Dynamics of the behaviour of a vertical wetland (French system) operating in warm-climate conditions, evaluated by means of variables continuously measured *in situ*

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ABSTRACT

The sewage treatment system in this study was operated with only the first stage of a French system of vertical wetlands, composed of two units in parallel and running with an extended feeding cycle (7 days). This research sought to evaluate and relate continuous variables measured *in situ* (dissolved oxygen (DO), pH and redox potential) throughout the feeding cycle, with measurements at distinct heights along the filter vertical profile. Additionally, the influence of the surface organic sludge deposit was investigated. A close link between the hydraulic behaviour and the effluent quality was verified, with both being related to the batch volume and the instantaneous hydraulic loading rate. The drop in DO as the feed days progressed could be related to the loss of hydraulic conductivity. A thicker sludge layer decreased the aeration capacity of the filter. The effluent was observed to be aerated when percolating through the medium. DO and pH data suggested that nitrification varied along the filter depth, the batch duration and the feed cycle. The monitored parameters may be indicative of the behaviour of other parameters.

Key words | extended feeding cycle, hydraulic behaviour, monitoring, on-site parameters, physical–chemical parameters

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HIGHLIGHTS

- For warm-climate areas there is limited literature regarding daily changes observed in the French system of vertical wetlands during a sequence of batches and through a complete feeding cycle.
- The system was operated with the first stage of a French system composed of only two units in parallel and with an extended feeding cycle of 7 days, and the results are based on continuous variables measured *in situ*.
- A close link between hydraulic behaviour and effluent quality was verified, corroborating the need to understand the hydraulics of vertical flow wetlands.
- The use of outflow hydrographs and simple monitoring parameters may contribute to a broad performance evaluation of pulse-loaded vertical flow wetlands.

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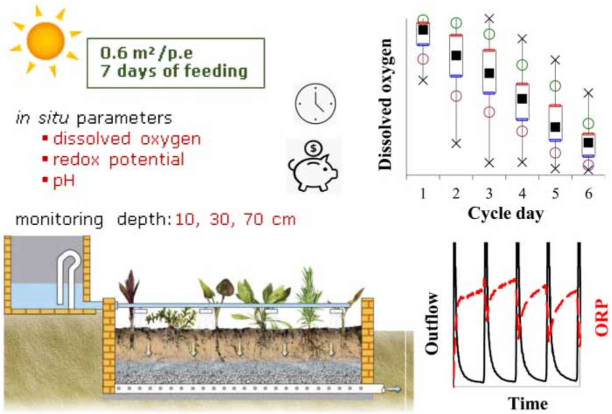
GRAPHICAL ABSTRACT



INTRODUCTION

Rodriguez-Dominguez *et al.* (2020), based on a vast review of constructed wetlands (CW) publications from Latin America, concluded that CW are not widespread in the region, possibly due to lack of design guidelines and stakeholders' knowledge. They also indicated that among the CW technologies, the least studied one is the French system (FS). There are thousands of FS units of vertical CW for raw sewage treatment, and they have been studied for several years, especially by the group from INRAE (formerly IRSTEA and CEMAGREF), in France. Throughout their studies, Molle *et al.* (2005) have detailed the main construction and operational criteria of these systems for temperate climate conditions. In these systems the medium is unsaturated and there are cycles of influent application in batches, percolation and drainage, creating cyclical variations in redox potential, with both oxidation and reduction phases (Kadlec & Wallace 2009). The FS removes contaminants that are aerobically degraded due to the high oxygenation rate of the medium resulting from the intermittent feed (Platzer & Mauch 1997). In addition, the traditional FS is composed of two stages: the first with three parallel filters, operated with alternating feeding and resting phases (usually 3.5 days of feeding and 7 days of rest) and receives raw sewage as influent (Molle *et al.* 2005); the second stage consists of two units, with feeding alternation every 3.5 days. The possibility of using only the first stage of the FS (Yadav *et al.* 2018) and having this first stage composed of only two units in parallel is an option when applying these vertical filters in tropical climate environments in order to reduce the area required by the system (Manjate *et al.* 2015; Molle *et al.* 2015; Lombard-Latune & Molle 2017; García Zumalacarregui & von Sperling 2018).

Several studies have been devoted to investigating the FS treatment performance over a relatively long period (Cota *et al.* 2011; Morvannou *et al.* 2017). This monitoring is important to improve the understanding of the transformations taking place and thus derive design and operating parameters. Studies, including those of Molle *et al.* (2006), Arias Lopez (2013), Molle (2014), Silveira (2015), Silveira *et al.* (2015), Dubois & Molle (2018), have contributed towards understanding the behaviour and resilience throughout the feeding and resting periods. However, the information obtained by those authors refers to systems installed in temperate climate conditions. For warm-climate areas there is little literature available regarding daily changes observed in the system during a complete feeding cycle, highlighting the studies performed by Andrade Moraes *et al.* (2019) and Trein *et al.* (2019b). In addition to the scarcity of information, one of the biggest problems in wastewater treatment plants (WWTP) located in developing countries (mostly tropical, like Brazil) is the monitoring of effluent quality with appropriate parameters and frequency.

Most studies include parameters analyzed in the laboratory, but only on monthly or weekly sampling frequencies (Garfi *et al.* 2014). Parameters continuously measured *in situ* can help towards a better understanding of the system, are quick and easy to measure, and can indicate the need for immediate intervention in the system operation (Kayser *et al.* 2003). These are also critical parameters because they are closely related to other parameters of interest: for example, dissolved oxygen (DO) is associated with organic matter removal and nitrification, pH with the nitrogen cycle, and the electrical conductivity with dissolved solids. Moreover, the hydraulics of the system is directly related to these parameters.

This research therefore evaluated and related outflow pattern with parameters measured *in situ* throughout a sequence of batches in an extended feeding cycle, with measurements at different heights of the bed column (vertical filter profile). Additionally, the influence of the surface organic deposit layer was analyzed. The system operated with only the first stage of the FS, composed of two units in parallel and an extended cycle of 7 days of feeding and 7 days of resting. Although this study focuses on parameters measured *in situ*, it does not indicate that traditional parameters should not be monitored. The overall evaluation of the system performance was also composed by conventional monitoring parameters, but these are not covered here.

