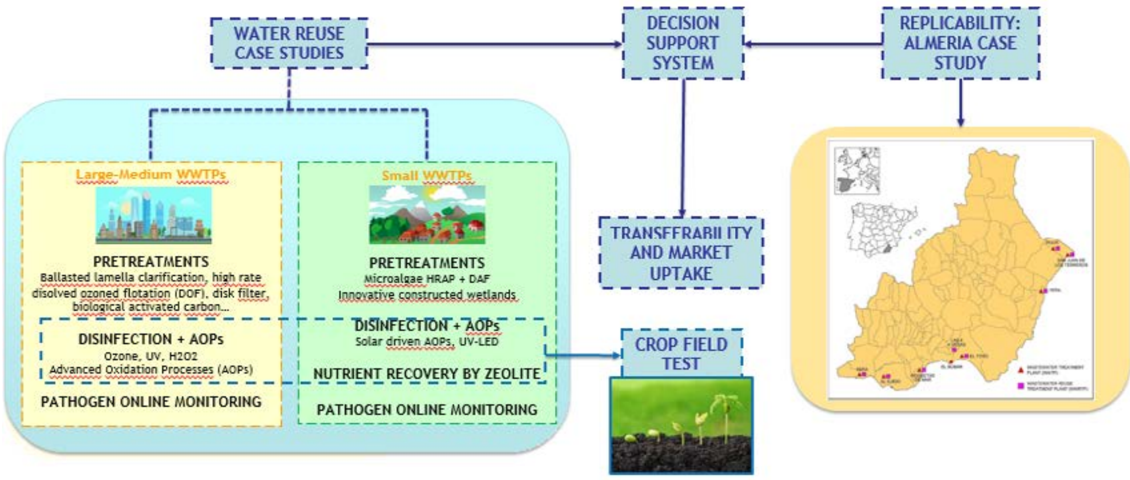

Figure 1. *LIFE PHOENIX diagram*

The aim of this study is to present a novel cost-effective WW treatment process to produce high quality reclaimed water and recover nutrients in small WWTPs.

EXPERIMENTAL PLANT AND TECHNOLOGIES

The experimental plant is located in El Toyo WWTP facility and treats raw wastewater with a system based on extensive technologies. This plant is based on a combination of three different technologies: high rate algae pond (HRAP) for microalgae cultivation; vertical flow constructed wetlands (VFCW) and solar photo-Fenton disinfection plant. The plant will be operated and validated during the next 3 years in the frame of LIFE PHOENIX project. However, previous results from previous similar experiences (H2020 INCOVER, LIFE BIOSOLWARE, LIFE ULISES projects) will be here presented and discussed.

Microalgae raceway and constructed wetlands

A 3,000 m^2 microalgae raceway pond followed by a harvesting process with two flotation units and 200 m^2 vertical wetlands have been operated under real conditions during one full year. The plant presents a full WW treatment with natural based solutions that provide an alternative treatment configuration for small WWTPs based on extensive technologies. The main process steps are:

- **Pretreatment:** a 1 mm sieve avoids the entrance of big solid particles into the biological system.
- **High Rate Algae Pond**: the raceway bioreactor has an area of 3000 m^2 and is able to treat up to 250 m^3 of WW per day (1.500 PE). Two alternative mixing systems are installed and compared: a conventional paddlewheel and a LEAR® (Low Energy Algae Reactor), a low energy mixing system patented by Aqualia, which is based in a submerged axial propeller.
- **Harvesting:** from the raceway the brew is pumped to a dissolved air flotation unit (DAFAST) to separate algae biomass from treated wastewater.
- **Vertical constructed wetlands:** the liquid effluent obtained after harvesting is fed into four 50 m^2 VFCWs (Figure 2), which were proposed as a polishing treatment to nitrify and reduce turbidity, TSS organic matter and pathogens. Two different plant species were planted in the bed: *Arundo donax* and *Phragmithes australis*; to be tested and compared in alternative units.
- **Dewatering:** harvested microalgae biomass is dewatered in a decanter centrifugation unit.

Figure 2. *Panoramic view of El Toyo WWTP with the experimental demo pant (raceway and wetlands)*

Photo-Fenton solar disinfection

New technologies available for WW regeneration include advanced oxidation processes (AOP). In addition, those processes that are able to use sunlight as a source of radiation, are especially interesting. Among these solar treatments stands out the photo-Fenton process. Its efficiency is explained by the generation of hydroxyl radicals ($OH^{\cdot}$), which has a high oxidation potential. The mechanism is based on the reactions shown in Figure 3, where Fe^{2+} is oxidized to Fe^{3+} by reaction with hydrogen peroxide, generating OH and shrinking back to Fe^{2+} by UV-Vis radiation action, generating another radical. Together, a redox cycle is established between Fe^{2+} and Radiation-activated $Fe^{3+,}$ consuming hydrogen peroxide and generating hydroxyl radicals.

In this sense, the use of solar radiation allows to reduce operating costs in an environmentally sustainable way.

The results of research on the photo-Fenton solar process, in terms of both disinfection and removal of emerging contaminants, are promising. In addition, it is worth mentioning that the global interest in research in applications of the photo-Fenton process is steadily increasing, surpassing forty annual publications in prestigious scientific journals, over the past three years. Therefore, interest in this process has increased in recent years, presenting itself as an alternative to conventional treatments (UVC radiation, ozonation and chlorination, mainly). One of the main factors that research focuses on is the design of a reactor that is effective in capturing solar UVA light and certain amounts of visible radiation (up to 600 nm), as well as being simple and economical to build. In this sense, low-cost raceway pond reactor (RPR) operated in continuous mode have been proposed as an alternative to conventional tubular reactors with composite cylinder-parabolic (CPC) solar collectors operated in batch mode. The RPR reactors consist of open channels through which water flows driven by a paddle wheel agitator (Figure 3).

Figure 3. *Solar photo-Fenton RPR tertiary treatment plant at demonstrative scale (LIFE ULISES project, grant agreement no. LIFE18 ENV/ES/000165). S cheme of the redox cycle of the solar photo-Fenton process.*

PRELIMINARY RESULTS

Microalgae system

Performance of the HRAP was studied by monitoring the inlet (pretreated WW) and the outlet of the DAFAST (clarified water). Next plot (Figure 4) shows the annual evolution of removal efficiencies (COD, TN, TP and TSS) during one full year continuous operation:

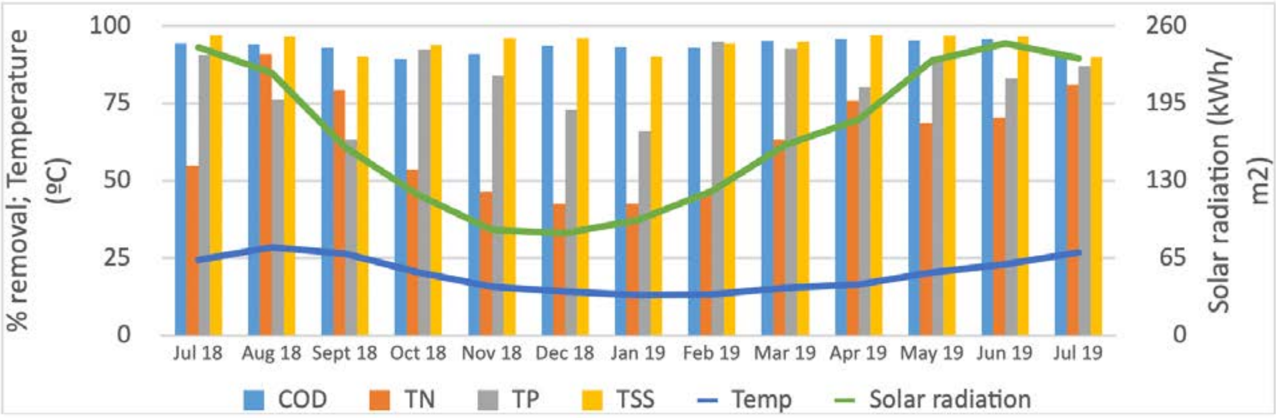

Figure 4. *HRAP monthly removal efficiencies for COD, tN, tP and TSS; ambient temperature and solar radiation*

Concerning other monitored water quality parameters, turbidity was reduced almost 98% respect to the pretreated wastewater and pathogens over 99% removal of both *Total coliforms* and *E. coli* during the microalgae process, reaching 3 and 2 logs removal respectively. If wastewater discharge to the media is considered, the effluent from DAFAST meets the current regulation according to the European Directive 91/271/EEC.

