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NOVEL HYBRID MODELS FOR THE DESIGN OF UPFLOW ANAEROBIC FILTERS SEPARATED IN PHASES

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Abstract: In this paper, three novelties are presented, the type of reactor, design models, and design experimental factors. The upflow anaerobic filters are separated into two and three phases identified as DI-FAFS and TRI-FAFS (abbreviation in Spanish). The second novelty consists of modifying and combining the traditional models for trickling filter to create hybrid models. The third novelty is based on the combination of three experimental factors: the volumetric organic load has been set at 2.25, 3.45 and 4.64 kg COD m^{-3} d^{-1}, the temperature at 20, 27 and 34 °C, the ratio of depths in reactors. The wastewater is the landfill leachate. The conceptual model is based on equations deduced from a mass balance under stationary conditions $dS/dt = 0$ and advective $dS/dZ \neq 0$; formulating eight equations applicable to the DI-FAFS and TRI-FAFS reactors, resulting a $R^2_{adjusted}$ greater than 0.7.

Keywords: Design, modelling, filters separated in two phases, filters separated in three phases, validation.

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INTRODUCTION

In many cities in developing countries, the main disposal methods for municipal solid waste are the open dumping and sanitary landfill. Uncontrolled dump sites are smoke with a lot of leachate generation causing severe environmental pollution (Chandrappa and Das, 2012). The composition of landfill leachate varies greatly, depending on waste quantity and composition, the decomposition rate and age of the waste, and landfilling technology (Kängsepp *et al*., 2008; Wojciechowska and Obarska-Pempkowiak, 2008). The strong raw leachate from the new landfill has a COD of 14 000 mg/l, a BOD/COD ratio of 0.7 (Henry *et al*., 1987). The partially stabilized leachate from the older landfill has a COD varying between 3000 and 3750 mg/l, a BOD/COD ratio approximated to 0.3. This creates the need to develop leachate treatment systems mainly of the biological type like wetland (Wojciechowska and Obarska-Pempkowiak, 2008) or biological filters (Kängsepp *et al*., 2008); which attempt to remove the pollutants to reduce the environmental load and protect the surface water and groundwater.

Regarding to the design models for wastewater treatment systems with biofilm attached, these are used to estimate the performance in the organic matter removal. The most important design models are the following: 1) the rotary biological contactors (Chesner *et al*., 2012); 2) the trickling filters (Velz, 1948; Stack, 1957; Schulze, 1960; Germain, 1966; Yang *et al*., 2011); 3) the activated biofilters (Arora and Umphres, 1987); 4) the submerged filters (Marquez and Navas, 2002); and 5) the anoxic and anaerobic biofilm systems (Maldonado *et al*., 2017).

Among the biofilters, one of the most used for the wastewater treatment is the trickling filter (Luo *et al*., 2014). For this reason, there is a lack of design equations for anaerobic biological filters. From Schulze (1960), a type of mathematical modeling has been carried out under two conditions using the same mathematical structure for trickling filter: advective and non-stationary conditions. Germain (1966) modifies the equation proposed by Schulze (1960) to estimate the performance of the trickling filter under advective and non-stationary conditions by making a generalization of the power of the hydraulic load included within the contact time. Yang *et al*. (2011) perform a mathematical modeling on a derivative of second order with respect to the substrate, to the horizontal direction of entry of the substrate into the biofilm of biofilter. Arora and Umprhes (1985) propose a mathematical structure for a variant of the trickling filters called activated biofilter modifying the model proposed by Phepls (1944) and Velz (1948) expressing the contact time as a function of the volumetric organic load (VOL). Wang *et al*. (2015) proposes a regression model for the removal of ammonium in a biofilter based on the addition of functional nitrogen genes to contribute to nitrification.

In this investigation, upflow anaerobic filters separated in two and three phases have been selected to treat the leachate generated from "El Guayabal" sanitary landfill located in the coordinates: N: 08°00'48,83" E: 72°30'44,14", Colombia. This sanitary landfill has been operated from 15 years ago; receiving an average of solid waste generated of 850 t d^{-1} from 20 Municipalities (Perozo, 2015).

In this study, three novelties are presented, the type of reactor, design models, and design experimental factors. With respect the reactor type, it is tested the assemblage of upflow anaerobic filters separated in two and three phases identified as DI-FAFS and TRI-FAFS (abbreviation in Spanish). The support media are plastic tubes on which the microorganisms are attached. The most studied biological filter is the trickling filter, which is an aerobic biological filter (Water Pollution Control Federation, 1988). There is a lack of studies about of anaerobic filter performance. The main advantage of anaerobic filter is that reproduce the natural condition of the soil.

The second novelty consists to create hybrid models by combining the equations proposed by trickling filters to explain the performance of DI-FAFS and TRI-FAFS reactors. The hybrid models are based on the superposition of effects of variables contained in the following equations: 1) (Germain, 1966; Albertson and Davies, 1984), 2) (Van't Hoff, 1884; Shulze, 1960; Germain, 1966; Albertson and Davies, 1984), and 3) (Shulze, 1960; Germain, 1966; Albertson and Davies, 1984).

The third novelty is based on the combination of three experimental factors: the volumetric organic load (VOL) has been set at 2.25, 3.45 and 4.64 kg COD m^{-3} d^{-1}, the temperature at 20, 27 and 34 ° C, the ratio of depths in reactors. In this study, it is considered the substrate concentration in the control volume of the biological filter through the VOL. Likewise; the residual liquid temperature is tested in a typical range that occurs in a tropical country. The maximum temperature of 34°C has not been evaluated under experimental conditions in biological filter studies, however, this is a common temperature value in tropical countries.

MATERIALS AND METHODS

This research has been developed by the following activities:

Design and construction of the DI-FAFS and TRI-FAFS reactors

In the DI-FAFS reactor, Phase 1 with a volume V_1 and Phase 2 with a volume V_2; the total volume of the reactor is V_{total} = 3.8 L. The sum of the depths of the phases is of 1.20 m. The heights of each series were varied based on the percentage ratio Phase1/Phase2 in DI-FAFS and TRI-FAFS: 80%/20%; 50% 50% and 20%/80% of the total depth (**Fig. 1**).