MATERIAL AND METHODS

The research was conducted at the Centre for Research and Training in Sanitation (CePTS UFMG/Copasa), located at the Arrudas WWTP, in the city of Belo Horizonte (Brazil) (19°53′42″S latitude, 43°52′42″W longitude). Belo Horizonte is located in the subtropical humid climate Cfa or Cwa according to the Köppen classification, with an annual average temperature of 21.8 °C. January is the warmest month (mean of 23.4 °C) and July the coldest month (mean of 19.1 °C), and the mean annual rainfall is 1.602 mm·year^{-1} (INMET 2018). After preliminary treatment at the Arrudas WWTP, consisting of screening (coarse screens, medium screens and sieves with openings of 10 cm, 15 mm and 6 mm, respectively) and grit removal (square mechanically cleaned grit chambers), a small aliquot of the WWTP was diverted to the FS under study.

The FS was designed to treat 13 $\text{m}^3\text{·d}^{-1}$ (approximately 100 population equivalent, p.e.) of raw sewage, and consisted of a storage tank and three parallel units (only two used in this study) planted with the grass species Tifton 85 (*Cynodon* spp.). Each filter had a surface area of 29.1 m^2 and the average surface hydraulic loading rate on the operating filter was 0.45 $\text{m}^3\text{·m}^{-2}\text{·d}^{-1}$, higher than the French recommendation of 0.37 $\text{m}^3\text{·m}^{-2}\text{·d}^{-1}$ (Molle *et al.* 2005; Morvannou *et al.* 2015). On the other hand, because the influent chemical oxygen demand (COD) concentrations were low, the resulting average surface organic loading rate was 202 g COD $\text{m}^{-2}\text{·d}^{-1}$ (together with 182 gTSS $\text{m}^{-2}\text{·d}^{-1}$) (García Zumalacarregui & von Sperling 2018), which is lower than the maximum recommended by Lombard-Latune & Molle (2017) of 350 g COD $\text{m}^{-2}\text{·d}^{-1}$ for tropical climate conditions. However, it should be noted that there were only two filters in operation, whereas in France three units are used, leading to lower overall loading rates. Details of the influent concentrations can be found in Trein *et al.* (2019a). The filters had three layers arranged in increasing particle size from top to bottom: main layer (0.40 m): gravel mesh 2.4–12.5 mm; intermediate layer (0.15 m): gravel mesh 4.8–25 mm; and drainage layer (0.15 m): gravel mesh 19–50 mm. The effluent was drained out of each unit by a 100 mm perforated central pipe, connected to a double passive aeration chimney. The system started operation in 2009, and since then it had different arrangements and operating conditions.

The typical operation of the first stage of a FS comprising three units in parallel varies from 3 to 4 consecutive days of feeding and 7 days of resting (Dotro *et al.* 2017). However, because this study only used two units (I and II) and the operational cycle consisted of alternations every 7 days, the feeding cycle was composed of 7 days and the resting cycle of another 7 days in each unit. The two units in operation received similar loads and differed only in the height of the surface sludge layer (organic deposit layer). In May 2018, Unit I exhibited an average sludge height of 7.2 cm, while in Unit II it was only 0.5 cm, because the sludge had been previously removed for research purposes (details in Trein *et al.* (2019a)).

The FS does not work with continuous flow, i.e. the feed is performed in the form of pulses (batches), and, in this study, the operating unit performed on average 24 pulses per day of 0.55 m^3 each. When the storage tank was filled to a volume of 0.55 m^3, which occurred after 45 min to 1 h, a siphon was hydraulically actuated and the raw sewage was distributed on top of the operating unit. The instantaneous hydraulic loading rate HLR_{inst} [pulse volume/(surface area × tank discharge time)] was 4.7 $\text{L·m}^{-2}\,\text{min}^{-1}$. According to Dotro *et al.* (2017), the HLR_{inst} should be greater than 8 $\text{L·m}^{-2}\,\text{min}^{-1}$ (0.5 $\text{m}^3\text{·m}^{-2}\text{·h}^{-1}$). Therefore, considering the value suggested by those authors, the installed siphon could be considered insufficient, and this could potentially affect drainage. According to Platzer & Mauch (1997), removal mechanisms work properly when the filter is well drained between two consecutive pulses, thereby avoiding anaerobic conditions (Zhao *et al.* 2009). The efficiency therefore drops when constant clogging or ponding occurs.

Plastic sampling ports were introduced at depths of 10 and 30 cm in the filtering units and connected to pipes for collecting the percolated effluent samples. The top surface was considered as the reference level (depth = 0). The YSI 6600 v2 multiparameter probe, which measures and stores parameters *in situ* (DO, pH, temperature and redox potential), was installed to measure the percolate at the two intermediate depths, plus the total depth (70 cm), which is represented by the filter effluent. Figure 1S (Supplementary Material) shows a sectional sketch of the system with the position of the plastic sampling ports and multiparametric probes. The sample ports were inserted inside the filters more than a year before the beginning of the *in situ* measurements, so that the filter and microorganisms had sufficient time to adapt to it.

The probe was programmed to acquire data on DO, pH and oxidation–reduction potential (ORP) every 40 s. It should be noted that in none of the monitoring campaigns were the three depths of interest monitored simultaneously, because there were only two probes available. The parameters of Unit II were also accompanied by effluent flow measurements (see details in Andrade Moraes *et al.* (2019)). In some pulses, samples were also collected every 5 min and total suspended solids (TSS) concentrations were measured using a Hach TSS Portable Hand probe.

Seven specific monitoring campaigns were conducted: three in Unit I and four in Unit II (see Table 1S). Each monitoring campaign lasted 7 days, accompanying the whole duration of the extended feeding cycle. In addition, local air temperature and rainfall were measured by a Davis Vantage Pro2 Weather Station installed within the CePTS. Six of the seven monitoring campaigns were conducted within the rainy season, but not all necessarily associated with rainfall events.