Regarding the harvesting step, solids recovery efficiency in the DAFAST ascends over 97%, reaching a high concentration factor (x48). Harvested biomass in the DAFAST system contains 44 g TS/L, what suggests a concentration efficiency considerably higher than settling processes. This microalgae biomass offers the opportunity to be used as biofertilizer with high nitrogen and phosphorus content. Introducing this concept to the whole WWT process would boost circular economy concept by returning nutrients contained in WW back to the land and to the food chain.

Wetlands system

The outflow from the wetlands units presents a high water quality, with very low TSS content (<0.3 mg/L), turbidity (<0.5 NTU) and pathogen content (E. Coli < 100 CFU/100 mL), what makes possible its direct reuse for irrigation according to the current Spanish regulation (RD1620/2007) and all uses except A Class quality according to the new European Directive. This is an interesting result after testing VFCWs, which have been scarcely studied in combination with other processes but offer an interesting approach in the near future taking into account the increasing water reuse demand and the update of current water reuse regulations according to the new EU-WWR.

Photo-Fenton solar disinfection

Previous works are focused on the study of the combined effect of solar radiation and the concentration of reagents on the elimination of pathogenic microorganisms and emerging pollutants. To this end, neutral pH treatment has been investigated using the Fe^{3+}-EDDS complex and ferrous sulfate salts, respectively, and as an alternative to reduce operating costs, the application of the Fe^{3+}-NTA complex is currently studied. In addition, in order to use more oxidizing conditions that allow the elimination of antibiotic resistance genes, continuous operation at acid pH is investigated. In order to reuse acid pH treated effluents, further neutralization in calcium carbonate columns obtained as a by-product of the marble industry has been proposed. This would reduce the costs of reagents associated with neutralization. In addition, the calcium carbonate bed acts as a filter retaining the precipitated iron after neutralization. The results obtained are promising. Regarding the total load of emerging contaminants detected in secondary WWT effluents, variable in the range of 20 to 50 µg/L, it is possible to eliminate more than 80% by operating RPR photoreactors with hydraulic residence times close to 30 minutes. At the same time, the concentration of E. coli, close to 10^4 CFU/mL, is reduced below the limit (1 CFU/mL) established in RD1620/2007. In addition, phytotoxicity and cytotoxicity potentials are attenuated and androgenic/glucocorticoid activity and residual water estrogenicity are eliminated. Laboratory studies indicate that with a hydraulic residence time of 30 min and mild oxidation conditions (30 mg/L of hydrogen peroxide and 5 mg/L of iron), 305 $m^3/m^2\cdot$year of WW can be regenerated in 5 cm depth in liquid RPR, with a treatment cost estimated within the current cost of tertiary treatments.

CONCLUSIONS

The feasibility of the microalgae-bacteria consortium using HRAP combined with constructed wetlands has been demonstrated to be a suitable WWT able to provide a high quality effluent that can be directly subjected to disinfection for its reuse in agriculture. Solar photo-Fenton technology eliminates pathogens and emerging pollutants by the action of solar radiation and AOP. The combination of these technologies presents a novel cost-effective WWT process to produce high quality reclaimed water and recover nutrients in small WWTPs.

ACKNOWLEDGEMENTS

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SIMULATION OF AN ACTIVATED SLUDGE PROCESS USING THE ASM1 AND ASAL1 MODELS: A CASE STUDY

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Abstract
The IWA and ASAL models are widely used in research on the Activated Sludge Process for wastewater treatment. In this work, to evaluate and compare the performance of an activated sludge system with extended aeration, data from the North Wastewater Treatment Plant of the Monterrey Metropolitan Area, Nuevo León, Mexico, which treats a flow of 2,520 liters per second, were used. STOAT® software was used for the simulation; it was found that the mathematical models ASM1 and ASAL1 could not reliably estimate the behavior of BOD, NH_4+ and NO_3- in the Wastewater Treatment Plant effluent; a fact that highlights the need to propose a modified model that allows describing an adequate way the behavior of this type of systems, implementing as output components the variables that - according to the regulations - have to be monitored in municipal treatment systems.

INTRODUCTION

The Activated Sludge Process is one of the most widely used wastewater treatment processes in the world. According to Metcalf and Eddy (2003), the basic activated sludge treatment process consists of the following three components: (1) an artificially aerated reactor; (2) solids-liquid separation unit, usually in a settling tank; and (3) a sludge recirculation and purge system, to return and control the number of microorganisms in the aerobic reactor. In most activated sludge wastewater treatment plants, aeration is carried out by bubble dispersion. To evaluate and estimate the removal efficiency of carbonaceous and nitrogenous compounds, dynamic mathematical models have been developed as part of the design and operation of a treatment system. The International Water Association (IWA) has developed activated sludge models. The ASAL models are based on Biochemical Oxygen Demand (BOD) and the IWA models on Chemical Oxygen Demand (COD) and are instrumental in the design, optimization, and operation of municipal activated sludge treatment plants. Mathematical modelling is used to describe and verify the kinetic processes involved in the biological treatment of wastewater and is applied to predict the behavior of the processes, being applied to the design, evaluation, and control of treatment systems (Escalas and Barajas, 2006). The objective of this work was to build, calibrate and validate the ASM1 and ASAL1 models with STOAT® software, applied to the North Wastewater Treatment Plant (WWTP) of the Monterrey Metropolitan Area, Nuevo León, Mexico.

METHODS

The North WWTP has a capacity of 3,000 liters per second (L/s) and handles a treated flow of 2,520 L/s. This work is focused on the study of the activated sludge system of treatment train No. 1 of the WWTP. The arrangement presented is as follows: (1) influent regulation and homogenization tank; (2) Pretreatment, where coarse solids and sands are removed, in addition to elevating the water to facilitate gravity flow (influent pumping); (3) Primary Treatment, which consists of primary sedimentation of the wastewater, where a large portion of the suspended solids are removed, including a significant fraction of particulate organic matter; (4) Secondary Treatment, which refers to an activated sludge process, including the aerobic reactor, the secondary settler, the sludge return system, the sludge purge, and the aeration system; and finally (5) Disinfection of the effluent. The receiving body of the discharges from this plant is the Pesquería River. The process consists of five parallel trains. Each includes an aeration tank of 25,500 m^3 and a secondary sedimentation tank with a surface area of 2,632.98 m^2 and a tank height of 5.50 m. During the operation of the process, the dissolved oxygen (DO) concentration is 0.53 mg/L. First, water and activated sludge characterization data were obtained from the WWTP. The modelling and simulation procedure, based on Wu *et al.* (2016), consisted of configuring the activated sludge system, where the size of the infrastructure was defined; the models to be used were selected;

steady-state influent characterization data were used; and sludge recirculation and purge data were established. Subsequently, the initial operating conditions were obtained; dynamic simulation was carried out; the initial operating conditions were restored; and the calibration of kinetic and stoichiometric parameters was carried out. Finally, the model was validated.

