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Impact of Prior Physico-Chemical Treatment on the Clogging Process of Subsurface Flow Constructed Wetlands: Model-Based Evaluation

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Abstract The objective of this study was to check the effect of the use of a physico-chemical treatment on the clogging process of horizontal subsurface flow constructed wetlands by means of dynamic modelling. The hydraulic submodel was based on series as well as parallel branched complete stirred tanks of equal volume. The model was validated with data obtained from 2 identical experimental wetlands, which had a surface area of 0.54 m^2 and a water depth of 0.30 m, and that were monitored over a period of 5 months. One of the wetlands was fed with settled urban wastewater, whereas the other with the same wastewater, but previously treated with a physico-chemical treatment. In the model, pore volume reduction depends on the growth of bacteria and on solids retained. The effluent concentrations of COD and ammonium in both experimental wetlands were very similar in all the conditions tested, and therefore the physico-chemical treatment did not improve the removal efficiency. The model indicated that after 120 days of operation in some regions of the wetland fed with settled wastewater the porosity decreased in a 17%, whereas in the other wetlands it only decreased as much as 6%. The use of a prior physico-chemical treatment is a good alternative for avoiding an anticipated clogging of subsurface flow constructed wetlands.

Keywords Biofilm · Dynamic model · Horizontal flow · Pore volume reduction · Wastewater

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1 Introduction

The use of subsurface flow constructed wetlands (SSF CWs) is increasing worldwide for the sanitation of small rural communities. Practical experience gained from full scale applications and intensive research efforts in recent years have led to considerable improvements on the design and operation of these systems. From a practical point of view, the main drawback of the technology that still needs to be solved is the progressive clogging of the granular medium (Blazejewski and Murat-Blazejewska 1997). Clogging is a complex process that involves the inert retained solids, biofilm growth, plant decay products, granular medium properties and chemical precipitation. The appearance of clogging usually takes several years since the start of operation of the systems. Nowadays, when a SSF CW is built, it is unknown whether it will clog during the projected life-span of the system, which is normally 20 or more years. Long term evaluation of the clogging process could be simulated through the use of dynamic models.

One evident way to reduce bed clogging is to remove the influent particulate matter. In practice, for reducing the particulates the wastewater flows through septic or Imhoff tanks prior to be conveyed to the SSF CW. However, the efficiency of this type of primary treatments for the removal of organic particulated solids is very limited, in the range of 30 to 40% in terms of BOD_5 (Tchobanoglous 2003). Significantly higher removal efficiencies could be attained by applying a physico-chemical treatment prior to the SSF CW (Metcalf and Eddy 1998). With this treatment, the removal of particulates can rise giving place to removal rates of even more than 90% in terms of BOD_5. The main disadvantages of the use of this treatment are a greater primary sludge production, the extra cost for chemicals and the fact that one can deviate considerably from the concept of a low-technological natural treatment system. However, at the same time, there is a clear reduction of the risk of clogging and high phosphorus removal could be attained (Meers et al. 2006). Recent developments in compact physico-chemical treatments seem to make this technology very suitable for their use in conjunction with SSF CWs (Tchobanoglous et al. 2003).

To our knowledge, the effect of the removal of particulate organic matter through the use of a physico-chemical treatment on the efficiency of SSF CWs has scarcely been investigated. The reduction of the content of particulate compounds will reduce the contaminant loads and it may have positive effects in the long term operation on the efficiency of the SSF CWs for the removal of organic matter, nutrients and other contaminants commonly found in urban wastewaters. Thus, the implementation of a prior physico-chemical treatment would not only reduce the risk of clogging, but it could also improve the effluent quality of the systems.

In this study the effect of the use of a physico-chemical treatment on the clogging process (reduction of pore volume) is evaluated by means of dynamical modelling. The model has been validated with data obtained from two experimental horizontal SSF CWs which were used to evaluate the effect of a physico-chemical treatment on the removal efficiency. One of the wetlands was fed with wastewater which was previously settled, and the other one with the same wastewater but in addition treated with a physico-chemical process.