Statistical tests were conducted with the aid of the Statistica 10.0 software, using a significance level $\alpha = 5\%$. The tests were non-parametric: the Mann–Whitney *U*-test when testing the difference between only two samples (for example, difference of a given parameter between Units I and II) or the Kruskal–Wallis analysis of variance (ANOVA) when testing the difference between multiple samples (for example, difference in a given parameter between days of the feed cycle: days 1 to 7).

RESULTS AND DISCUSSION

Relationships between flow and parameters measured *in situ* along a sequence of batches

Details of the dynamic behaviour of the effluent flow in this system can be found in Andrade Moraes *et al.* (2019). Pulse feeding caused a fast peak in outflow, followed by a slow decline (as described by Kadlec & Wallace (2009)), tending to zero. At the beginning of the 7-day feeding cycle the system required time to soak and retain moisture; from the middle to the end of the feeding cycle, the stored liquid volume in the system increased, probably due to a reduction in filter permeability. The fluid percolation time increased over the days of the feeding cycle in both units (Andrade Moraes *et al.* 2019). However, according to those authors, the system was still able to maintain the liquid drainage capacity during the extended 7-day feeding cycle.

The data displayed in Figure 1 is from a typical sequence of six consecutive pulses from Unit II, but is representative of longer periods. The parameters DO, ORP, pH and TSS were shown to be directly associated with the outflow pattern. This finding corroborates the need to understand the hydraulic behaviour of a vertical flow wetland, materialized in its effluent hydrograph.

The minimum DO concentration in the final effluent occurred when the outflow increased (Figure 1(a)). Thereafter, the concentration increased until it began to decline with the end of the peak flow. This may be related to the percolation time, i.e. the faster the liquid infiltrates and percolates through the porous medium, the shorter the time to promote attachment to biofilm and organic matter degradation, and therefore less oxygen is consumed by microorganisms. Moreover, oxygen transfer into the filter media is dependent on operating conditions, and occurs by the convection mechanism during sewage application and diffusion throughout the resting period (Platzer 1999; Molle *et al.* 2008). The behaviour of the ORP was also a function of the outflow (Figure 1(b)). The maximum point on the flow hydrograph was when the lowest ORP occurred, which indicates the reducing characteristics brought about by the fast insertion of the raw sewage into the filter during the feeding. After that, the ORP grew continuously, indicating a predominance of oxidation conditions inside the filter medium. Therefore, the highest ORP values occurred the moment before the peak flow. Figure 1(c) demonstrates the relationship between the outflow and ORP variation with time. It was observed that when this variation had negative values, the filter would be in reducing conditions, and when the variation was null or positive it would be in an oxidative state (Figure 1(c)). This is in agreement with Figure 1(a) and 1(b), indicating the prevailing oxidative conditions inside the filter medium.

Kayser *et al.* (2003) monitored a vertical flow wetland fed by the effluent from a pond system, where part of the final effluent was recirculated. They observed that both