RESULTS AND DISCUSSION

The North WWTP was simulated with STOAT® software. Two years of records (quantitative and qualitative) were used for influent and effluent, respectively (see Fig. 1). The evaluation of model results was divided into carbonaceous and nitrogenous compounds. Calibration of the kinetic and stoichiometric parameters of the models was performed according to the ranges of variation in the literature (Henze *et al.*, 1987).

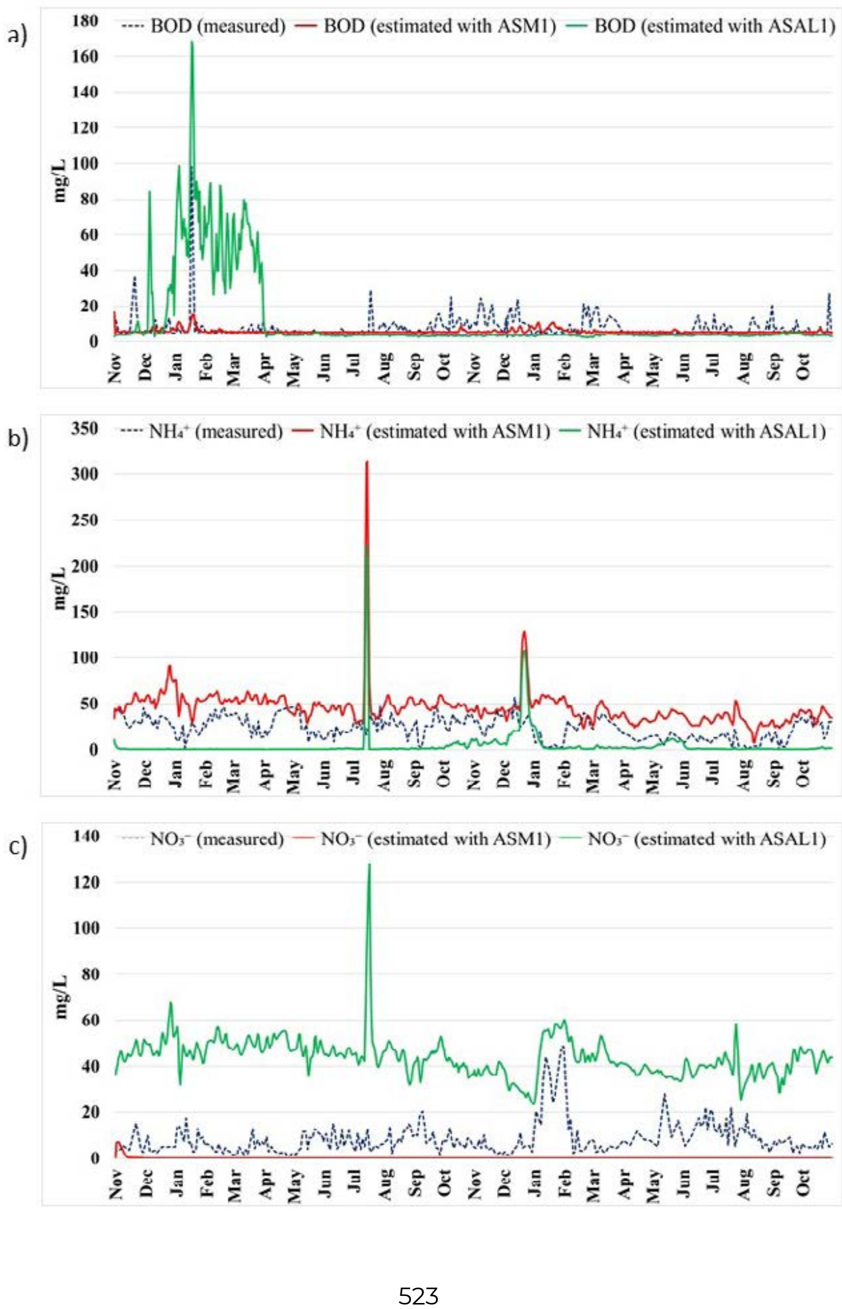

Figure 1. *Daily treated effluent data from the North WWTP of the Monterrey Metropolitan Area, Nuevo León, Mexico (November 2018 - October 2020). Estimates from calibrated models ASM1 and ASAL1; a) BOD, b) NH_4^+ and c) NO_3^-.*

With respect to the comparison of the measured data and the results obtained with the typical ASM1 and ASAL models (Table 1), for BOD, 6.21 and 21.3 mg/L root-mean-square error (RMSE) were found, with 27.2 and 153 percent daily average difference. Regarding NH_4^+, those models in comparison with the measured data, evidenced an RMSE of 30.99 and 27.07 mg/L; and a daily difference in percentage of 251 and 91.0, respectively. Regarding NO_3^-, results in the order of 11.1 and 37.0 mg/L were obtained for the RMSE, with 99.4 and 715 percent daily average difference, respectively.

Table 1. *Statistical analysis of measured and estimated data with ASM1 and ASAL1 models.*

	Root-mean-square error, mg/L	Daily average difference, %
BOD (measured data vs ASM1)	6.21	27.19
BOD (measured data vs ASAL1)	21.28	153.01
NH_4^+ (measured data vs ASM1)	30.99	250.63
NH_4^+ (measured data vs ASAL1)	27.07	90.96
NO_3^- (measured data vs ASM1)	11.07	99.38
NO_3^- (measured data vs ASAL1)	36.99	715.07

The high coefficients of variation obtained, mostly above 0.75, could be explained by the susceptibility of the models to the high variability of the influent concentrations; added to the fact that the fractionation of the components of both models -since the characterizations for the state variables were not available- was estimated using the typical bibliographic coefficients; which increases the degrees of freedom in the results.

CONCLUSIONS

The North WWTP was modeled with STOAT® software. The influent and effluent were measured for two years. The evaluation of the model results was divided into carbonaceous and nitrogenous compounds. The calibration of the kinetic and stoichiometric parameters of ASM1 was performed based on literature values. The experimental results do not show a good relationship with model results, since RMSE values are mostly higher than the averages of the measured data and daily average differences usually exceed 90%.

The typical mathematical models were not able to reliably describe the behavior of BOD,

NH_4^+ and NO_3^- in the effluent of the North Wastewater Treatment Plant of the Monterrey Metropolitan Area, Nuevo León, which points to the need to propose modifications to both modelling approaches to explain the behavior of the variables that, by regulation, are determined in municipal treatment systems in Mexico.

The dynamic simulation of WWTP processes is a very important tool that allows: predicting the response to changes in flow rates, influent concentrations and operating conditions; and develop new models, where components and reaction processes are proposed, to obtain the conversion equations that will allow us to meet the proposed objective, such as optimizing the process, evaluating a new design, carrying out a reengineering, establishing control schemes in the operation of the WWTP and/or evaluate the performance of the wastewater treatment systems.

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THE PROCESSING AND ANALYSIS OF DATA AS AN INNOVATIVE TECHNOLOGY FOR IMPROVING THE EFFICIENCY AND MAINTENANCE OF WASTEWATER PLANTS (SMALLWAT21)

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Alicia Poncela Huerta[4]

Abstract

Goal 6 (UN SDG): *Ensure availability and sustainable management of water and sanitation for all.*

A water treatment plant monitoring platform contains a lot of knowledge about plant systems that can be used to develop possible proactive maintenance programs.

"More than 80% of wastewater resulting from human activities is discharged into rivers or the sea without any treatment, resulting in pollution."

Plant operators can use such platforms to analyze performance data on assets and processes. This allows them to build an accurate picture of the maintenance tasks to be performed and schedule them optimally, as the information can be analysed to forecast what might happen under a set of circumstances.