2 Material and Methods

2.1 Experimental Wetlands

The investigation of the effect of the physico-chemical treatment on the removal efficiency was carried out by means of two identical experimental horizontal SSF CWs placed on the roof of the Hydraulics, Coastal and Environmental Engineering Department (Technical University of Catalonia, Barcelona, Spain). Details of the experimental wetland's design and construction and on the treatment performance can be found in Caselles-Osorio and García (2007). Briefly, the two identical wetlands consisted in a plastic container (0.93 m length, 0.59 m width and 0.52 m height) filled with gravel extracted from near the outlet of a horizontal SSF CW of a pilot system located at Les Franqueses del Vallès, Barcelona. This pilot system started to operate in March 2001 for treating the urban wastewater of a housing development and was out of commision at the end of 2005 (García et al. 2005). The containers used for the experimental wetlands had a drainage pipe located on the bottom of one side (Fig. 1). In both experimental wetlands the gravel layer depth was 0.35 m (D_{60}=3.5 mm, C_u=1.7 and initial porosity of 40%) and the water level was maintained approximately 0.05 m under the gravel surface to give a water depth of 0.30 m. The experimental wetlands

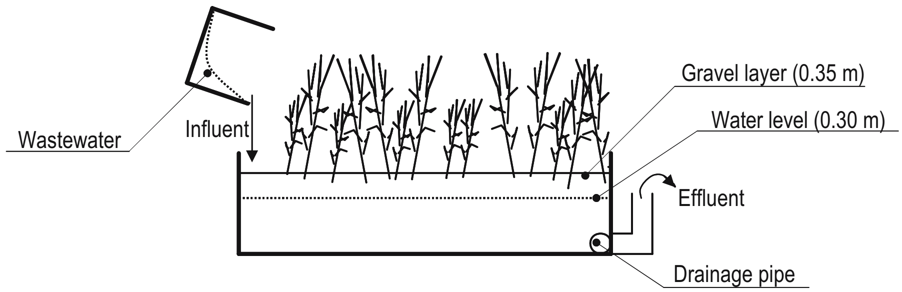


Fig. 1 Schematic representation of the two identical experimental wetlands used in this study. Each wetland was fed with wastewater from the same origin but with different pretreatment

were planted in June 2004 with rhizomes of common reed (*Phragmites australis*).

2.2 Experimental Procedure

Both wetlands were fed once per day (intermittent mode) with fresh urban wastewater obtained daily from a nearby sewer. One of the wetlands was fed with settled wastewater (S), and the other with the same wastewater, but previously processed with a physico-chemical treatment (PC). This treatment was carried out using a standard jar-test device, and consisted in 1 min of coagulation (coagulant: Tanfloc-SG; dose: 70 mg/l), 15 min of flocculation and 30 min of sedimentation. For feeding wetlands the influent was poured by hand into the inlet zone over a period of 20 min each day. The wetlands were initially supplied with a flow of 10 l/day from June to November 2004. The experiments were conducted from November onwards; at that time well grown plants were already present in the containers.

Experiments were carried out during two periods, the first from November 2004 to January 2005 and the second from February to March 2005. In these two periods the same different operational conditions were applied to both wetlands in order to evaluate the effectiveness of the wetlands under a range of hydraulic retention time and organic surface loading rate (Table 1). The nominal hydraulic retention time (water residence time) of both periods was set from previous studies with similar experimental wetlands (Caselles-Osorio and García 2006; Caselles-Osorio et al. 2007). The daily flow was calculated from the nominal hydraulic retention time and the theoretical volume of the experimental wetlands. The organic surface loading rates resulted from the applied flow and the organic matter concentrations. These loading rates were similar to those frequently observed in full-scale horizontal SSF CWs. Two to three influent and effluent grab samples were taken weekly and analysed for turbidity, COD, ammonium and sulphate using standard methods described in APHA-AWWA-WPCF (2001). All analyses were performed inmediately after sampling.