Acclimatization of the DI-FAFS and TRI-FAFS reactors

The adaptation of the previously adhered biofilm was achieved by supplying a volumetric mixture of municipal wastewater to the reactors of: Series 1 [low]: 1700 mg/l; Series 2 [medium]: 2600 mg/l and Series 3 [high]: 3500 mg/l.

Start-up and operation of the DI-FAFS reactors

In the experimental design, it was established that the temperatures of the substrate during the operation of the reactors would be 20°C (low), 27°C (medium) and 34°C (high). The treatment of the leachates in the DI-FAFS and TRI-FAFS reactors was started with a constant temperature of 20ºC in all the series, but with variable VOLs in each of them, as follows: Series 1 VOL_{low} = 2.25 kg/m^{3}d, Series 2 VOL_{medium} = 3.45 kg/m^{3}d and Series 3 VOL_{high} = 4.64 kg/m^{3}d.

Development of mathematical models for the DI-FAFS and TRI-FAFS reactors

The calibration and validation of the models were carried out to determine the best models of statistical adjustment that allows to predict the concentration of organic matter (COD) in the effluents of the DI-FAFS and TRI-FAFS.

THEORETICAL FORMULATIONS

The formulation of hybrid models is a combining the conceptual and empirical approaches based on the mass balance and the derivation of the parametric equations of the components included in the balance from the experimental data.

The mass balance corresponding to the organic matter contained in the liquid volume into the upflow anaerobic filter can be observed in the **Eq. 1**:

$$\frac{\partial \bar{S}}{\partial t} dV = QS - Q\left(S + \frac{\partial S}{\partial Z} dZ\right) + dVr_s \qquad (1)$$

where $(\partial \bar{S}/\partial t)\ dV$: rate of the substrate accumulation with the time into of the system limits, Q: volumetric ratio, m³/d, QS: quantity of substrate entering to the element volume, $Q(S + \partial S/\partial Z)\ dZ)$: quantity of substrate leaving the element volume, r_s: rate of organic matter flow into the biological layer, dVr_s: flow of the substrate from the elementary volume to the interior of the biological film, w: section width, m, Z: depth of the filter, m. Assuming stationary conditions $(\partial \bar{S})\ /\ (\partial t\ =\ 0)$, **Eq. 2** can be simplified to derive at:

$$Q\frac{\partial S}{\partial Z} dZ = dVr_s \qquad (2)$$

Atkinson and Davies (1974) have adapted the Equation of Monod, generated under experimental conditions showing that the substrates or the nutrients to the growth of the microorganisms are presents in limited quantities. The mathematical solution of **Eq. 2** is obtained by integrating between the limits S_e and S_i; 0 and D, which is represented by **Eq. 3**:

$$\frac{S_e}{S_i} = exp[-(fhk_o) * wD/Q] \qquad (3)$$

where S_e is concentration of organic matter in the effluent, mg/l; S_i is Concentration of the organic matter in the affluent, mg/l after mixing the residual water to be treated, mg/l, f is factor of the proportionality, h; thickness of the biological film, m; k_0: maximun rate of reaction, d; mean concentration of the substrate expressed as BOD_5 or COD in an elemental volume within a liquid mass, mg/l, w: thickness of the biological layer.

Equation 4 is equivalent to the reaction kinetics of a substrate such as BOD formulated according to a first-order derivative proposed by (Phelps, 1944):

$$\frac{dL_t}{dt} = -k\ L_t \qquad (4)$$

where L_t is the quantity of BOD of the first phase remaining in the water at time $t(d)$, and k is the reaction constant $(1/d)$. The integration of **Eq. 4** between S_e and S_i, t = 0 and t; results in **Eq. 5**:

$$\frac{S_e}{S_i} = e^{-kt} \qquad (5)$$

Combining **Eq. 3** with **Eq. 5**, the parameter k can be explained in terms of f, h and k_0 (**Eq. 5**); at the same manner t is equal to wD/Q (**Eq. 3**); transforming a mathematical expression dependent on principles of the kinetics by a geometric relations associated with the depth of the filter (D) as it has been proposed by Howland (1958).

$$k = fhk_0 \qquad (6)$$

$$t = wD/Q \qquad (7)$$

Equations 6–7 are modified to incorporate the concentration of the substrate in the inflow S_i and express the wastewater flow (Q) in terms of the volumetric organic load as it has been proposed by (Yu *et al*., 1998) considered in this study as a variable that