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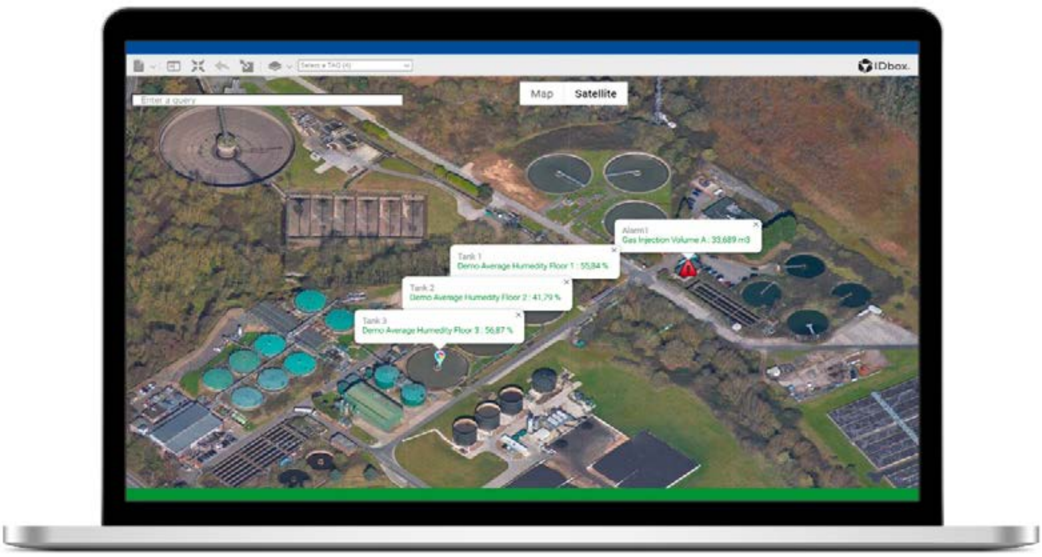
This allows water companies to have preventive maintenance, as they will be able to plan in advance their response to certain casuistry or early detection of anomalies. In addition, along the same lines, the platform's analysis gives managers the necessary information to estimate specific plans and procedures in anticipation and mitigation of possible future problems throughout the network, such as leaks, outages, excess flows, etc. Analytical tools such as heat maps, periodic clustering or scatter plots can be used to identify patterns of behaviour and early detection of anomalies.

INTRODUCTION

Monitoring and optimization of wastewater plants can go beyond a single plant to incorporate satellite plants and even reach the entire wastewater network. For example, integrating data from different sources, such as bringing together weather data, data from utilities such as Red Eléctrica Española (REE) and the plant's own energy consumption, allows the platform to build a picture of what is happening and could happen in certain weather situations, also pinpointing where consumption forecast events or alarms are likely to occur.

The fact that the platform allows information to be exploited from multiple devices and locations also means that operators can access the stored information without having to be familiar with the plant's own technology. Thus, they can quickly learn which processes and variables are involved in the plant's behaviour.

Image 1. *Interactive map of a water treatment plant with dynamic indicators from the IDboxRT platform.*



The water sector is awash with vast amounts of data. Water utilities are full of suppliers whose pumps, chemical dosing systems and various equipment are capable of functioning as IoT (Internet of Things) devices. However, most water companies are currently analysing how digital technology can affect their business model. Most water utilities are able to collect data from their plants, but the big challenge is knowing how to use it and for what purpose, so that it provides value for plant and wastewater optimization. This can be done through monitoring platforms that integrate, process and analyze the measurements taken by the various instrumentation equipment and agnostic sensors to obtain plant-wide or network-wide KPIs (key performance indicators).

Target 6.b (SDG) *"Support and strengthen the participation of local communities in improving water and sanitation management."*

Taken to the example of an industrial plant, the platform could collect data on wastewater inflows and outflows in the treatment process. This data could describe the quality of the wastewater entering the plant and the quality of the treated water being discharged at the other end. Monitoring tools bring all this data together to build a complete picture of how the plant is performing in real time.

A case study shows how wastewater plants typically pump air into a biological treatment process using a blower, which introduces a quantity of air based on a set point, which is almost always fixed when the plant is installed, and does not consider future changing conditions. A plant optimization platform can monitor the incoming wastewater and adjust the blower setpoint to the optimum level to match that water quality or volume. This is a continuous improvement, as the biological treatment process runs more efficiently and reduces the energy consumed in the process, ultimately reducing costs. It could even be adjusted remotely via the platform, avoiding the need for manual intervention by an on-site operator, which also reduces the cost of operating personnel.

Image 2. *Dashboard and synoptic view of wastewater treatment plant in IDboxRT monitoring tool.*

RESULTS AND CONCLUSIONS

The water sector has a clear trend towards digitalization, strongly driven at the European level for better development and sustainability of natural resources. Monitoring this sector is the key vector to achieve the UN Sustainable Development Goals (SDGs) to ensure universal access and sanitation, preserve resources, protect public health and enable more sustainable development.

Target 6.4 (SDG) *"By 2030, significantly increase the efficient use of water resources in all sectors and ensure sustainability of freshwater withdrawals and supplies to address water scarcity and significantly reduce the number of people suffering from water scarcity."*

This integration of technological innovation in the companies of the sector leads to an increase in business benefits. The integration of all plant devices in use means the elimination of information islands. The inspection and monitoring of the network in real time also allows visualization from any device, which leads to optimal decision making and significantly reduces response times. On the other hand, thanks to these monitoring tools it is possible to reduce consumption due to the early detection of leaks. Furthermore, thanks to alarms, notifications and advanced trend analysis, innovative monitoring technologies ensure increased network performance, high efficiency in incident resolution and efficient data management.

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BIOAUGMENTATION IN WASTEWATER COLLECTION SYSTEMS TO REDUCTION CONTAMINATION LOAD, BIOLOGICAL NUTRIENTS REMOVAL AND ELIMINATION BAD ODORS

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Abstract

A study is presented on the implementation of In-Pipe technology, biological treatment of wastewater, generated in the urban nucleus of a municipality of about 18,000 inhabitants (Lora del Río, Seville), based on bioaugmentation in the sanitation network. The system consists of the continuous dosage of a concentrate of selected microorganisms that gives rise to a modification of the existing biofilm in the sanitation network through competitive exclusion. This leads to a partial purification of the pollutant load of the residual water, the elimination of bad odors in the collector network and at the point of discharge and a biological decrease in nutrients.

INTRODUCTION

The studied municipality generates almost 3,730 m3 of wastewater daily; It lacks a WWTP, discharging its water into the public channel with a characterization assimilable to urban wastewater.

Bioaugmentation is the use of highly specialized microorganisms to increase and improve the degradation capacity of the natural microbial population. It consists of inoculating live microorganisms that have been specially isolated and selected for their high capacity to degrade pollutants promoting their biodegradation, in this case, directly in the sanitation network.

The bacteria dosed are mostly facultative anaerobic, planktonic, and capable of sporulation. They have a faster growth rate than the intestinal anaerobic bacteria that are typically part of the sewer biofilm, mainly due to their ability to use oxygen as the final electron acceptor under aerobic conditions, but also due to their flexibility in low oxygen and anoxic conditions. Therefore, they can grow under different concentrations of oxygen in a way that allows them to shut out anaerobic intestinal bacteria in the competition for available nitrogen and carbon compounds necessary for their proliferation.Additionally, they do not use the assimilative metabolic pathway of sulfate reduction.

Bacteria that are part of this bioaugmentation mainly use sulfur in the form of sulfate through its reduction following the assimilative metabolic route. During this process, sulfate is first transported into the cell activated by ATP-sulfurylase. The sulfate (APS, or adenosine phosfosulfate), once internalized, following the assimilative metabolic pathway that initiates ATP kinase, it is reduced to hydrogen sulfide and then immediately incorporated into the amino acid cysteine by O-acetylserinesulfhydrylase.Therefore, the excretion into the environment of the hydrogen sulfide responsible for bad odors is inhibitedand its subsequent oxidation to sulfuric responsible for corrosion.