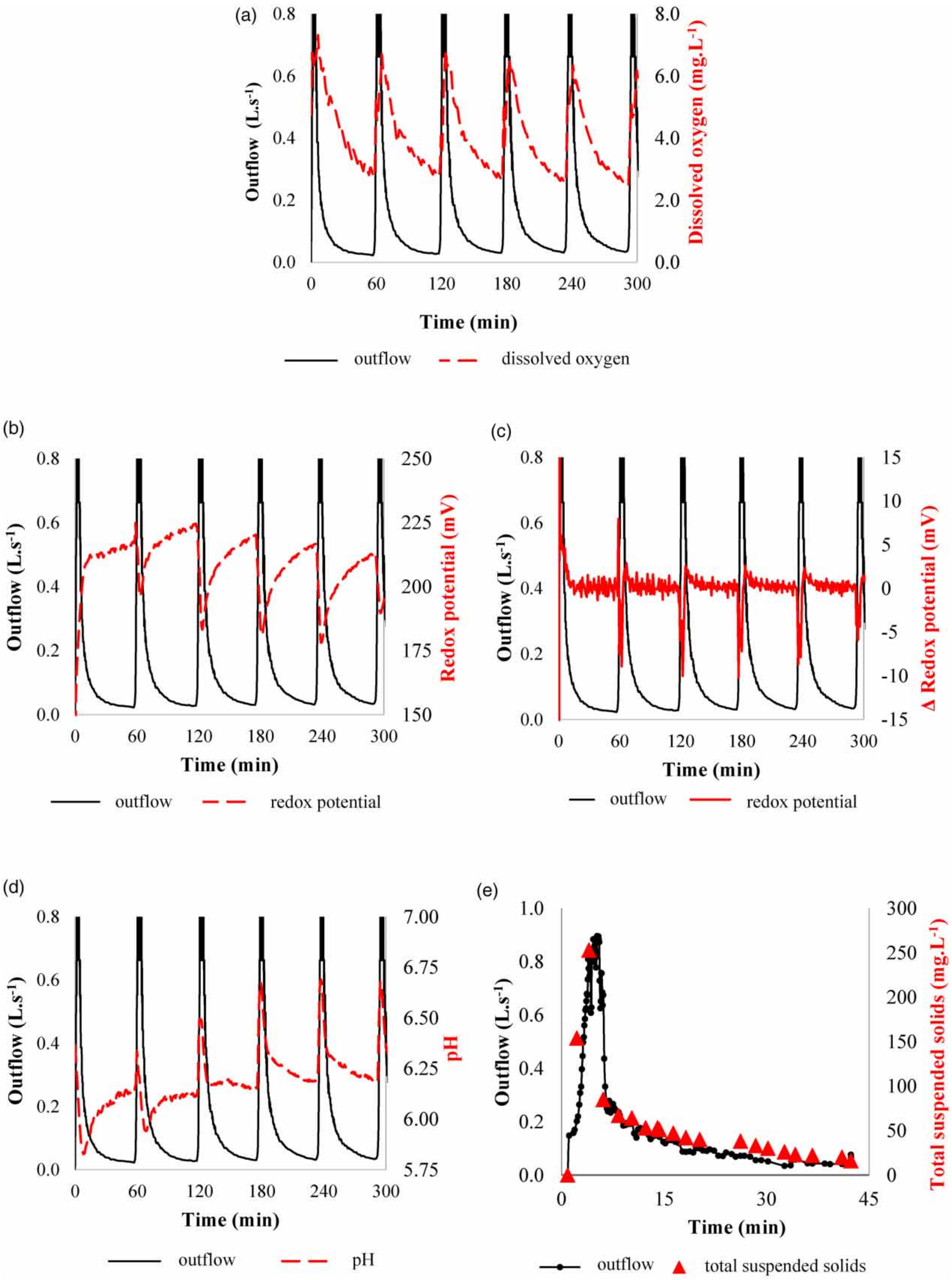


Figure 1 | Relationships between outflow and measured values of constituents in the final effluent of Unit II during a sequence of batches: (a) DO; (b) redox potential; (c) redox potential variation; (d) pH; (e) TSS.

DO and ORP decreased immediately after the feeding pulse, and then gradually increased until the initial value. However, in the present study, the data showed that only the ORP dropped instantly (Figure 1(b)), while DO recovery was only possible in the next pulse when there was a renewal of direct DO supply (Figure 1(a)). The distinct behaviour between this system and the system presented by Kayser *et al.* (2003) may be due to the material used in the filter (gravel in this system and sand in theirs), the areal loading rate (0.6 in this one and 2.5 $m^2 \cdot p.e^{-1}$. in theirs), the instantaneous hydraulic loading rate (not specified in theirs), the quality of the influent (raw sewage in this one and pre-settled in theirs) and the interval between two pulses (1 h in this one and 6 h in theirs). Kayser *et al.* (2003) found that after feeding, DO fell abruptly, but between two batches there was enough time for DO recovery, which did not occur in this study. Therefore, changes in flow affect both the redox potential and the oxygen content inside the filter (Silveira *et al.* 2015).

Regarding pH, its increase coincided with the elevated outflow, and the decay was associated with outflow (Figure 1(d)). The pH behaviour may be related to the nitrogen cycle. Paing *et al.* (2015) reported that pH change in the effluent may be due to alkalinity consumption associated with nitrification (release of H^+ ions). Those authors found lower pH values in the effluent from the second stage of the FS, in which nitrification is expected to occur intensely. Nitrification is also capable of occurring in the first stage of the FS in warm-climate areas with only two parallel filters (Manjate *et al.* 2015; Molle *et al.* 2015; Lombard-Latune *et al.* 2018). Indeed, during this research, consistent nitrification was found to take place (data not shown here), as reported by Trein *et al.* (2019b).

The data obtained by Kayser *et al.* (2003) in a vertical flow wetland showed that (1) ammonium concentration in the final effluent was variable during the pulse, with peak concentrations when ORP was minimal, and (2) during ORP recovery, the ammonium concentration of the effluent was decreasing and kept to a minimum until the beginning of next batch. Following this dynamic, when ORP started to increase, nitrification growth also occurred. In our system, during ORP recovery (see Figure 1(b)) was when the pH (see Figure 1(d)) was at the lowest (and its variation was practically insignificant), which could indicate that there was maximum nitrification.

Figure 1(e) shows the outflow and TSS concentration for a single pulse only. This behaviour was the same as reported by Manjate (2016). The TSS pollutograph is similar to the effluent flow hydrograph. A faster percolation velocity decreases the contact time of solids with sewage, which increases the concentration of solids in the effluent, justifying the higher concentration of pollutants in the first minutes of the pulse. Organic matter indicators, such as total organic carbon, COD and biochemical oxygen demand (BOD), are also likely to follow the same profile.

Among the parameters exhibited by Trein *et al.* (2019b), TSS showed significant difference between the two units. The thicker sludge layer reduced the peak and smoothed the pollutographs, causing the unit with more mature sludge to present significantly lower TSS concentrations.

Based on the consistency of these results, it can be postulated that in the FS, with the use of effluent hydrographs combined with simple on-site parameters, it is possible to infer about other physical–chemical parameters, reducing the frequency and number of parameters to be monitored in real situations.

Behaviour of parameters measured *in situ* throughout the days of the feeding cycle

Figure 2 illustrates the evolution of DO, ORP and pH at the effluent from Unit II (the unit with the smallest surface deposit layer) over the feeding cycle. The samples considered were all those that comprised Unit II (dry). Data were only obtained up to the 6th day of the feeding cycle because the battery inserted into the device discharged before completing the 7 days. At low flow rates, the probe installed in the treated effluent collection sampling port may also have read values of atmospheric air rather than data from the liquid. Therefore, DO data points greater than 8 $mg \cdot L^{-1}$ were excluded. It is important to highlight that all data presented in this study were influenced by the aeration chimney connected to the drainage pipe up to the surface.

A progressive decrease in DO was also observed, and the rate of decline appeared to be constant, showing a linear relationship with time (Figure 2(a)). On day 1, the 50% percentile (median) of DO is close to saturation, and on day 6 it drops to 1.7 $mg \cdot L^{-1}$, indicating that the system is losing its aeration capacity. As reported by Andrade Moraes *et al.* (2019), an increase in feeding days led to the system permeability declining, which progressively increased the liquid retention and reduced the volume of pores available to be occupied by atmospheric air, causing a decrease in oxygenation of the medium. The statistical test was performed to evaluate the difference between the concentration medians on the different days. The results indicated that there was significant difference between all samples. DO dropped significantly every day. The ORP during the cycle remained positive, indicating that the system maintained its aerobic characteristics (Figure 2(b)), an important requirement for organic matter degradation and nitrification (Torrens *et al.* 2009). Between days 2 and 4, the data variability was higher compared to the other days.