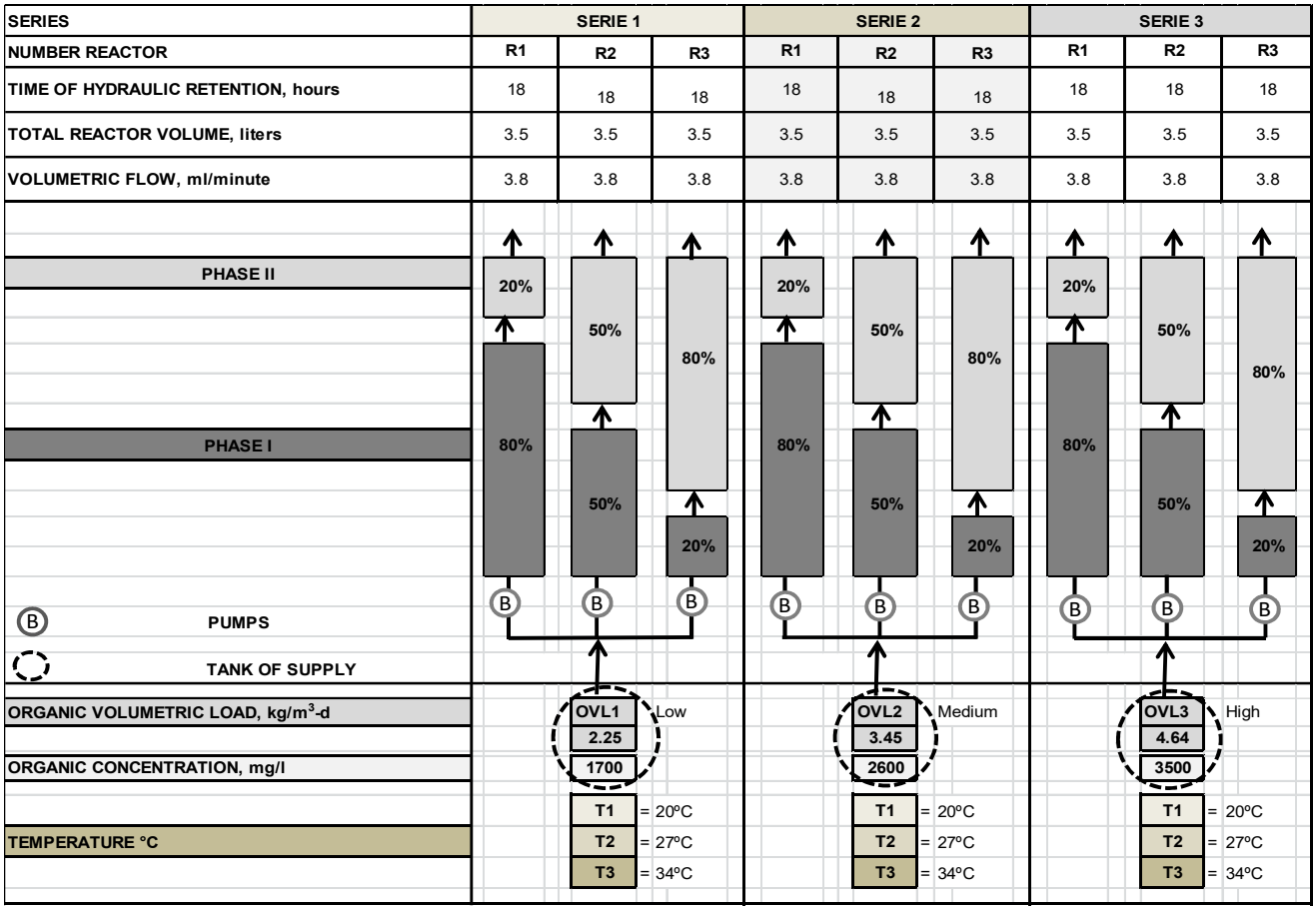

Fig. 1 General set of the TRI-FAFS reactors.

affects the description of the general kinetics of the DI-FAFS and TRI-FAFS reactors, resulting in Equations 8 and 9:

$$k = fhk_0 S_i \quad (8)$$

$$t = wD/QS_i; \text{ Being } t = COV^{-n} \quad (9)$$

The **Eqs. 8–9** have been modified to create empirical mathematical expressions for an upflow anaerobic filter with a plastic support medium, giving as a result the equations indicated in the **Table 1**. The Equations 10 and 14 are based on the equations proposed by a combination of two models for estimating the constant of treatability, which include in their original equations the following variables: Model proposed by Germain (1966): (1) filter depth, (2) wastewater flow, and (3) Specific surface. Model proposed by Albertson and Davies (1984): ratio of two different depths.

The **Eqs. 11–12** and **Eqs. 15–16** are a combination of four models. These models are the two models described by Equations 12 to 16 superimposed with the model of Van't Hoff (1884): temperature of liquid, Shulze (1960): constant of treatability for liquid temperature of 20°C. The equations have been proposed by (Van't Hoff, 1884; Shulze, 1960; Germain, 1966; Albertson and Davies, 1984). The **Eqs. 13–17** refer to the equations obtained by (Shulze, 1960; Germain, 1966; Albertson and Davies, 1984). The novelties introduced by these proposed equations are: 1) the substitution of wastewater flow by the volumetric organic load (VOL), taking into account the wastewater concentration and flow in the affluent, and the reactor volume 2) Introduction of hydraulic and biological variables to contribute to increase the explanation of the dependent variable represented by the concentration of organic matter in the effluent (S_e). The technique of applied mathematical modeling is forward, which consists of adding variables in each new model to improve the fit of the model, continuing the process until there is a moment that adding another term does not improve the model considerably (Gutierrez and De la Vara, 2004).

RESULTS

Comparison of DI-FAFS and TRI-FAFS reactors with other reactors

By comparing of the experimental conditions and performance of the biological filters constructed on a

Table 1. Equations proposed for the design of the upflow anaerobic filter separated in two (DI-FAFS) and three phases (TRI-FAFS).

N°	Equation	Parameters
	DI-FAFS Reactor	
10	$S_e = S_i exp\left[-k\left(\frac{D_1}{D_2}\right)^x (cov)^{-n}\right]$	k, x, n
11	$S_e = S_i exp\left[-k(A)^T\left(\frac{D_1}{D_2}\right)^x (cov)^{-n}\right]$	k, A, x, n
12	$S_e = S_i exp\left[-k(A)^{T-20}\left(\frac{D_1}{D_2}\right)^x (cov)^{-n}\right]$	k, A, x, n
13	$S_e = S_i exp\left[-k\left(\frac{D_1}{D_2}\right)^x (S_{a1})^m (S_{a2})^p (cov)^{-n}\right]$	k, x, m, p, n
	TRI-FAFS Reactor	
14	$S_e = S_i exp\left[-k\left(\frac{D_1}{D_2}\right)^x \left(\frac{D_2}{D_3}\right)^y (cov)^{-n}\right]$	k, x, y, n
15	$S_e = S_i exp\left[-k(A)^T\left(\frac{D_1}{D_2}\right)^x \left(\frac{D_2}{D_3}\right)^y (cov)^{-n}\right]$	k, A, x, y, n
16	$S_e = S_i exp\left[-k(A)^{T-20}\left(\frac{D_1}{D_2}\right)^x \left(\frac{D_2}{D_3}\right)^y (cov)^{-n}\right]$	k, A, x, y, n
17	$S_e = S_i exp\left[-k\left(\frac{D_1}{D_2}\right)^x \left(\frac{D_2}{D_3}\right)^y (S_{a1})^m (S_{a2})^p (cov)^{-n}\right]$	k, x, y, m, p, n

laboratory scale, whose contact media are constituted by natural and synthetic media reported by different researchers with those belonging to this study. The experimental conditions that have been found for the design of the DI-FAFS and TRI-FAFS (**Fig. 1**) anaerobic reactors of this study are different to those of the percolating filters and anaerobic reactors as follows:

Shulze (1960) provided VOLs to trickling filters as COD varying among 2.25, 3.45 and 4.64 kg m^3/d, both for the treatment of organic compounds from wastewater expressed as BOD_5 and COD, respectively. The filters were considered shallow between 0.9 and 1.8 m. The biofilter support medium was synthetic in this study and Yu *et al*. (1998); of metal mesh in Schulze (1960). In general, the removal efficiency of the organic matter can be attributed to the characteristics of the medium being remarkable that Schulze included 7 layers based on the medium of metal support for the biofilm and ratio of recirculation of effluent and sludge between 0 and 3; obtaining efficiencies between 58 and 73%. Yu *et al*. (1998) incorporated a polyurethane foam with high specific surface area reported equal to 1560 m^2/m^3 obtaining efficiencies of 92%; while in this investigation two and three stages were included with a specific surface area of 476.35 m^2 m^{-3}; obtaining efficiencies in the DI-FAFS reactor between 27 and 72.86%, while in the TRI-FAFS (**Fig. 1**) reactor between 84 and 95%, which could suppose that the upflow anaerobic filter ensures a biofilm with a better segregation of the microorganisms to the treatment processes according to the nature of the residual liquid remaining of the subsequent phases, providing a better performance than the uniform multilayer trickling filters (Schulze, 1957; 1960) and the single-phase trickling filters with a synthetic material corresponding to high specific surface area (Yu *et al*., 1998).

The separated phases of the trickling filters of synthetic support medium and organic affluent up to two stages have been applied by Germain (1966) coinciding with this investigation; it could also be considered that it fixed a higher hydrodynamic condition, based on reaction units of depth equal to 6.5 m in each stage; including a ratio of recirculation of the effluent and sludge between 1 and 2; as well as a range of VOLs that vary between 0.16 and 0.96 kg m^3/d. Germain (1966) showed that recirculation did not have a statistically significant effect on the removal of BOD_5 for filters deeper than 3 ft; and, therefore on the treatability parameter *k*, so that it has no impact on the coefficients of **Eqs. 10–17**; being equivalent to the scheme proposed by the present study. When comparing the results of the DI-FAFS and TRI-FAFS (**Fig. 1**) treatment systems corresponding to ranges of organic matter removal efficiencies between 27 and 72.86%; 84 and 95%; these systems would be in an equivalent capacity with their separate phases of replacing the recirculation line of the effluent and sludge to the reaction unit as the multiphase-multilayer system executed by Germain (1966).

Wang *et al*. (2015) have applied residual water based on ammonium, NH_4^+-N to a charge in biofilters constituted by polyurethane sponge; obtaining removal efficiencies of NH_4^+ between 67.3 and 92.7% and COD between 97.7 and 99.3%. The study by Wang *et al*. (2015) includes four layers; which have given comparable results with the TRI-FAFS (**Fig. 1**) reactor being the level of removal of the COD reached by including this third phase estimated at 84-95%; this is attributed to the fact that the number of layers or phases significantly influences the efficiency of elimination of organic matter from biological filters.

Schulze (1960); Yu *et al*; (1998) have reported variation in the temperature of the residual liquid within the experimental design as one of the control factors like the present study as follows: 14.1–22.1°C; 16–19°C; 15.1–27°C; 35°C; 20, 27 and 34°C; respectively. According to Metcalf and Eddy (1996) in an anaerobic digestion process of organic waste the optimum temperature ranges are mesophilic (30 to 38°C) and thermophilic (49 to 57°C). In general, the results that have been found are comparable in terms of

this characteristic of the residual liquid to be treated; specifically for a temperature of 20°C. This temperature is common for all the referenced studies; so this is a fact that directly influences the treatability constant that reflects the activity of microorganisms as established by Van't Hoff Arrhenius; (1884). In an extreme case, the maximum temperature is of approximately 35°C (Metcalf and Eddy, 1996); which is tested only by Yu *et al*. (1998) and this study; confirming that there is dependence on the temperature of the constants of the speed of the biological reaction; which influences not only the metabolic activities of the microbial population; but also has effects on the rate of gas transfer as methane (CH_4) and carbon dioxide (CO_2) occurred in an anaerobic digestion process as referred by Metcalf and Eddy (1996); hence, this variable is a significant factor in **Eqs. 11–12** and **Eqs. 15–16** proposed in this study, which are derived on the basis of the relationships proposed by Van't Hoff Arrhenius (1884) and Albertson (1984). Howland (1958) made an inclusion of the temperature factor as has been done in Equations (12) and (16) finding a coefficient of temperature A of 1.035, which is approximated to the one reported for the statistical adjustment of the DI-FAFS and TRI-FAFS (**Fig. 1**) reactors. The mean values of coefficient A are: DI-FAFS: 1.04797; TRI-FAFS (**Fig. 1**): 1.01381.

Comparison of experimental conditions for the analysis of the process of an upflow anaerobic filter separated in three phases (TRI-FAFS)

As a sample, the results of the COD and removal rate for the analysis of the process of an upflow anaerobic filter separated in three phases (TRI-FAFS) are presented for temperatures of 20°C in **Fig. 2**. In the **Figs. 2a–2c**, the COD in the affluent is constant of 1700 mg/l. The figures show two types of trends. During the first period, which comprise to 40 days, the COD in the effluent to the TRI-FAFS reactor is decreasing from 1400 and 1000 mg/l. During the second period, comprising between 40 and 80 days, the COD in the effluent to the TRI-FAFS reactor is decreasing from 1000 to 300 mg/l. The removal rate is increased from 40% to 90%. In general, for 20°C and the three concentrations in the affluent, the CODs vary between 0 and 500 mg/l and the removal rates vary between 82 and 88%.