The bacteria found in the dosed suspension are not restricted to environments with high oxygen concentrations. If faced with a limitation of this, they switch to anaerobic fermentation or use electron receptors such as nitrate to continue the degradation of COD with the added advantage of reducing nitrate to nitrite, helping to maintain the growth of Anammox bacteria by providing them with the necessary source of nitrite as they easily reduce nitrate to nitrite under anoxic conditions.

The dosed microbes are capable of solubilizing phosphate from insoluble sources, such as phosphate combined with calcium. A large proportion of the newly released phosphate is available for use by other microorganisms. Furthermore, they carry out fermentations in the anaerobic zones of the collectors to produce the volatile fatty acids (VFA) required by phosphate accumulating organisms (PAO) that prefer low molecular weight fermentation product substrates, particularly acetate. The COD from the influent must be broken down into a substrate that can be easily utilized by the PAO. Due to the ability of PAOs to store polyphosphates, they have the necessary energy to assimilate acetate in anaerobic areas.

METHOD

A trial of this technology was carried out in the Lora del Rio Project. Beginning with a thorough study of water quality and flow rates, various additional analyses were carried out beforehand to confirm and adapt to the actual data gathered during the trial period.

Subsequently, the sanitation network of the urban nucleus is studied, observing that it is divided into two zones with respect to the network. Zone 1 flows into the old Churre stream (Point 1, or P1) and zone 2 into Point 2 (P2) in the same channel. The union of flows from P1 and P2 is referred to as the Common Point (PC). Zone 1 collects approximatelytwo-thirds of the town's wastewater. The network does not have pumping stations, its flow being entirely driven by gravity.

Figure 1. *Dumping points*

The appropriate calculations are made and, in a first phase of 6 months, 9 continuous dosing points (24 hours / 7 days, dosed at 1.8L per month, are established for the characterization of the water to be treated at each point. and 4 concentrated activated dosing points (CAD) of 20 liters dosed all at once, every 30 days.

The dispensers are installed, which are made up of a container with the monthly microbial solution, a dosage pump, an electronic control board, and batteries; all this enclosed in a double hermetic box (Figure 2), which hangs from the walls of the collector access manholes, from which an ejector arm connected to the pump doses the solution into the wastewater stream (Figure 3).

Figure 2. *DoserComponents*

Figure 3. *Doser installed*

The start-up consists of activating the collectors by adding 20-40 liters of activated concentrate at each dosing point, and at the CAD points, which will be repeated after 48 hours.

Monthly maintenance operations are carried out, checking the correct operation of all components and the replacement of the microbial solution, as well as the batteries if necessary.

RESULTS AND DISCUSSION

The project, issued by public tender by the City, is scheduled for a three-year trial, with the primary objective being the reduction of COD, BOD5, and Total Suspended Solids (TSS), by the end of year 3(Table 1).

Lora del Río	COD(mg/l)		BOD5 (mg/l)		TSS (mg/l)		Average Reduction
	Initial	Objective	Initial	Objective	Initial	Objective	
P1	827	662	470	376	216	173	20%
PC	690	552	300	240	250	200	

Table 1. *Starting data and objectives*

The results of the bioaugmentation are compared monthly with the starting parameters in P1. In addition, a comparison with PC was analyzed to assess the total flow from all areas of the municipality. The results and variations can be seen in table 2.

Lora del Río		COD (mg/l)		BOD5 (mg/l)		TSS (mg/l)		Average Reduction
Point	Month	Value	Reduction	Value	Reduction	Value	Reduction	
P1	Jan/21	801	3%	500	-6%	155	28%	8%
P1	Feb/21	797	4%	400	15%	188	13%	10%
P1	Feb/21	745	10%	380	19%	180	17%	15%
PC	Mar/21	399	42%	200	33%	144	42%	39%

Table 2. *Results*

As seen in the results thus far, a progressive reduction in the three parameters analyzed has been observed.

Similar results were obtained from a pilot trial carried out in the municipality of Villalba de los Barros in 2011-12 under the order of PROMEDIOand the participation of UEX. In that case, the results of the reductions at 12 months of continuous dosing reached 46% in COD, 42% in BOD5, 40% in SST and 34% in TN.

In addition to the objective results presented, there has been an anecdotal observation (confirmed by City employees) of a dramatic reduction in foul odors, as measured by a reduction in citizen complaints to the sewer department. At present, the defense wall of the Guadalquivir River through which the old Churre stream drains its wastewater into the river, has become a place for walks for the inhabitants of the municipality.

CONCLUSIONS

The bioaugmentation system in the sanitation network, with the participation of selected microorganisms, leads to a transformation of the biofilm that covers the collectors, to partially start the treatment of wastewater and suppress bad odors by inhibiting production of hydrogen sulfide; if there is no secretion of this, the production of sulfuric does not occur, thus avoiding corrosive processes, both in the sanitation network and in the WWTP.

These reductions in the analyzed parameters of wastewater, in addition to causing less pollution in the discharge and eradicating annoying bad odors, can be fully extrapolated to lower operating costs of the network itself and of the future WWTP. These conclusions were confirmed in the previous experience in the Badajoz municipality when the hours of aeration in the WWTP biological reactor were reduced by 40%.

The lower polluting load influencing a WWTP would entail, in the same reduction ratio, savings in operating costs related to electrical costs, sludge management and chemical products associated with the elimination of bad odors.

This system can be especially useful both for those small municipalities that still do not have a WWTP, as well as for those (large and small) whose WWTP has become obsolete and an expansion is necessary, mainly due to receiving greater load for which it was designed.

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EFFECT OF SALINITY ON THE EFFICIENCY OF WASTEWATER TREATMENT OF AN ACTIVATED SLUDGE SYSTEM

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Abstract

In recent decades, the discharge of saline and hypersaline wastewater from industrial wastewater has increased significantly, affecting the efficiency of pollutant removal in the biological treatment of Wastewater Treatment Plants (WWTP). (He et al., 2017; Lefebvre et al., 2007). In the present work, the change in the removal efficiency of the Chemical Oxygen Demand, Total Suspended Solids, N and P from a biological treatment system was studied, as well as the effect on the microbial community when subject to progressive increases in saline water. Evaluating the changes in a biological purification system subject to brackish wastewater will lead to the optimization of municipal treatment systems receiving said discharges, reducing the volumes of liquid effluents that are sent to final disposal sites.

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INTRODUCTION

Today there are a large number of industries that often generate wastewater with high salt content, these include milk production, fish processing, oil production, chemical industry, pharmaceutical industry, among others. Highly saline wastewater (> 10,000 mg / L) usually has a very slow process of degradation of organic matter in biological purification systems, this due to the toxic effect of sodium on biomass (Cortes-Lorenzo *et al.*, 2012; Chen *et al.*, 2018). In the process for the biological treatment of saline wastewater, the salt content affects the physical and biochemical properties of the activated sludge, as well as the microorganisms present (Reid *et al.*, 2006; He *et al.*, 2017). The objective of this work is to determine the effect of salinity on the efficiency of wastewater treatment of an activated sludge system when subject to gradual increases in salinity in the affluent.

METHODOLOGY

A biological system of activated sludge was designed to which the retention times of each unit were determined, preserving the hydraulic characteristics of a WWTP (Annexes A). The anaerobic and aerobic reactors were inoculated with biological sludge from a municipal WWTP located in Mexico. Once the biological system had stabilized, saline water from the industrial discharge was added; This was carried out by adding doses equivalent to the progressive and cumulative increase of 10% of the normal electrical conductivity (EC) of the effluent wastewater to the real WWTP (that is, 190 μS / cm of progressive increase), period subject to the response of the bacterial consortia (which can be decreased or increased), until reaching the dosage equivalent to the total flow rate of the industry's water rejection. The analyzes set out in Table 1 were carried out.