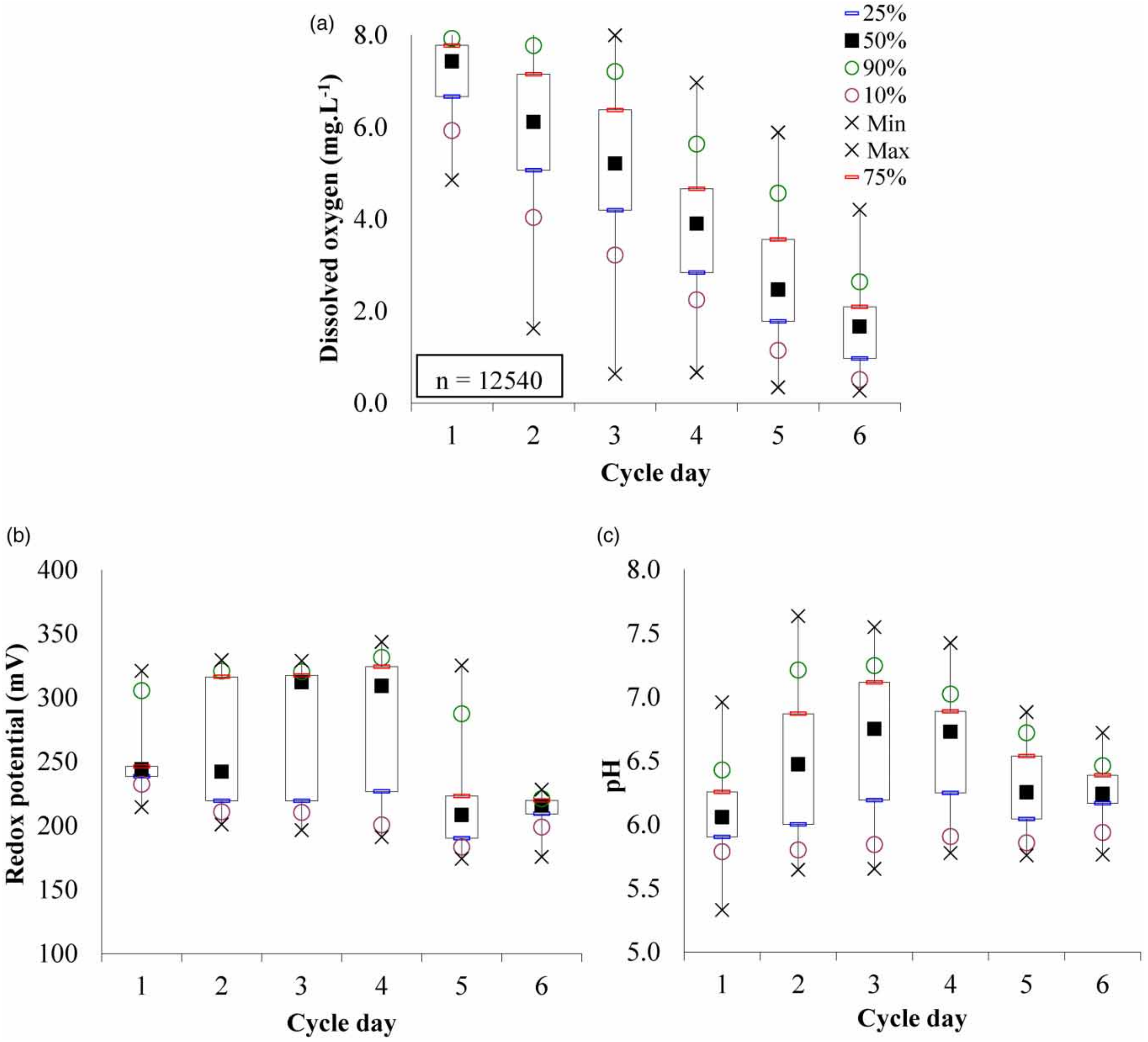


Figure 2 | Box-plots of measurements of constituents in the wetland effluent throughout the feeding cycle, undertaken in Unit II (in which the surface deposit had been previously removed): (a) DO; (b) redox potential; (c) pH. n = number of data points.

Although Lombard-Latune & Molle (2017) recommend that the liquid layer applied per batch should be between 2.5 and 5 cm for tropical climate conditions, the system was operated with only 0.18 cm. The number of daily batches (24 batches per day) was higher than suggested, which led to a short time interval between pulses. The low initial ORP values could be due to insufficient drainage periods between feedings (Ruppelt *et al.* 2019). However, it is noteworthy that during the initial operating years of this system (2010–2012), when it was fed for 2.5 days and rested for 4.5 days, Lana *et al.* (2013) evaluated the impact of the liquid layer applied per batch on the effluent quality: 0.940 m^3 (3.2 cm) every 2 h versus 0.560 m^3 (1.9 cm) every 1 h. The authors concluded that the lower wastewater volume per batch enhanced the removal efficiencies from 72% to 80% for COD, and from 56% to 60% for ammonium. They attributed this performance improvement to the longer percolation time provided by the smaller liquid layer. Although it is the same system, the operational conditions and the ages of the sludge deposit layer in Lana *et al.*'s (2013) study and this one were different. In summary, the designer should keep in mind that the frequency between batches must be calculated with discretion, so that there is a balance between the time needed for wastewater-bacteria contact and proper liquid drainage.

García Zumalacarregui *et al.* (2018) also reported the prevalence of aerobic conditions throughout the extended 7-day cycle in this same wetland system. However, those authors witnessed a decrease in TSS removal efficiency from day 4. Although there is potential for aerobic degradation and increased percolation time in the porous medium to occur over the days (allowing greater contact between liquid and biomass), the authors attributed this decline to other processes, such as deterioration in filtration capacity. Thus, the limitation of the system seems to be more physical than biochemical.

It was observed that the pH rose until day 3 and then began to decrease slightly (Figure 2(c)). The results of the statistical test indicated that there was no significant differences between days 5 and 6.