Comparison of the statistic adjustment of the models for the design of upflow anaerobic filter separated in two and three phases

The results of the parameters included in the equations proposed for the design of the upflow anaerobic filters separated in two and three phases are shown in the **Tables 2**–**3**, which contain the mean, minimum and maximum values of the parameters in the **Eqs. 10–13**. As a sample, the parameters of the Equation 12 are presented: k: 0.61333, 0.19135, 1.03531; A: 1.05137, 1.0288, 1.07393; x: –0.0626, –0.18146, 0.05607; n: 0.13010, –0.35332, 0.61352. By comparing the results of mean values of the parameters from **Eqs. 9–13**, k varies between 0.037 and 0.8363; x is a negative value, varying between –0.0626 and -0.3313; A between 1.03783 and 1.05137; n between -0.1107 and 0.13010. These ranges of values are close, which indicate that the contribution of each parameter has a similar weight to explain the value of response variable represented by Se or the COD remaining. In general, the standard error trends to be below of mean value of the parameters.

By comparing the value of the adjustment statistics of the data to the mathematical models 10 to 13 for the design of the upflow anaerobic filter separated in two phases (**Table 4**), the following aspects can be observed: a) the number of independent variables into each equation proposed varies between 3 and 5, b) the number of data in the calibration stage is 18, c) the number of data in the validation stage is 9, d) The determination coefficient indicates that between 0.68 and 0.89 of the variation observed in S_e is explained by the **Eqs. 10–13** with respect to the total variability, e) the coefficient of determination adjusted ($R^2_{adjusted}$) is the variable of preference in contrast with the determination coefficient (R^2) due to that this allows to measure if each new term added to the equation contributes to the explanation of the response variable, S_e. The terms in the **Eqs. 10–13** are increased from 3 to 5 independent variables. In this case, $R^2_{adjusted}$ is increased from 0.63 to 0.87 by increasing the terms from **Eqs. 10–12,** however this coefficient decreases by increasing to 5 terms in **Eq. 13**; meaning that the terms associated to the S_{a1}: surface area of the filter in phase 1 in m² and S_{a2}: surface area of the filter in phase 2 in m² do not contribute to the explanation of S_e as this is observed by adding the residual liquid temperature in the affluent in °C in **Eqs. 11–13**. The standard error of estimation and the mean absolute error give as a result the values of 211 and 141 in the **Eq. 12** close to the minimum in **Eq. 11**. The residues are smaller by comparing with the rest of equations. The Durbin-Watson statistic (d) estimated is into the range of the limits to the test with d_L and d_V selected to a significant level $\alpha = 0.05$; finding that there is not decision regarding to if the autocorrelation is occurring between consecutive residues. The **Eq. 12** is the model that best fits to the data; resulting in a combination of independent variables that estimate S_e values from the observations derived from the experiment.

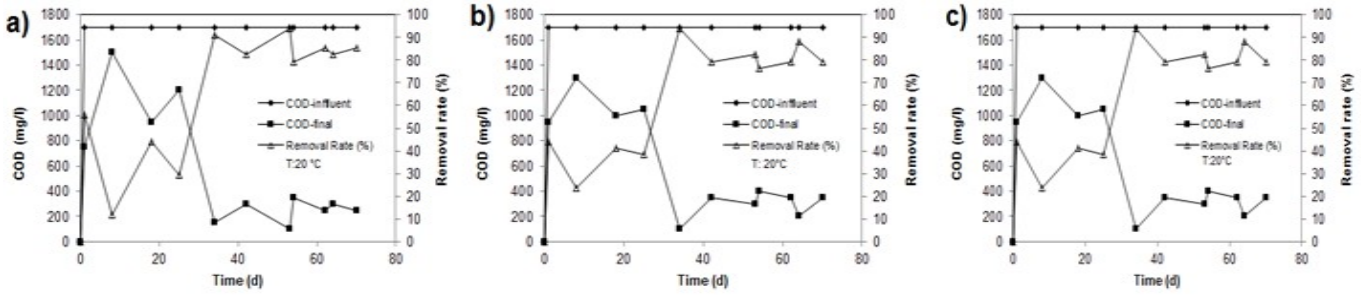

Figure 2. Chemical Oxygen Demand (COD) and removal rate for the analysis of the process of an upflow anaerobic filter separated in three phases (TRI-FAFS) under liquid residual temperature of 20°C.

Table 2. Parameters of equations proposed to the design of the upflow anaerobic filter separated in two phases.

Equation N°	Parameters	Mean	Standard Error	Mínimum	Maximum
10	K	0.8363	0.39664	−0.00913	1.68175
	X	−0.2146	0.07236	−0.36894	-0.06045
	N	0.0980	0.34536	−0.63804	0.83426
11	K	0.34157	0.12811	0.0667	0.61634
	A	1.03783	0.01037	1.0155	1.06009
	X	-0.1267	0.05094	−0.2359	-0.01746
	N	0.12361	0.20767	−0.3218	0.56902
12	K	0.61333	0.19674	0.19135	1.03531
	A	1.05137	0.01052	1.0288	1.07393
	X	−0.0626	0.05537	−0.18146	0.05607
	n	0.13010	0.22539	−0.35332	0.61352
13	k	0.0376	0.0894	−0.1555	0.2308
	x	−0.3313	0.2378	−0.8452	0.1825
	m	−0.2601	0.4259	−1.1804	0.6602
	p	−0.5626	0.4576	−1.5512	0.4260
	n	−0.1107	0.3102	−0.7809	0.5595

Table 3. Statistics of adjustment of the equations proposed for the design of the upflow anaerobic filter separated in two phases.