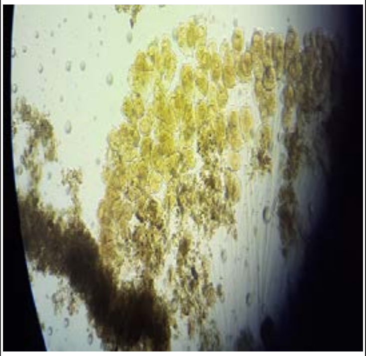
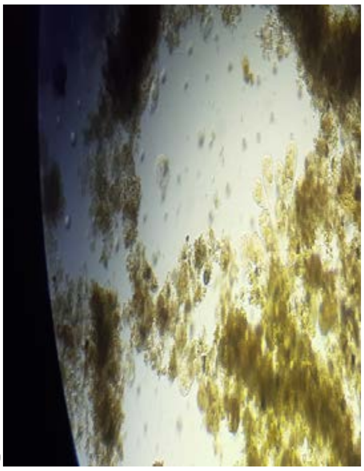
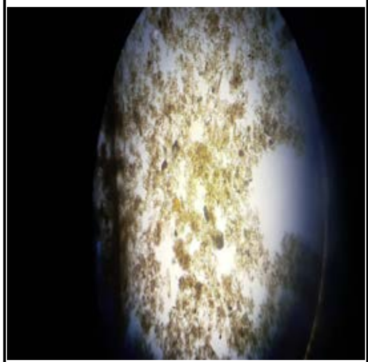
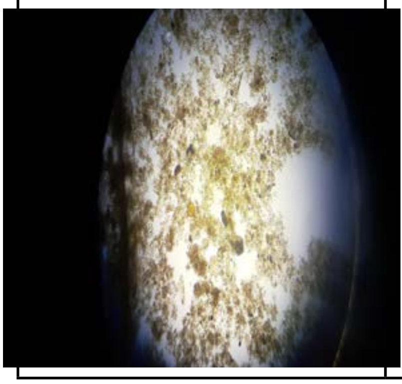
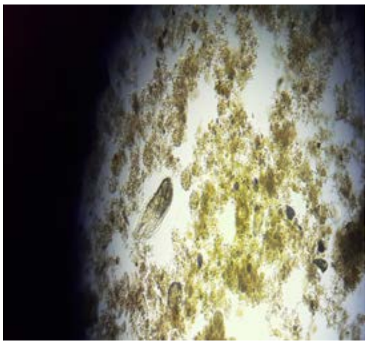
To evaluate the quality of the sludge, a drop of sample from the aerobic reactor was placed on a slide and taken to the microscope, the search for bioindicator organisms was carried out.

Table 1. *Parameters to monitor and location in the system on a laboratory scale.*

Site	Parameter								
	DQO	DQO Soluble	NH_4	NO_3	PT	SDT	pH	Conductivity	Temperature
Afluente PTAR	√	√	√		√	√	√	√	√
Reactor anaerobio efluente	√	√				√	√	√	√
Clarificador efluente	√	√		√	√	√	√	√	√

RESULTS AND DISCUSSION

Table 2. *Illustrations of the observation in the activated sludge in the stabilization stage of the system on a laboratory scale.*

A) 	B) 	C) 
D) 	E) 	

The images presented in Table 2 were taken in the stabilization stage of the system on a laboratory scale. In section A) and B), according to the results, organisms of the genus *Vorticella* were presented, these indicate a Mixed Liquor of very good quality. A large number of organisms of this type were found, so the laboratory-scale system has good oxygenation and is in optimal conditions (Ferrer-Polo *et al*., 2018). In section C), organisms of the phytoflagellate type were presented, which indicate a good age of the sludge and a good quality of the treated water (Ferrer-Polo *et al*., 2018). In section D), Rotiferous were presented, this indicates a low concentration of suspended solids from mixed liquor (SSLM). Finally, in section E): Spiroteric of the *Holosticha* genus, which is a good bioindicator and is associated with good quality effluents, it is generally associated with slightly loaded wastewater (López-Vázquez *et al*., 2017). Based on the results generated in the microscopic observation, a large number of microorganisms were presented and are associated with a medium-aged sludge with good stability and good clarification, therefore it was possible to stop the stabilization stage of the biological system, after two weeks of constant evaluation.

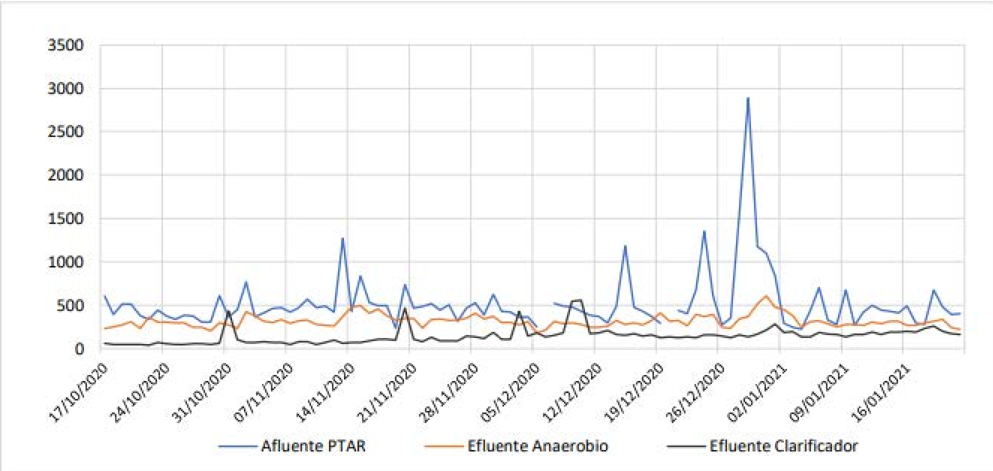


Figure 1. *Determination of the Chemical Oxygen Demand (COD) in the laboratory scale system.*

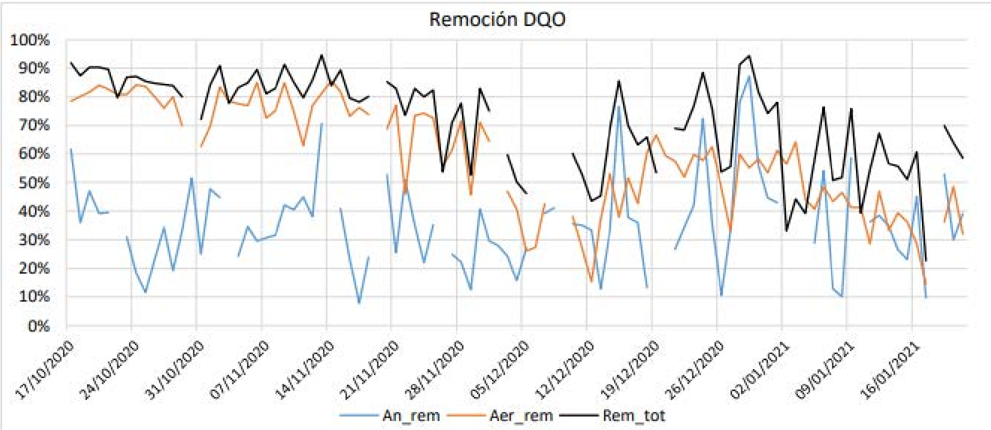


Figure 2. *Chemical Oxygen Demand (COD) removal efficiency in the laboratory-scale system.*