Garfi *et al.* (2014) compared the results of online continuously monitored parameters (redox potential, turbidity and ammonium) with conventional laboratory analyses carried out in a pilot subsurface horizontal wetland. Their study showed a good correlation between online turbidity and laboratory TSS ($r = 0.85$), online turbidity and laboratory BOD ($r = 0.88$) and online redox potential and laboratory BOD ($r = -0.62$). They stated that, for their system, when the redox potential or turbidity reached the values <100 mV (for Standard Hydrogen Electrode) or >5–10 NTU, respectively, BOD and TSS concentrations in the effluent would be higher than the limits allowed by European Council Directive 91/271/EEC, indicating the occurrence of operational problems. The authors also (1) demonstrated that online monitoring was not only simpler, but its cost was around two times lower compared to conventional monitoring, and (2) suggested that online measurements are more reliable because they are less susceptible to procedure variations.

Kayser *et al.* (2003) stated that the drop in effluent ORP due to the hydraulic overload followed the decay of nitrification. According to these authors, when a hydraulic overload occurred, the ORP decreased, increasing the reducing conditions inside the filter and ammonium concentration in the final effluent. Thus, they emphasized that if the ORP decays to a certain value defined as critical (which was evaluated for each system because it depends on the operating conditions and the effluent quality) and does not stabilize again, it indicates the possibility of surface ponding, suggesting the need of reducing the hydraulic loading or starting the resting period.

Influence of the surface organic deposit layer

Figure 3 illustrates the evolution of DO, ORP and pH at the effluent from Unit I (with thicker surface sludge layer) during the feeding cycle. The samples considered were those that comprised Unit I (unit with sludge layer) (dry). Data were only obtained upto the 6th day of the feeding cycle because, once again, the battery inserted into the equipment was discharged before completing the 7 days.

Note that Unit I follows the same trend observed in Unit II, i.e. decrease in aeration capacity over time (Figure 3(a)). The 50% percentile (median) of DO on day 1 reached 4.7 $mg{\cdot}L^{-1}$ and, on day 6, had already fallen to 0.3 $mg{\cdot}L^{-1}$. The rate of decline, however, seems to decrease over time. The Kruskal–Wallis ANOVA test was performed to evaluate the difference between the median concentrations on the different days. The test showed a significant difference every day, and thus day 1 had the highest DO concentration followed by a significant decrease each day.

To investigate the influence of the surface sludge layer, the Mann–Whitney *U*-test was performed to compare the median concentrations of Units I and II on the same day (days 1, 3, 4, 6). The results showed that the samples differed from each other, with DO from Unit II being significantly higher than that from Unit I. Figure 2(a) is reproduced again in Figure 3(b) for a better comparative viewing. The two units were basically distinguished by the height of the sludge deposit. In an investigation of the same system, García Zumalacarregui (2018) and Andrade Moraes (2019), based on a hydrodynamic tracer test and a gas-transfer propane test, found mean percolation times of 9.5 and 9.0 min on the 4th day of the cycle, and oxygen transfer rates of 194 and 230 $gO_2{\cdot}m^{-2}{\cdot}d^{-1}$, respectively, for Units I and II. Thus, the decrease of effluent DO in Unit I was more intense than in Unit II because the more mature sludge layer unit caused not only a lower oxygen transfer rate, but also a lower percolation rate that favoured biological degradation and consequently a greater oxygen consumption.

Kania *et al.* (2017) found in their research with other FS wetlands that micropores increased as the sludge aged. Furthermore, the macropores in the sludge deposit could affect air penetration and increase water retention in the layer. However, this did not necessarily imply that performance was degraded because the sludge layer played a positive role in solids retention and pollutant sorption (Molle *et al.* 2005; Kania *et al.* 2017). Additionally, moisture fosters the permanence of biological activity (Molle *et al.* 2015). The organic deposit is therefore closely related to hydraulic conductivity, oxygen transfer and biological activity (Molle 2014); however, over time this deposit may lead to clogging.

Although Unit I (with higher sludge accumulation) had a significantly lower DO concentration than Unit II, in a study of the same system conducted between 2017 and 2018 Trein *et al.* (2019b) verified that the final TSS concentration in Unit I was significantly lower than in Unit II, as