Eq.	p	n_1	n_2	R^2	$R^2_{adj.}$	SEE	MAE	d	d_L	d_V	RAL
(10)	3	18	9	0.68	0.63	327	246	1.49	1.1	1.5	0.19
(11)	4	18	9	0.89	0.86	193	124	1.89	0.9	1.7	0.03
(12)	4	18	9	0.89	0.87	211	141	1.55	0.9	1.7	0.21
(13)	5	18	9	0.71	0.62	281	188	1.87	0.9	1.8	-0.11

p: number of independent variables, n_1: number of data in the calibration stage, n_2: number odf data in the validation stage, R^2: determination coefficient, R^2adj.: coefficient of determination adjusted, SEE: Standard Error of Estimation, MAE: mean absolut error, d: statistic of Durbin-Watson, RAL: residual autocorrelation in lag 1

The adjustment statistics of the data to the mathematical **Eqs. 14–17** for the design the upflow anaerobic filter separated in three phases can be observed in **Table 5**. The terms in the **Eqs. 14–17** are increased from 4 to 6 independent variables. In this case, $R^2_{adjusted}$ is increased from 0.78 to 0.90 by increasing the terms from Equations 14 to 16; however this coefficient decreases by increasing to 6 terms in **Eq. 17**, meaning that the terms associated to the S_{a1}: surface area of the filter in phase 1 in m^2 and S_{a2}: surface area of the filter in phase 2 in m^2 does not contribute to the explanation of S_e as this is observed by adding the residual liquid temperature in the affluent in °C in **Eqs. 15–16**. The standard error of estimation and the mean absolute error give as a result the values of 42.4 and 30.30 in the **Eq. 16** being the minimum regarding to the rest of equations. The Durbin-Watson statistic (d) estimated is into the range of the limits to the test with d_L and d_V selected to a significant level α = 0.05; finding that there is not decision related to if the autocorrelation is occurring between consecutive residues. The **Eq. 16** is the model that best fits to the data; resulting in a combination of independent variables that estimate S_e values from the observations derived from the experiment.

Table 4. Parameters of equations proposed to the design of the upflow anaerobic filter separated in three phases.

Equation N°	Parameters	Mean	Standard Error	Minimum	Maximum
(14)	K	5.6316	3.15813	-1.14194	12.4051
	X	0.1237	0.12942	-0.15386	0.40130
	Y	0.3296	0.24754	-0.20128	0.86056
	N	0.1498	0.08958	-0.04227	0.34200
(15)	K	2.35958	1.02244	0.150735	4.56843
	A	1.01855	0.00695	1.00353	1.03358
	X	0.06773	0.09419	-0.1357	0.27122
	Y	0.20337	0.18790	-0.2025	0.60932
	N	0.06117	0.07350	-0.0976	0.21996
(16)	K	7.12703	2.6687	1.36164	12.8924
	A	1.01733	0.00348	1.0098	1.02487
	X	0.23998	0.08503	0.05628	0.42369
	Y	0.49974	0.15116	0.17317	0.82631
	N	0.13613	0.07302	-0.0216	0.29389
(17)	K	0.52555	97.3742	-208.322	209.373
	X	0.76322	40.1529	-85.3563	86.8828
	Y	2.32336	99.306	-210.667	215.314
	M	-0.28398	32.425	-69.8289	69.261
	P	-1.06191	52.487	-113.636	111.512
	N	0.09727	1.3858	-2.87497	3.06952

Table 5. Statistics of adjustment of the equations proposed for the design the upflow anaerobic filter separated in three phases

Eq.	p	n_1	n_2	R^2	$R^2_{adj.}$	SEE	MAE	D	d_L	d_V	RAL
(14)	4	18	9	0.82	0.78	62.35	42.91	1.41	0.93	1.69	0.32
(15)	5	18	9	0.90	0.87	48.56	32.24	2.14	0.82	1.87	-0.09
(16)	5	18	9	0.93	0.90	42.24	30.30	1.74	0.82	1.87	0.081
(17)	6	18	9	0.86	0.81	55.08	32.98	1.55	0.71	2.06	0.168

p: number of independent variables, n_1: number of data in the calibration stage, n_2: number odf data in the validation stage, R^2: determination coefficient, R^2adj.: coefficient of determination adjusted, SEE: Standard Error of Estimation, MAE: mean absolute error, d: statistic of Durbin-Watson, RAL: residual autocorrelation in lag 1.

As a sample, in the calibration stage of the **Eq. 12** for the design of the upflow anaerobic filter separated in two phases (DI-FAFS) is found the relation between variables that contribute to explain the adjustment of the data observed to the equations proposed represented in graphs as follows (**Fig. 3**): (a) the values S_e observed against the values S_e estimated in mg l^{-1} show a close approximation to the linear ratio 1:1, many of these are located on the linear representation (**Fig. 3a**), and (b) the residual autocorrelation is not significant in the series of experimental observations in *k* periods, the bars in the diagram do not exceed the limits of autocorrelation, which means that the **Eq. 12** describes the dynamic structure of the series of observations each *k* periods (**Fig. 3b**). The residues are distributed randomly, indicating that the estimated variable by the Equation 12 does not follow a pattern and the equation is adjusted to the set of observed values corresponding to S_e (**Fig. 3c**). The residues are fitted to the normal probabilistic distribution function because of the most of estimated values are superimposed on the linear graph (**Fig. 3d**).

DISCUSSION

Equations 10–17 explain the operation of the DI-FAFS and TRI-FAFS reactors, being the result of the adaptation of the formulated equations for trickling filters and anaerobic biofilters; finding similarity in the magnitude of the parameters of the residual liquid temperature (*A*) with the values reported by (Howland, 1958) for the value proposed by Vant'Hoff (1884) and the power of the Organic Load Volumetric (*n*) as (Arora and Umphres, 1987).