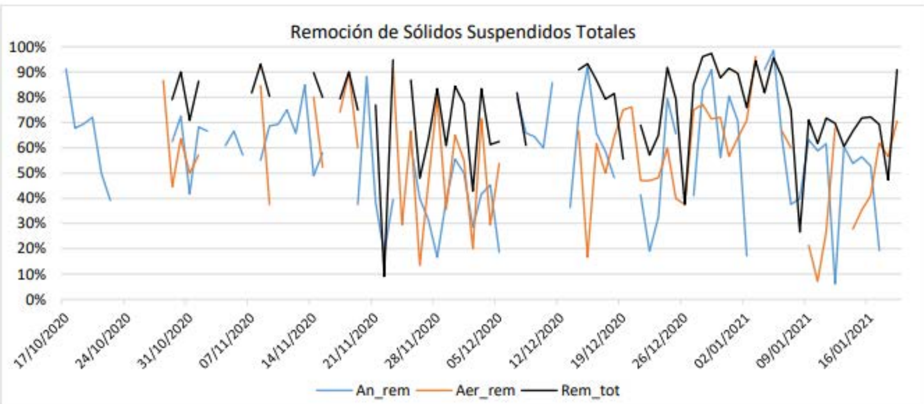


Figure 3. *Total Suspended Solids (TSS) removal efficiency in the laboratory scale system.*

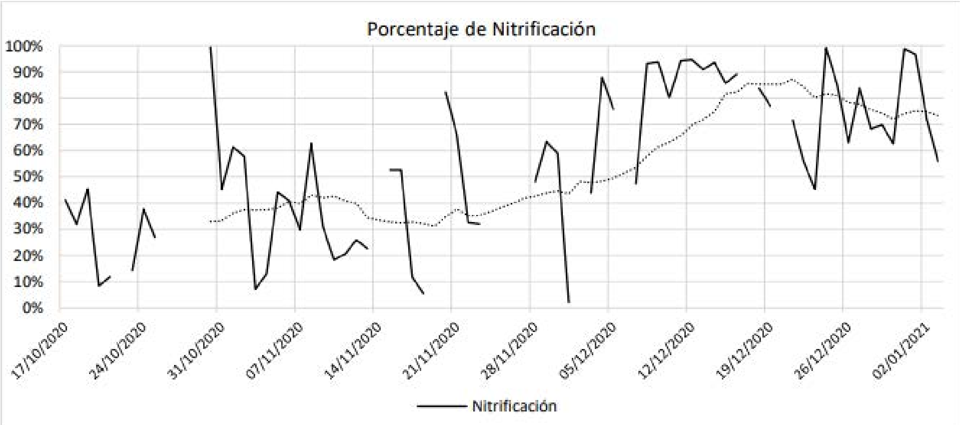


Figure 4. *Nitrification percentage of the system at laboratory scale.*

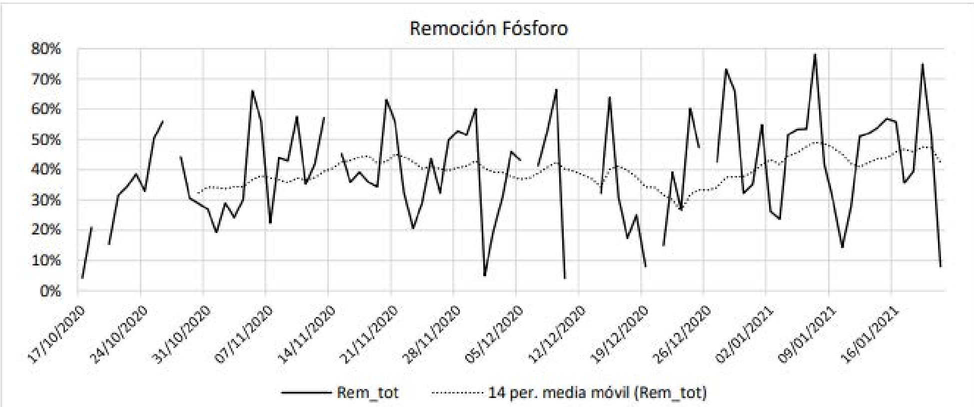


Figure 5. *Phosphorus Removal Efficiency on a laboratory scale system.*

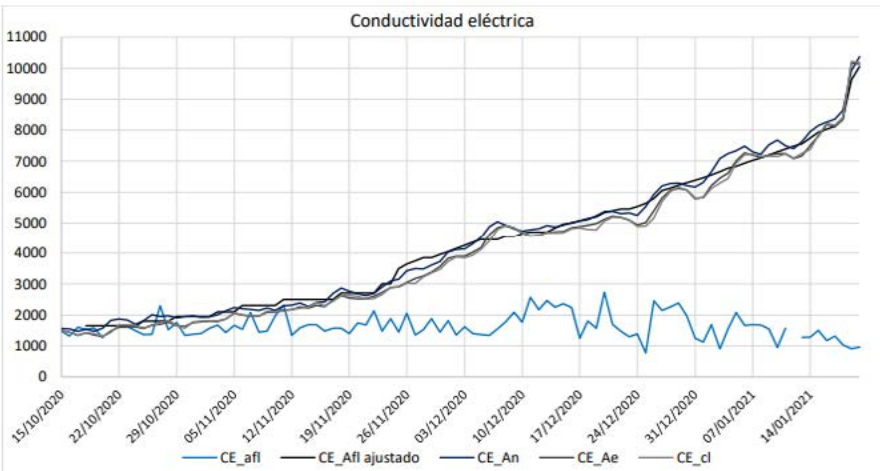


Figure 6. *Electrical Conductivity (EC) of the system on a laboratory scale.*

CONCLUSIONS

It was found that even for electrical conductivities of up to 10200 μS/cm in the reactors, the mean removal efficiency of SST was not significantly affected in the laboratory-scale system. The efficiency in the nitrification rate was higher during the period of increase of EC in the reactors, than during the period of stabilization of the system. The nitrification process was not affected by the increase in EC in the laboratory-scale system. The sustained increase in electrical conductivity carried out in the system did not affect the phosphorus assimilation efficiency by the bacterial consortia. Based on the results generated in the macroscopic observation, in week 1 of experimentation (conductivity 1650 μs/cm) a large number of bioindicator organisms with movement were observed, this indicated a good quality of sludge, as well as a good clarification, when increasing the conductivity to 7570 μs/cm reduces the population of bioindicators, however rotifers and ciliates are still present, indicating that the process maintained adequate conditions for its operation.

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DESIGN AND IMPLEMENTATION OF A ROTATING BIOLOGICAL CONTACTOR PROTOTYPE AS AN ALTERNATIVEFOR WASTEWATER TREATMENT AT INTERMEDIATE CITIES

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Renato Montoya[2]

Abstract

The increasing scarcity of water has turned wastewater into a valuable and necessary resource for agricultural production, which requires adequate treatment to guarantee its safe reuse.Rotating Biological Contactors (RBC) are an aerobic treatment system of attached biofilm made up of a series of discs mounted on a shaft and submerged in a tank.Due to the rotation of the system, the microorganisms adhered to the discs are capable of carrying out wastewater depuration processes. These treatment systems provide high performance rates occupying small areas of land, thus made an attractive option for the decentralized treatment of wastewater in intermediate cities or small populated centers.However, its use is limited in developing countries mainly due to the cost of its implementation. At the end of 2018, the Universidad PrivadaBoliviana and the Aguatuya Foundation have designed and implemented a pilot-scale

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treatment plant based on a rotating discs reactor using locally available materials and services.The objective of this project was to evaluate the performance of this prototype in its start-up and operation phases.The maximum global efficiencies obtained were: 79, 78 and 88% for TCOD, SCOD and TSS respectively, thus demonstrating that the prototype, designed and implemented with polystyrene discs, is efficient in the removal of organic matter and suspended solids for the characteristics for which has been designed.