2.3 Dynamic Model Development

A new mechanistic and dynamic model of C and N transformations in horizontal SSF CWs has been developed that reflects the competition between bacteria and plants for nutrient uptake on the one

Table 1 Mean operational conditions of both identical experimental horizontal subsurface flow constructed wetlands (fed with settled (S) and previously processed wastewater with a physico-chemical (PC) treatment) in the two periods of study

	Period-Wastewater type			
	1st-S	1st-PC	2nd-S	2nd-PC
Date	Nov–Jan	Nov–Jan	Feb–Mar	Feb–Mar
Flow, l/day	20	20	30	30
Nominal hydraulic retention time, day	3	3	2	2
Surface loading rate, g COD/m^2.day[a]	12.6 (3.2)	6.4 (2.1)	19.7 (5.7)	10.4 (5.1)

In brackets the standard deviations of the averages

[a] Based on field measurements

hand, and the competition between different microbial groups for substrates and electron acceptors on the other hand. A detailed description of the model can be found in Rousseau (2005). For biochemical transformations of C and N compounds, aerobic and anoxic reactions have been considered [mainly following the recommendations given in the Activated Sludge Models, Henze et al. (2000)], as well as anaerobic reactions (following the model proposed by Kalyuzhnyi and Federovich (1998) on the competition of sulphate reducing and methanogenic bacteria). Bacterial groups considered include aerobic heterotrophs, fermenters, sulphate reducers, methanogens, nitrifiers and sulphide oxidisers. Microbial reactions are represented by standard Monod equations with switching functions. Note that biofilms growing in SSF CWs are very thin (Shipin et al. 2005) and therefore diffusion limitations in the biofilm are neglected for reasons of simplicity.

The inclusion of particulate substances in the model allows assessment of the clogging process. It is assumed that suspended solids are mostly removed near the inlet of the horizontal SSF CWs (at less than one-third of the total length) under normal operating conditions. Only at higher flow rates, wash-out of solids has been foreseen to the subsequent parts of wetlands (tanks in the model presented here) or via the effluent. Wash-out has been modeled proportional to the flow rate and to the solids content. The flow rate depends on the influent flow rate, rain inputs and evapotranspiration outputs. Clogging is then evaluated in the model by means of pore volume reduction. To this end, the masses of all inorganic and organic solids (including bacteria) are added up and converted to a volume by means of their estimated density and water content, as described in Blazejewski and Murat-Blazejewska (1997). Note that macrophyte root biomass is not considered in pore volume reduction.

Data obtained with the experimental wetlands were used to calibrate and validate the model. The model requires 28 input variables the values of which should be provided by the modeller: 23 regarding the influent (flow, COD, ammonium, etc) and 5 related to environmental factors (day length, air temperature, water temperature, rain and season). The description of the processes in the horizontal SSF CWs requires a total number of 110 stoechiometric, kinetic and other parameters. Simulations were carried out using WEST™ modelling and simulation software (Hemmis NV, Kortrijk, Belgium). The hydraulic submodel was based on series as well as parallel branched complete stirred tanks of equal volume.

The kinetic processes (and their parameter values) considered in the model used in the present study are similar to the kinetic processes employed to describe the reactions of the activated sludge and those that occur in anaerobic reactors. However, the environment present in horizontal SSF CWs differs greatly from that found in a conventional mechanical treatment plant. Wetland's environment is highly complex and heterogeneous in its physical, chemical and biological properties. Therefore, the extrapolation of the kinetics and their parameter values of conventional plants to horizontal SSF CWs constitute a pragmatical approach until the development of experimental methods for estimating kinetic parameters in horizontal SSF CWs.

3 Results and Discussion

3.1 Experimental Wetland's Removal Efficiency

The general performance characteristics of the two experimental wetlands during the two studied periods are given in Table 2. Details of the day-to-day performance may be found in Caselles-Osorio and García (2007). The physico-chemical treatment reduced the turbidity of the wastewater as well as its COD concentration (in that case approximately by half). During the first period the effluents of both wetlands were very clear and had a low turbidity, but in the second period the water became less transparent and showed a characteristic light-white color in the two wetlands. This increase in turbidity seemed to be related to two factors that may occur at the same time: precipitation of carbonates and formation of elemental sulfur from hydrogen sulphide (Caselles-Osorio and García 2007). These two factors in anaerobic environments are related to the presence of hydrogen sulphide, which is released through the activity of sulphate reducing bacteria (Wright 1999). The increase in turbidity linked to the presence of hydrogen sulphide is supported by the results of sulphate: during the first period the influent and effluent sulphate concentrations were identical in the two wetlands, while in the second period the concentrations were lower in the two effluents. This indicates

Table 2 Averages (standard deviations in brackets) of the water quality parameters in the influent (Inf.) and the effluent (Eff.), and of the percentage removal efficiencies (Rem.) of the two identical experimental horizontal subsurface flow constructed wetlands