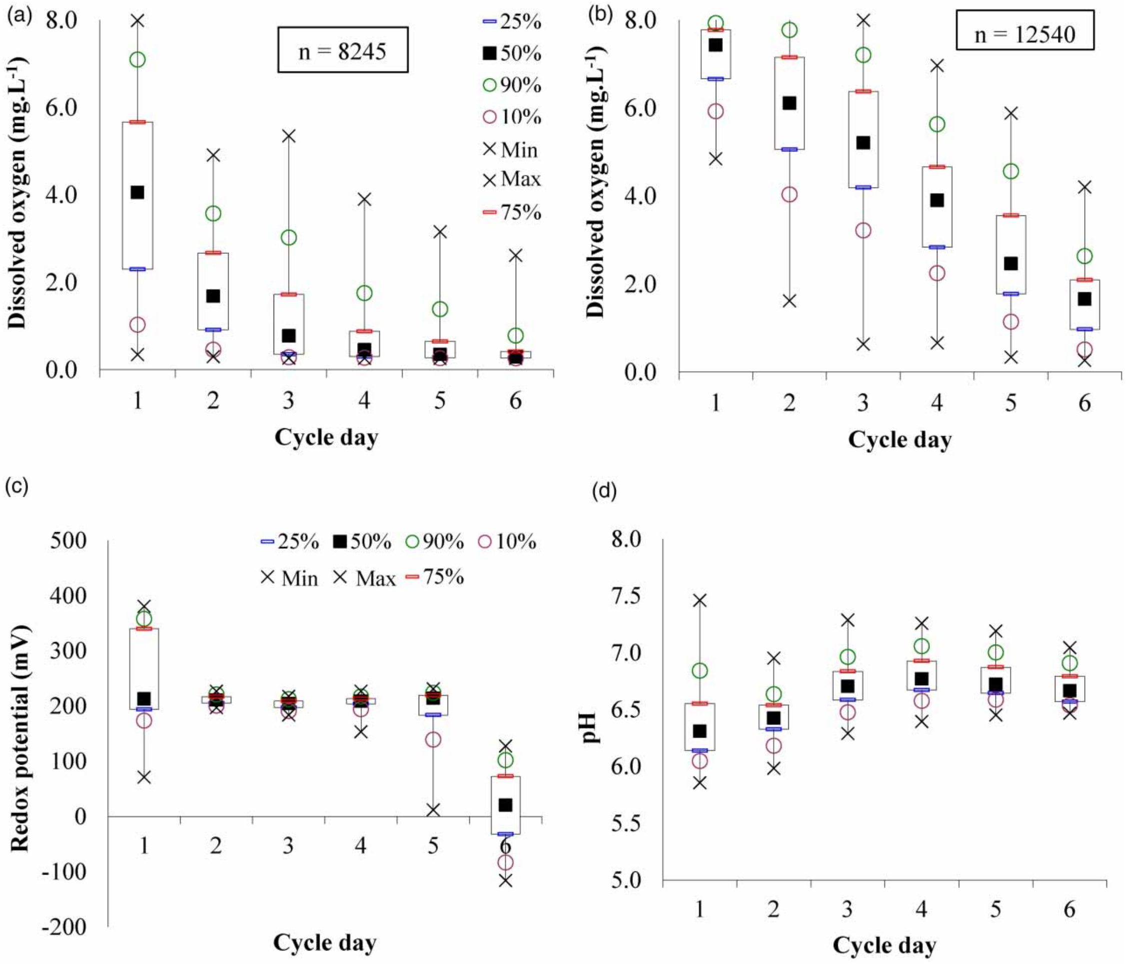


Figure 3 | Box-plots of measurements of constituents in the wetlands effluents throughout the feeding cycle (a) Unit I (with sludge layer) – DO; (b) Unit II (without sludge layer) – DO; (c) Unit I – redox potential; (d) Unit I – pH. n = number of data points.

already mentioned. They also observed that there was no significant difference between the concentrations of COD, total nitrogen and ammonia nitrogen (NH_4^+-N). The removal efficiencies of Units I and II were, respectively, 60% and 63% for COD, 84% and 76% for TSS, and 54% and 59% for ammonium. Therefore, the authors concluded that the higher sorption capacity and slower infiltration rate of Unit I compensated for its lower DO concentration, which was still sufficient to degrade organic matter and nitrify. The authors also noted that during this period, organic matter removal was not high, but still sufficient to comply with the local discharge legislation limits ($COD \leq 180\,mg{\cdot}L^{-1}$ or removal efficiency $\geq$55%). More details regarding the variability of the effluent quality in terms of these and other parameters can be found in Trein *et al.* (2019b).

Redox potential throughout the cycle maintained positive values (Figure 3(c)), but lower than those obtained for Unit II (Figure 2(b)). However, on day 6 the median dropped sharply (Figure 3(c)). Therefore, as of day 5 the system showed anoxic characteristics. Between days 2 and 5 the data variability was also lower compared to the other days.

The pH samples demonstrated a similar behaviour to Unit II: increase over time followed by a slight drop (Figure 3(d)). The pH variability of Unit I was slightly lower than Unit II. The statistical test (Mann–Whitney *U*-test) indicated that there was no significant difference between days 1 and 2, and 3 and 6, but there was a difference between the other combinations.

Trein *et al.* (2019b) carried out daily monitoring campaigns of the effluent covering the 7-day feeding cycle for the same system under study. In relation to effluent COD, the authors noticed a higher concentration on day 1, attributed to the probable detachment of solids and biofilm after the resting period. They also reported that variation of effluent COD in Unit II (thinner sludge layer) was greater due to its lower capacity of retention and filtration of solids, compared with Unit I. In relation to nitrogen, the results showed that nitrate nitrogen (NO_3^--N) concentration of the final effluent decreased throughout the feeding cycle, reflecting a reduction in nitrification.

In the 7-day feeding cycle, day 3 was a good representation of the system's typical conditions. Additionally, an extended feeding cycle may not only reduce the operating costs, but also provide better conditions for the organisms to survive during the resting period because it retains more moisture. However, the performance in the last days of such an extended cycle can be affected (Trein *et al.* 2019b).

The DO data in the rainy season showed that even with precipitation, the final effluent DO tended to decrease as the cycle progressed. It was also observed that Unit I maintained higher DO concentrations than Unit II. According to Molle *et al.* (2005), the rainy season leads to low sludge mineralization, contributing to limited oxygen infiltration and renewal. Rainwater occupies the filter pores that could be occupied by air. Precipitation therefore changed the conditions of the medium and could also have different effects. The oxygen present in the liquid was controlled by the precipitation intensity and also by the state of the filtering medium. The pH values decreased with the occurrence of rain.

Relationship between parameters measured *in situ* and depth in the filter column

Figure 4(a) shows the evolution of DO along the depth of Unit I in some consecutive pulses. It was observed that the lowest concentrations occurred at 10 cm and the highest at 70 cm (final effluent), indicating that the effluent was aerated as it moved downwards. The particle size of the three filter layers increased with depth, while retained solids and a more developed biofilm decreased with depth. Thus, the filter macropores, i.e. the draining porosity, is larger at the bottom, which allows the increase in DO transport mechanisms by advection (Langergraber & Šimůnek 2005).

Ruppelt *et al.* (2019) investigated the influence of ORP in the removal of different pollutants at two depths (upper and lower filter) of a vertical filter. They concluded that (1) a relationship between ORP in the upper filter and COD removal efficiency could be expected: greater drops in ORP increased COD removal ($r = 0.58$), and low COD removals were expected if the ORP did not decrease after loading; (2) at high ORP values, good nitrification performance was expected, and fast ORP recovery in the upper filter led to low ammonium concentrations in the final effluent ($r = 0.5$); and (3) the higher the initial ORP, the better the ammonium removal efficiency ($r = 0.42$) and *Escherichia coli* ($r = 0.41$). When comparing ORP along the depth in a continuous sequence of pulses in our system, the data showed that ORP at 10 cm reached lower values (achieving negative values) and a more pronounced drop compared to 70 cm. This indicated that the upper filter was both a more reducing region and had greater variability in ORP values, influenced by the more active biological community in that region. However, Ruppelt *et al.* (2019) noted that the drop of ORP at the lower filter was more pronounced.