Equations 12–16 are proposed for the design of anaerobic upflow filter separated in two phases and three phases DI-FAFS and TRI-FAFS, respectively; resulting in a combination of independent variables that estimate the values of S_e approximated to the observations derived from the experiment. The independent variables are: the depth ratio of the separated filters in phase 1 with respect to phase 2 (D_1/D_2), the depths in phase 2 with respect to phase 3 (D_2/D_3), the temperature of the residual liquid T, and the volumetric organic load (VOL); corresponding to the three factors set in the experimental design. In both cases, the value of the power n is positive and in a range between 0 and 1. This power substituted in **Eqs.**

12–16 represents an equivalent to the contact time of the substrate t with the microorganism to achieve the removal of the affluent substrate to the DI-FAFS or TRI-FAFS filter, as it can be verified in the formulation represented by **Eq. (9)**. **Equations 12–16** resulted in a $R^2_{adjusted}$ greater than 0.7; the standard error of estimation and the absolute average error resulted in the minimum value in **Eqs. 12–16** with respect to the rest of the equations.

The separated phases of the trickling filters of synthetic support medium and organic affluent up to two stages have been applied by Germain (1966) and Lou *et al*. (2014) coinciding with this investigation. In the first, the reaction unit has depth of 6.5 m and in each stage; a ratio of recirculation of the effluent and sludge between 1 and 2; a range of VOLs that vary between 0.16 and 0.96 kg BOD_5 m^{-3} d^{-1}. In the second, the reaction unit has depth of 0.6 m in each stage; there is not recirculation of the effluent; a range of VOLs that vary between 0.16 kg COD m^{-3} d^{-1}. Germain (1966) showed that recirculation did not have a statistically significant effect on the removal of BOD_5 for filters deeper than 3 ft; and therefore on the treatability parameter k (**Eqs. 10–17**); being equivalent to the scheme proposed by the present study. Lou *et al*. (2014) found COD removal efficiencies greater than 90%. The results of the DI-FAFS and TRI-FAFS treatment systems are compared in terms of the ranges of organic matter removal efficiencies, which vary between 27 and 72.86%; 84 and 95%. These systems would be in an equivalent capacity with their separate phases of replacing the recirculation line of the effluent and sludge to the reaction unit as the multiphase-multilayer system executed by Germain (1966) and Lou *et al*. (2014). There is equivalence between the COD removal efficiency obtained for a trickling filter separated in two stages with respect the TRI-FAFS reactor.

CONCLUSIONS

1) The number of stages or phases of the novel biological filters influences the removal of the COD. The COD removal efficiencies are increased between 1.3 and 3.5 times from DI-FAFS reactor to TRI-FAFS reactor.

2) The novel hybrid models created from the modification and adaptation of the formulated equations for trickling filters have given significant statistically results, finding that the Equations (12) to (19) explain the operation of the DI-FAFS and TRI-FAFS reactors through the COD removal in a successful proportion, especially in the TRI-FAFS reactor.

3) The novel combination of three experimental factors has given significant statistically results, the Equations (12) and (16) are proposed for the design of anaerobic upflow filter separated in two phases and three phases DI-FAFS and TRI-FAFS, whose independent variables are the three experimental factors: (1) the depth ratio of

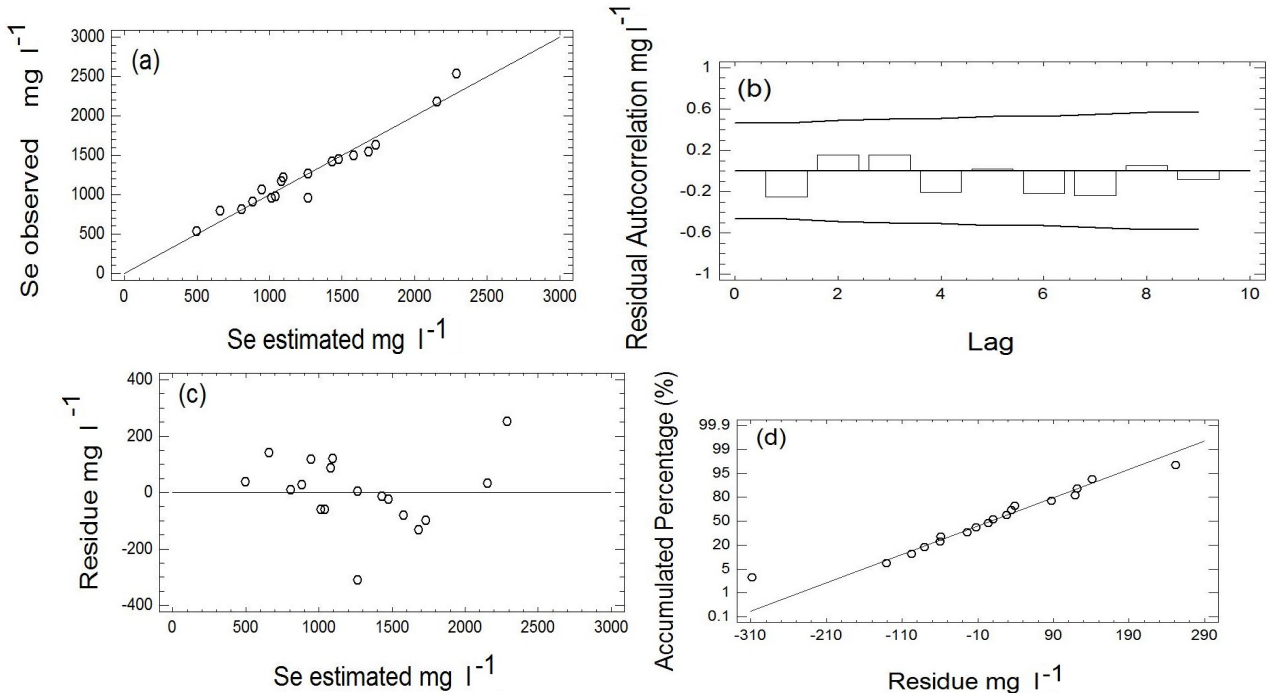

Fig. 3. Adjust of the data for estimating of the substrate concentration in the effluent (S_e) to the **Eq. 12** in the stage of the calibration of design model of the upflow anaerobic filter separated in two phases: (a) S_e observed mg l^{-1} against S_e estimated mg l^{-1}, (b) autocorrelation of residues, (c) estimation of the residual against S_e estimated, and (d) adjust of the residues to a normal distribution function.

the separated filters in phase 1 with respect to phase 2 (D_1/D_2), the depths in phase 2 with respect to phase 3 (D_2/D_3), and (2) the temperature of the residual liquid T, and 3) the volumetric organic load (VOL).