Parameter	1st Period						2nd Period					
	Inf.	Eff.	Rem.	Inf.	Eff.	Rem.	Inf.	Eff.	Rem.	Inf.	Eff.	Rem.
	S	S	%	PC	PC	%	S	S	%	PC	PC	%
Turbidity, NTU	99 (40)	8.0 (15)	92	19 (11)	9.1 (19)	52	120 (46)	44 (27)	63	20 (14)	34 (43)	–
COD, mg/l	350 (87)	26 (13)	92	180 (59)	25 (12)	86	360 (110)	43 (20)	88	190 (94)	26 (12)	86
NH_4^+, mg N/l	32 (6.8)	2.3 (1.3)	93	28 (8.2)	1.3 (0.9)	95	36 (7.8)	12 (5.7)	67	34 (7.3)	13.4 (5.7)	62
SO_4^{2-}, mg/l	170 (33)	170 (67)	0	170 (29)	170 (63)	0	190 (43)	150 (75)	21	190 (36)	150 (85)	21

Calculations based between 26 to 39 measurements depending on the parameter.

that during the second period a certain degree of sulphate reduction occurred and as a result sulphide was formed.

COD removal was in all cases very high (around 90%). Effluent COD concentrations of the two wetlands were very similar in the first period. In the second period they were slightly higher in the wetland fed with PC (with a systematic pattern). This trend was related with the higher flow rate in the second period, which could have washed out a higher amount of retained particles than in the first period in the wetland fed with settled wastewater (Caselles-Osorio and García 2007). The higher flow rate in the second period had a significant effect on the removal efficiency of ammonium in both wetlands. As can be seen, the effluent ammonium concentrations of the two wetlands were clearly higher in the second period than in the first one. In fact, the removal efficiency decreased from approximately 90 to 70% in both systems. During the second period ammonia was on average slightly higher in the effluents of the wetland fed with PC, but without a systematic pattern.

3.2 Hydraulic Submodel

The initial hydraulic submodel was based on multiple tanks of equal volume in series. The results of a tracer test that was performed by Matamoros et al. (2007) with a non-reactive pharmaceutical product (clofibric acid) were employed to validate this hydraulic submodel. This tracer test was conducted in two wetlands identical to the two of the present study, and operating with the same flow rates. It was found that three to nine tanks-in-series mimicked quite well the tracer response curves. Also it was observed that as the number of tanks was increased (from three to nine), the adjustment of the simulated values to the observed ones did not improve. However, as the number of tanks increased, the simulation of the delay time (the time elapsed until de tracer reaches the outlet) was better. For this reason in the initial attemps of the reactive simulations it was agreed to run simulations with a hydraulic submodel with 9 tanks-in-series. With this hydraulic set up (and even with less tanks-in-series) the reactive simulations results for COD and ammonium were quite good, but effluent nitrate concentrations were far too high due to a lack of denitrification. This was due to the fact that it was considered that all the tanks were aerobic (in contact with the atmosphere and under the effect of the roots of the plants (that released oxygen)). Note that nitrate concentrations were nearly undetectable in a few measurements made on the effluents of both experimental wetlands, as is commonly observed in these systems (Tanner et al. 2002). With the nine tanks-in-series model, one simulation of 120 days (which was the experimentation time) took several hours on a Pentium 4 computer.

A combined solution for both above stated problems (high nitrate concentrations and computation time) was devised that basically consists of (1) less tanks and (2) series as well as parallel branched tanks. Figure 2 gives a schematic representation of the final hydraulic lay-out used for simulation, which was based on eight tanks. In this lay-out the splitters (split) allow to divide the flow into customisable fractions that are conveyed to aerobic (ae) as well as anaerobic (ana) complete stirred tanks. One could imagine the aerobic tanks to represent the upper layer of the wetlands, both in contact with the air and with the plant roots whilst the so-called anaerobic tanks represent the bottom layer below the rootzone. After each set of