INTRODUCTION

The increasing demand of water for consumption, industrial production, and agricultural production, aggravated by the low availability of the resource due to factors such as the effects of climate change, population growth, surface, and underground pollution, among others, has turned the reuse of treated or untreated wastewater,a common practice in agriculture in Bolivia. Nonetheless, the reuse of wastewater is related to impacts on human health and the environment. The implementation of treatment systems constitutes a mechanism to reduce the risks associated with the reuse or discharge of wastewater. However, in many cases poor wastewater treatment is performed. In Bolivia, 78% of the treatment plants do not function properly, in most cases due to inadequate operation and maintenance and poor project engineering(MMAyA, 2020). A pre-conceived idea regarding the treatment of wastewater in regions with scarce availability of resources for sanitation is that it should be of low cost and with low mechanical, energy, and specialized labor requirements. Rotating Biological Contactors (RBC) or biodiscs reactors are aerobic wastewater treatment systems of adhered biofilm that are made up of a series of rotating discs mounted on a shaft and partially submerged in a tank through which the wastewater flows. The microorganisms adhered to the discs carry out organic matter depuration processes.These systems represent a treatment alternative that achieves high rates of elimination of organic matter, has low land requirements, low energy consumption and low operating and maintenance costs, however, their use has been limited due to the cost of implementation, mainly due to the high cost of discs and the need to import them(Cortez, Teixeira, Oliveira, & Mota, 2008). In 2018, the Universidad Privada-Boliviana (UPB), in a project competition organized by the CIMAS program of the Agenciaespañola de cooperación para la investigación y el desarrollo AECID, obtained funds for the development of a research project. Under a research collaboration agreement in treatment technologies, the UPB and the AGUATUYA foundation carried out the design, implementation, and evaluation of a prototype of a wastewater treatment plant (WWTP) that uses anRBC as the main biological process for wastewater depuration. The main objective was the design, implementation and evaluation of a treatment plant prototype based on a Rotating Biological Contactor as an alternative for wastewater treatment at intermediate cities.

METHODOLOGY

The methodology of this study considered design of unit processes that composed the prototype, implementation, start up and evaluation.

Design of the RBC prototype

This study considered the following treatment sequence: a primary settler, an RBC and a secondary settler. For the design of the RBC, the characteristics of the wastewater from Tolata intermediate city, collected from 6 composite samplings carried out between August and December 2018 were considered(Echeverría R. et al., 2019). The BOD, COD and TSS values considered for the design were: 396, 795 and 361 mg/l respectively. The wastewater flow generated by Tolata was 1.3 l/s. For the design of the prototype, it was considered to treat one tenth of this flow, that is, 0.13 l/s. The RBC has been designed based on parameters recommended in the literature by some authors such asRomero Rojas, (2004)andMetcalf & Eddy (2003)and considering other practical examples such as those presented byDautant (2018), Pérez (2010)andMata (2012). In addition, some considerations set out in the guide for the application of the biodiscs process for the treatment of wastewater prepared by the U.S Environmental Protection Agency have been considered(U.S. EPA, 1971). A summary of the technical design specifications of the RBC is presented in Table 1.

Table 1. *RBC Design Technical Specifications*

Design Flow	0.1 l/s
Shaft arrangement	parallel to flow direction
Submerged area	40%
Disc diameter	0.9 m
Disc thickness	2.5 mm
Disc material	Polystyrene
Tank material	Stainless Steel
Number of the stages	3
Disc rotation speed	2.5 RPM
Useful adhesion Surface	187 m^2
Number of the discs	151
Reactor length	2.5 m
Reactor volume	1.4 m^3

Implementation

RBC has been built using local available materials. Polystyrene has been used to manufacture the discs. Both the tank and the baffles that separate it into stages are made of stainless steel to avoid corrosion damage that could cause contact with wastewater. The tanks used as homogenization tank, primary settler and secondary settler have been acquired from local industries and the chosen volume is close to the design one. The prototype has been located in the northwest sector on the premises of the Tolata WWTP.The resulting prototype can be seen at Figure 1.

Figure1. *RBC prototoype*

Start-up

The start-up consisted of letting the wastewater flow until the adaptation of a biofilm on the surface of the discs.

Prototype performance evaluation

The functioning of the prototype was evaluated based on the physicochemical characterization of the wastewater at different points of the treatment.Grab samples were collected at the inlet and outlet of each process that compose the treatment.The codes that refer to the sampling points can be seen in Figure 1.Analyzed parameters were pH, temperature, dissolved oxygen (DO), total chemical oxygen demand (TCOD) and soluble chemical oxygen demand (SCOD), total suspended solids (TSS), ammonia nitrogen (NH_3-N) and phosphorus (P). On-site measurements were carried out and taken samples were analyzed at UPB laboratories following normalized procedures of the *Standard Methods for Examination of water and wastewater*. (APHA/AWWA/WEF, 1999).

RESULTS AND DISCUSSION

The average results of the physicochemical characteristics of six monitoring campaigns carried out between August 2019 and December 2019, in addition to the efficiency of the process, are presented in Table 2.

Table 2. *Average results of the wastewater characterization*

Sample point	pH	Temperature °C	D.O (mg/l)	TCOD (mg/l)	SCOD (mg/l)	TSS (mg/l)	NH_3-N (mg/l)	P (mg/l)
HT	7.4	20.5	0.30	686	294	211	81.6	11.8
PS	7.3	19.9	0.53	574	259	149	79.7	11.0
RBC-1	7.8	18.6	1.27	360	137	103	72.1	10.2
RBC-2	7.8	18.2	1.26	281	115	81	66.7	10.0
RBC-3	7.9	17.6	1.51	227	93	62	61.1	9.8
SS	7.6	18.0	0.83	215	83	46	60.3	10.2
Efficiency %				68±13	64±17	82±14	16±33	10±17

RBCbiofilm adaptation time was achieved approximately one month after start-up. This period is similar to those reported in literature (Pylnik & Dueck, 2012). The overall efficiency achieved by the system is in a range of 68-79%, 68-78% and 81-88% for the removal of TCOD, SCOD and TSS, respectively. This mean does not consider the efficiency measured in the system adaptation period.

RBC by itself presents an average efficiency of 63, 65 and 60% in the removal of TCOD, SCOD and TSS respectively. The first stage of an RBC is the one that supports the highest organic load, so the area of the first stage should be at least twice of the area of the rest of the stages(Dautant, 2018). This criterion was considered for the design of this unit and it proved to be an adequate parameter to guarantee its optimal operation. The dissolved oxygen levels recorded in this study are in the range of 1.0 to 1.3 mg DO/L, below the levels recommended in the literature(Akhbari, et al., 1992), that mention that these levels should be between 1.0 and 2.0 mg DO/l. In high altitude regions, such as Bolivia, oxygen levels decrease due to atmospheric pressure, so optimizing these parameters is essential for a good performance of an aerobic treatment system.Low DO levels were corrected by increasing the rotational speed from 2 to 3 RPM. No significant decrease in nutrients was reported, which is desirable for the reuse of water in crop irrigation. Under optimal operational parameters, the maximum efficiencies achieved by RBC were: 81, 74 and 76% for TCOD, SCOD and TSS.

CONCLUSIONS

A WWTP prototype consisting of a homogenization tank, a 3-stage RBC and a secondary settler was designed, implemented, and evaluated. The material used as the support medium was polystyrene. The adaptation period of the system was approximately one month. The overall efficiency obtained in the removal of TCOD, SCOD and TSS was 63, 65 and 60% respectively. Once the start-up period had concluded, the maximum global efficiencies were 79, 78 and 88% for TCOD, SCOD and TSS respectively, and the maximum efficiencies of the RBC were: 81, 74 and 76% in the removal of TCOD, SCOD and TSS. The dissolved oxygen levels recorded ranged from 1.00 to 1.32 mg/L at a speed of 3 RPM. The designed system has proven to be efficient in the removal of organic matter and suspended solids, so its use in full-scale treatment plants is recommended.

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