Figure 4(b) was based on data from Unit II (rainy). Although they were samples from Unit II in the rainy season, it was assumed that the behaviour could be

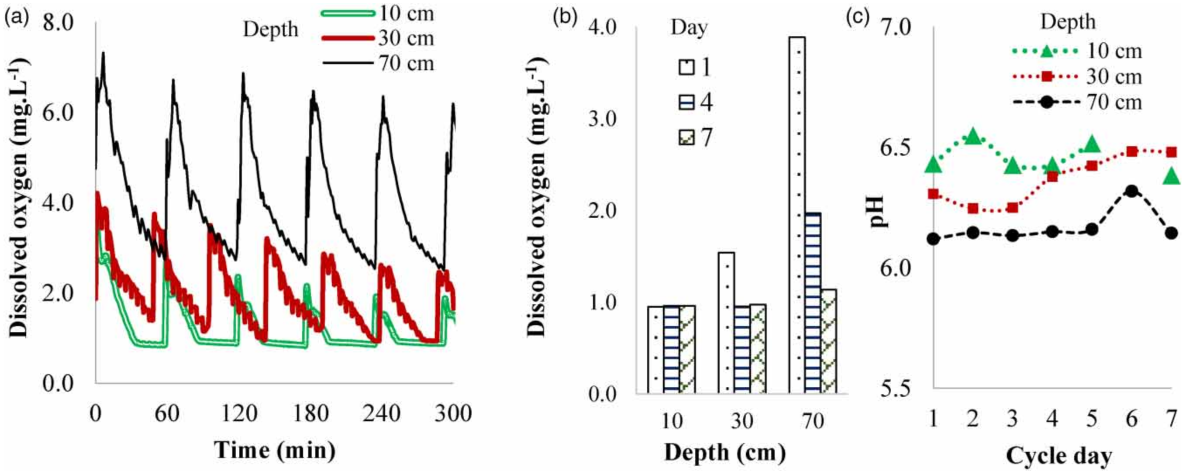


Figure 4 | Unit II – (rainy): (a) relationship between filter depth and DO profile along a sequence of batches. (b) DO (median) as a function of depth on different days of the feeding cycle; (c) pH (median) at different depths along the days of the feeding cycle.

extrapolated to other cases. It was observed that at a depth of 10 cm, throughout the cycle, the median DO concentrations were around 1 mg·L^{-1} (Figure 4(b)). Day 1 had greater DO variability in the vertical medium profile, and it was when the effluent was most aerated. As the feeding cycle progressed, the oxygen transfer capacity decreased. On day 7, the DO varied little as a function of depth.

Figure 4(c) shows that the pH tended to be higher at 10 cm and decrease with depth. Langergraber & Šimůnek (2005) observed in their models that the NH_4^+-N concentration decreased with vertical distance, while NO_3^--N concentration increased, suggesting that nitrification increased with depth.

According to Molle *et al.* (2005), nitrification is limited by the presence of DO, competition with organic matter and hydraulic retention time. Nitrification may therefore intensify in the final portion of the filter medium. Molle *et al.* (2015) found that an 80-cm tall filter promoted better nitrification than a shallow = 30-cm unit. Millot *et al.* (2016) investigated the influence of the filter height (40 cm and 100 cm) and doubling the hydraulic and organic loads on the nitrification in the first stage of an experimental FS. The authors observed that with the larger depth, a greater removal of ammonium was achieved (from 62% to 81%), but it did not lead to a significant gain in TSS and COD removal because the most effective removal mechanisms took place mainly in the upper layers (around 60% of dissolved COD was removed in the first 20 cm of the filter). Moreover, doubling the hydraulic load caused a decrease in the removal of ammonium from 62% to 44% and of dissolved COD from 59% to 44%. Nevertheless, the authors emphasized that a compensation could be made so that the quality of the effluent would not be impaired: a decrease in area.p.e^{-1} (or increase in hydraulic load) combined with an increase in filter height.

CONCLUSIONS

This work sought to contribute to a better understanding of system performance by evaluating the *in situ* behaviour of parameters such as DO, pH and redox potential in a vertical wetland system consisting of the first stage of the FS operating with only two units and an extended feeding time (7 days, greater than the usual 3.5 days in France). The parameters were continuously measured along the filter batches, feeding cycle and vertical profile. Furthermore, the influence of the surface organic deposit layer and rainfall were investigated. It should be noted that the results obtained were for this non-typical FS.

When observing output hydrographs and pollutographs in a sequence of batches, a close association between hydraulic behaviour and effluent quality was verified. Several parameters were linked to the outflow hydrograph, such as TSS, redox potential, pH and DO.

The progressive drop in DO as the feeding days progressed could be related to the decrease in hydraulic conductivity, which contributes to greater fluid retention within the system, with less space for atmospheric air insertion and permanence.

When comparing the filters with different sludge deposit layers, it was observed that the larger layer was associated with a decrease in the aeration capacity of the filter. Regarding variations along the vertical profile of the filters, it was observed that DO concentrations increased with depth. The largest variation was in the main filter layer, when comparing the depths of 10 cm and 30 cm.

In several WWTP, especially in small systems located in developing countries, sampling and routine monitoring is a problem. The present study showed that the use of outflow hydrographs, combined with simple parameters measured by probes, may assist in the understanding of the behaviour of a highly dynamic system, such as a French vertical wetland, and allow possible inferences of mechanisms that are associated with the system performance.

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DATA AVAILABILITY STATEMENT

All relevant data are included in the paper or its Supplementary Information.

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