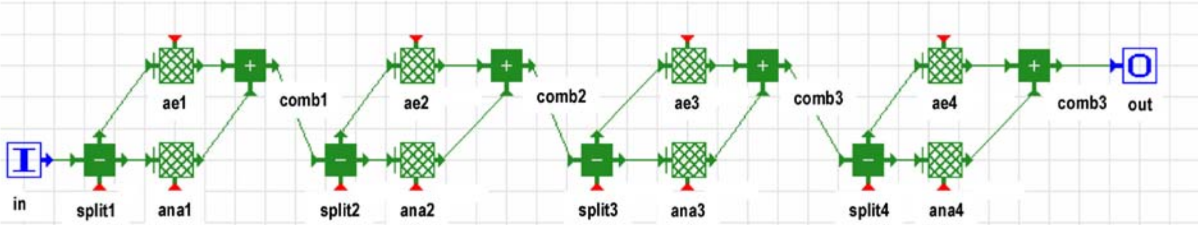


Fig. 2 Schematic representation of the hydraulic lay-out used for the simulations of the identical experimental horizontal subsurface flow constructed wetlands. *ae* aerobic tank, *ana* anaerobic tank, *split* splitter, *comb* combiner, *I* input, *O* output

tanks, a combiner (comb) mixes both water flows and then feeds them to the next splitter. Note that a similar lay-out has been used before by Benedetti et al. (2004) for the description of activated sludge wastewater treatment with imperfect mixing and anaerobic zones. Also note that with this lay-out the number of tanks fits in the range for which the hydraulic validation was conducted (three to nine).

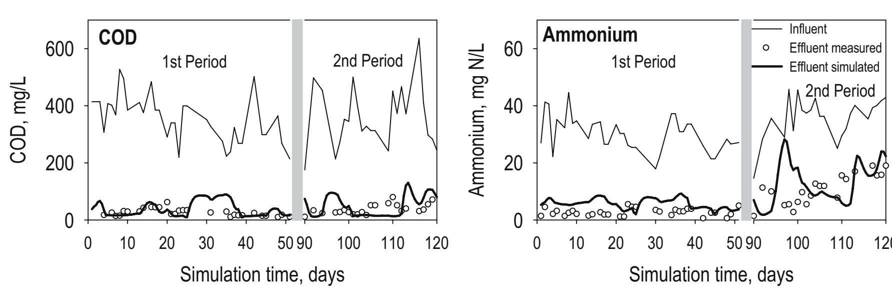
As several reports point out that the majority of settled substances can be found near the bottom of the horizontal SSF CWs (Kadlec and Knight 1996), the splitters were recoded in such way that the fraction of solids going to the ana (bottom) tanks could be adjusted. The volume ratios between the ae and ana tanks were determined via trial and error, as well as the flow split and particle split fractions. The volume ratio considered for ae tanks was 0.7 and for ana ones 0.3. The fraction of flow to ae tanks was 0.6 and to ana 0.4. The fraction of particles to ae tanks was of 0.4 and to ana 0.6.

3.3 Effluent COD and Ammonium Modelling

For details on the parameter values as well as on the COD fractionation, the reader is referred to the original description of the model (Rousseau 2005). Initial conditions were attained using average (constant) concentrations and a flow rate of 20 l/day (steady state simulation). Figure 3 compares simulated and measured effluent concentrations of COD and ammonium of the wetland fed with settled wastewater as example. Until day 90, a flow of 20 l/day of wastewater was applied whereas from day 90 onwards the flow was 30 l/day. Between day 50 and 90 there are however no data on the influent and effluent COD. This period was bridged in the simulation of all parameters by using a constant influent (data not shown). Measured and simulated effluent COD concentrations seem to be in good agreement, with some exceptions. The higher predicted concentrations between day 25 and day 35 seem to be caused by more extreme washout of solids to the subsequent tanks because of several violent rain storms that happened during this period. Note that the model considers for the water balance the rain inputs as well as the evapotranspiration outputs. Rain measurements and data necessary for evapotranspiration calculations were obtained from a nearby meteorological station located at less than 5 km from the wetlands location. At day 96, the peak seems to be an underestimation by the model of the capability of the wetland to deal with the increase in organic loading rate in the second period. Both the measured and simulated effluent data show that the higher surface organic loading rate applied during the second period had a very low influence on the COD removal efficiency.