4) The phases of hydrolysis, acidogenesis and methanogenesis associated with an anaerobic process according to Metcalf and Eddy, 1996 could have occurred due to the high removal of organic matter obtained in the TRI-FAFS reactor; this could be confirmed by installing liquid phase separators-biogas CH_4 and CO_2 at the output of each phase; in a progressive measure from lower to higher production towards the methanogenesis phase. Additionally, it could be confirmed with a sampling of bacterial prevalence in each phase.

REFERENCES

Albertson, O. E., & Davies, G. (1984, October). Analysis of process factors controlling performance of plastic bio-media. In tbe 57th Annual Meeting of the Water Pollution Control Federation, New Orleans, LA.

Arora, M. L., and Umphres, M. B. (1987). Evaluation of activated biofiltration and activated biofiltration/activated sludge technologies. Journal (Water Pollution Control Federation), 183-190.

Atkinson, B., Davies, I.J., and How, S.Y. (1974). The overall rate of substrate uptake (reaction) by microbial film. Part I and II. Trans, Inst. Chem. Eng.

Chandrappa R., and Das, D.B. (2012). Solid Waste Management. Principles and Practice. Springer Science & Business Media.

Chesner, W. H., Iannone, J. J., & Bender, J. (1978). Review of current RBC performance and design procedures. A report prepared for USEPA under contract No. EPA-68-02-2775.

Germain, J.E. (1966). Economical Treatment of Domestic Waste by Plastic-Medium Trickling Filters. Journal (Water Pollution Control Federation), 38 (2), 192-203.

Gutiérrez Pulido, H., & Salazar, V. (2004). Control estadístico de calidad y seis sigma/Humberto Gutiérrez pulido, coautor Román de la Vara Salazar.

Henry, J.G., Prasad, D., and Young, H. (1987). Removal of organics from leachates by anaerobic filter. Water Research, 21, 1395–1399.

Howland, W. E. (1958). Flow Over Porous Media as in a Trickling Filter. Proc. 12th Ind. Waste Conf.t Purdue Univ., Ext. Ser. 94, 435. Kincannon D. F. and Stover E. L. (1982) Design methodology for fixed film reaction-RBCs and biological towers. In Civil Engineering for Practicing and Design Engineers (2), 107±124. Pergamon Press, New York.

Kängsepp P., Kõiv M., Kriipsalu M., and Mander Ü. (2008). Leachate Treatment in Newly Built Peat Filters: A Pilot-Scale Study. Wastewater Treatment, Plant Dynamics and Management in Constructed and Natural Wetlands, Springer, 89-98.

Luo, W., Yang, C., He, H., Zeng, G., Yan, S., & Cheng, Y. (2014). Novel two-stage vertical flow biofilter system for efficient treatment of decentralized domestic wastewater. Ecological engineering, 64, 415-423.

Marquez A., Nava J. (2002). Eliminación de componentes orgánicos en aguas residuales mediante un reactor del tipo Biopelicula Sumergida Aireada. Revista Ingenieria UC. 09(02), 02-08.

Maldonado J., Rodríguez J.A. and Cajiao A. M. (2017). Treatment landfill leachate in filters anaerobic upflow of two phases (DI – FAFS). Revista Ingeniería UC, 24 (1), 91 – 104.

Metcalf and Eddy. (1996). Ingeniería de aguas residuales. Tratamiento, vertido y reutilización. Tomo 1. Mc Graw Hill Interamericana de Editores. México.

Perozo, L.M. (2015, November 3). Aseo Urbano llega a sus 15 años con un parque ambiental innovador. Somos, Extracted on December 8, 2017 from http://somoslarevista.com/2015/11/aseo-urbano-llega-a-sus-15-anos-con-un-parque-ambiental-innovador/.

Phelps, E. B. (1944). Stream Sanitation. John Wiley and Sons, Inc., New York, N. Y.

Schulze, K. L. (1960). Load and Efficiency of Trickling Filters. Water Pollution Control Federation. 32(3), Part I, 245-261.

Stack, V. T., Jr. (1957). Theoretical Performance of the Trickling Filtration Process. Sewage and Industrial Wastes, 29 (9), 987-1001

Stover, E. L., & Kincannon, D. F. (1982). Rotating biological contactor scale-up and design. Oklahoma state univ Stillwater.

Van't Hoff, J. H. (1884). In Etudes de Dynamiques Chimiques (F. Müller and Co., Amsterdam), 114; translated by T. Ewan as Studies in Chemical Dynamics (London, 1896).

Velz, C. J. (1948). A Basic Law for the Performance of Biological Filters. Sewage Works Journal, 20 (4), 607-617.

Water Pollution Control Federation (1988). O & M of trickling filter, RBCs and related processes. Manual of practice OM-9 1987.

Wojciechowska, E. and Obarska-Pempkowiak, H. (2008). Performance of reed beds supplied with municipal landfill leachate. Wastewater Treatment, Plant Dynamics and Management in Constructed and Natural Wetlands, Springer, 251-265.

Yang, C., Yu, G., Zeng, G., Yang, H., Chen, F., and Jin, C. (2011). Performance of biotrickling filters packed with structured or cubic polyurethane sponges for VOL removal. Journal of Environmental Sciences, 23(8), 1325-1333, 1325-1333, 10.1016/S1001-0742(10)60565-7.

Yu, H., Wilson, F., and Tay, J. H. (1998). Kinetic analysis of an anaerobic filter treating soybean wastewater. Water Research, 32(11), 3341-3352.