The increase in flow rate from the first to the second period had a clear effect on the ammonium

Fig. 3 Simulated and measured COD and ammonium concentrations in the wetland fed with settled wastewater. Between days 50 and 90 there were no data on the influent and effluent COD



simulated effluent data, as it was also observed from the measured concentrations. During the first period, the model seems to underestimate ammonium removal by some 3 to 4 mg N/l. Increasing the ae tanks volume and/or the oxygen transfer rates only helped partially, and resulted in very high effluent nitrate concentrations, which were not observed in reality. When the flow rate was changed at day 90, the simulation shows a marked effluent peak which was not observed in reality. This seems to indicate that the experimental wetlands had a quicker adaptation capacity than what the model predicts. Indeed, Kadlec et al. (2005) have observed using isotopic techniques that ammonium is rapidly and completely sorbed in the inlet region of horizontal SSF CWs, and subsequently released in major part back to the water. Therefore the peak could be related with an underestimation of the ammonia sorption capacity.

3.4 Pore Volume Reduction Modelling

Pore volume reduction is a process that in the model depends on the growth of the bacterial groups considered, and on the retention of organic and inert solids coming from the wastewater and the decay of plants. Chemical precipitation and subsequent accumulation of the precipitated solids is another mechanism involved in pore volume reduction (Platzer and Mauch 1997), but for reasons of simplicity was not considered in the model. Figure 4 shows the simulated spatio-temporal distribution of the two most abundant bacterial groups (>50%), i.e. aerobic heterotrophs and fermenters, in the wetland fed with settled wastewater and in the tanks where they attained the higher concentrations. Note that in this figure bacterial concentrations are expressed in terms of COD. Aerobic heterotrophs logically have the highest densities in the ae tanks and near the inlet, where substrate is abundant, whereas the fermenting bacteria are more abundant in the ana tanks near the outlet. As can be seen, the higher loading rates in the second period resulted in increasing concentrations of both bacterial groups. All the other bacterial groups were in concentrations lower than 250 mg COD/l and therefore their contribution to pore volume reduction was less significant that in the case of aerobic heterotrophs and fermenters.

The simulated spatio-temporal evolution of porosity is shown in Fig. 5 for both experimental wetlands and in the tanks where it was attained the lower porosities with time. Note that field measurements of the porosity were not available. As can be seen, after 120 days the lowest porosities are mostly found in anaerobic tanks. This correlates well with the fact than more than 50% of the solids are routed to the bottom layers. However, in reality one mostly observes the sharpest decline near the inlet, and not around the middle of the bed length as it is observed for both wetlands. This indicates a problem with the wash-out settings. Adjusting these settings however resulted in higher nitrate effluent concentrations. Despite this problem, the simulation results clearly show that the pore reduction is significantly more important in the wetland fed with settled wastewater (S) than in the wetland fed the same wastewater but processed with the physico-chemical treatment (PC). If one takes into consideration the tanks with the highest decrease in porosity as example, according to the model in the wetland fed with S the porosity reduced in a 17% (from 0.35 to 0.29, in ana3), while in the wetland fed with PC in a 6% (from 0.35 to 0.33, in ana2). On the other hand, in agreement with the predictions of the model, field measurements indicated that the hydraulic conductivity decreased in the wetland fed S in a 20% in comparison to the SSF fed with PC (Caselles-Osorio and García 2007).

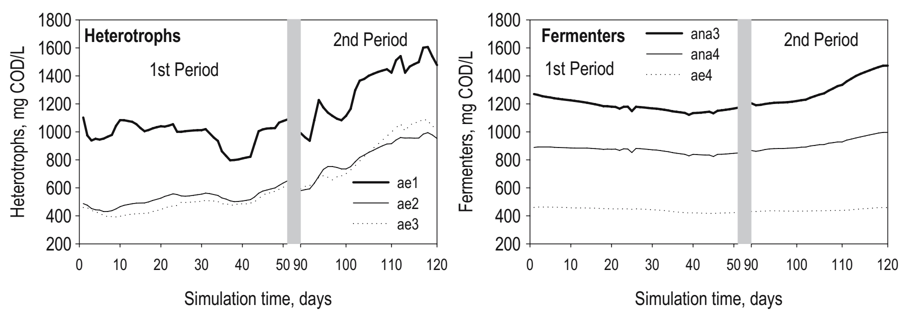


Fig. 4 Spatio-temporal distribution of the aerobic heterotrophs and fermenters in the wetland fed with settled wastewater. Only the results in tanks where the microorganisms attained the higher concentrations are shown. *ae* aerobic tank, *ana* anaerobic tank. The exact location of the tanks can be seen in Fig. 2

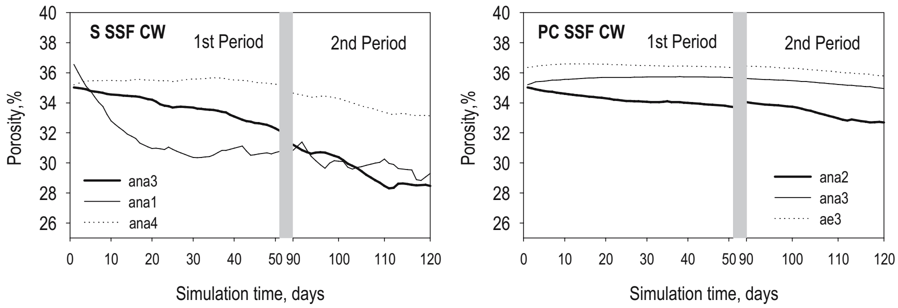


Fig. 5 Spatio-temporal distribution of the porosity in the two identical experimental horizontal subsurface flow constructed wetlands (SSF CWs). Only the results in tanks where the decrease in porosity was the highest are shown. One wetland was fed with settled wastewater (S) and while the other the same wastewater but processed with a physico-chemical treatment (PC). *ae* aerobic tank, *ana* anaerobic tank. The exact location of the tanks can be seen in Fig. 2

Pore volume reduction in ana tanks is mainly due to organic particles accumulation as can be seen in Fig. 6. In this figure, the first tanks of both wetlands have been considered as example, and the particulate components have been sorted in three categories: organic particles (which include slowly and very slowly biodegradable substrates), inert particles (non-biodegradable) and bacteria biomass (the sum of biomass of all the considered bacterial groups). In opposition to ana tanks, in ae tanks pore volume reduction is mostly caused to bacteria growth. Differences in the relative accumulation of particulate components between ae and ana tanks seem to be related with the fact that hydrolysis is considered in the model more efficient in aerobic conditions. Also note that a higher fraction of particles (0.6) goes to the ana tanks.

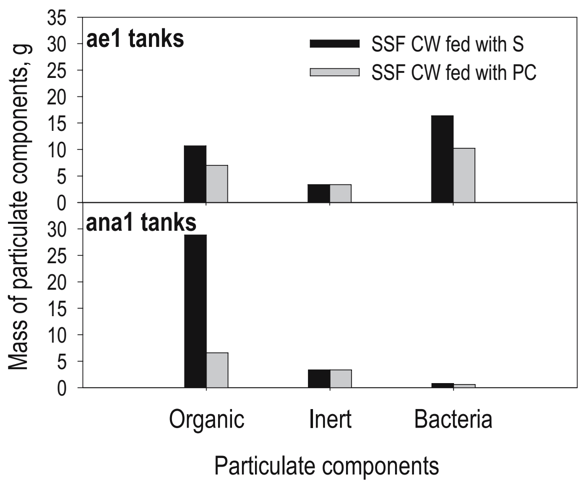


Fig. 6 Simulated masses of all particulate components in the first aerobic tanks (*ae*) and anaerobic (*ana*) tanks in the identical experimental horizontal subsurface flow constructed wetlands after 120 days. The exact location of the tanks can be seen in Fig. 2

4 Conclusions

The results of this study indicate that the application of a physico-chemical treatment prior to SSF CWs does not improve their efficiency significantly under the experimental conditions tested. Nevertheless, the dynamic model has shown that the physicho-chemical treatment reduces the risk of clogging. In fact, after 120 days of operation, in some regions of the wetland fed with settled wastewater the porosity decreased by 17%, whereas in the wetland fed with the same wastewater but previously processed whith a physico-chemical only decreased by 6%.

The use of a prior physico-chemical treatment is therefore a good alternative for avoiding an anticipated clogging of SSF CWs. And this is specially true if the disadvantages of the physico-chemical treatments (extra cost for chemicals and more sludge) are, in addition, compensated with for example phosphorus removal. Another conclusion of this work is that clogging prediction tools have to incorporate the retained particles and the biofilm growth because both are important processes for pore volume reduction. Models based exclusively on retained particles can underestimate the rate of pore volume reduction.

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