Biochemical performance modelling of non-vegetated and vegetated vertical subsurface-flow constructed wetlands treating municipal wastewater in hot and dry climate

Short title: Modelling of constructed wetlands treating municipal wastewater

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A B S T R A C T

Wastewater treatment and subsequent effluent recycling for non-drinking purposes such as irrigation contributes to the mitigation of the pressure on freshwater resources. In this study, two vertical sub-surface flow constructed wetland (VSSF-CW) pilot plants were operated to treat municipal wastewater and their effluents were reused for irrigation purposes. One of the wetlands was vegetated with *Phragmites australis* (Cav.) Trin. ex Steud. (common reed) to compare its efficiency of pollutant removals with the non-vegetated system, which had the same design. COMSOL Multiphysics 3.5a was operated for the Activated Sludge Model 2 (ASM2) to predict the chemical oxygen demand (COD) and ammonia-nitrogen (NH_4-N) concentrations. The effluent quality of both treatment systems was assessed for several parameters. Computer simulations show a good compliance between the measured and predicted values of COD and NH_4-N for the vegetated system. The calibrated model could be effectively used to predict the behaviours of those parameters as a function of time. Moreover, the effluents of both vegetated (VFp) and non-vegetated (VF) VSSF-CW were significantly ($p < 0.05$) improved compared to influent. Significant ($p < 0.05$) effects due to the presence of *P. australis* were observed for removals of total suspended solids (TSS), 5-day biochemical oxygen demand (BOD_5), COD, NH_4-N and ortho-phosphate-phosphorus (PO_4-P). However, significant increases ($p < 0.05$) were noted for electrical conductivity (EC), total dissolved solids (TDS), nitrate-nitrogen (NO_3-N) and sulphate (SO_4) of both effluents compared to the raw wastewater. Except for EC, NH_4-N and SO_4, all water quality parameters complied with irrigation water standards.

Keywords:

Activated sludge model

Domestic wastewater

Nutrients

Phragmites australis

Reed bed

1. Introduction

In recent decades, the water demand has significantly increased due to factors such as world population growth and rapid industrial development. Consequently, the need for treatment and disposal of more wastewater has put pressure on the environment. Conventional wastewater treatment processes require high investment costs and have considerable operational energy demand (UNESCO, 2009). There has been a recent focus on treated wastewater to be used as an alternative water resource for non-potable uses such as irrigation, particularly in arid and semi-arid regions (Almuktar et al., 2018). Therefore, recycling of wastewater treated by sustainable methods such as constructed wetlands has gained attention worldwide for economic and environmental reasons.

Constructed wetlands have been introduced as an alternative sustainable technology for treating wastewater. The processes in wetlands are similar to those in traditional wastewater treatment plants (Sameanram, 2015). They have low construction and energy consumption costs, and require low effort in terms of operation and maintenance. Traditional energy consumption can be substituted by renewable energy resources such as solar and wind power (USEPA, 1995; Stefanakis and Tsihrintzis, 2009). Typically, treatment wetlands are defined as flooded land with a partially saturated permeable soil layer vegetated with aquatic macrophytes and lined with an impermeable layer. The hydraulic retention time (HRT) characterises how long wastewater is treated in the wetland system (Brix, 1994). This means that the HRT is a measure of the average length of time that a soluble compound remains in a constructed wetland. This parameter can be calculated by dividing the volume of the bed by the influent flowrate. Depending on water surface level characteristics, constructed wetlands can be classified as free surface flow (FSF) when the water surface is above the ground level. Otherwise, they are classified as sub-surface flow (SSF) wetlands (USEPA, 1995).

Furthermore, engineered SSF-CW might be managed as horizontal SSF (HSSF) or vertical SSF (VSSF) systems (Vymazal and Kröpfelová, 2009). Wetland ecosystems offer biological, chemical and physical processes of wastewater treatment for degradation of organic matter and other contaminants (Payne et al., 2015).

Influent wastewater within VSSF-CW is designed to flow vertically downward through the top layer, which is commonly planted with macrophytes and then infiltrated into a graded gravel bed containing the rhizosphere. After a specific HRT, wastewater is harvested through a perforated pipe network embedded at the bottom of the constructed wetland allowing oxygen to penetrate the system; therefore, improving the nitrification process and aerobic biodegradation of organic matter (Vymazal, 2014).

Most HSSF-CW are relatively low in terms of their oxygen diffusion capacity compared to VSSF-CW supporting nitrifying bacteria to oxidize NH_4-N (Stefanakis et al., 2014). Denitrification is not sufficiently accomplished in VSSF-CW (Al-Zreiqat et al., 2018). Therefore, a combination of two or more types of wetlands can be used as a hybrid system to enhance the biological and physiochemical processes in wastewater treatment (Almuktar et al., 2018).

Various types of constructed wetlands have been successfully used for the treatment of domestic, municipal, industrial and agricultural wastewater as well as road runoff (Vymazal and Kröpfelová, 2009; Kadlec and Wallace, 2009; Mustafa, 2013; Huang et al., 2015; Stanković, 2017). The effects of varying operational parameters such as vegetation, media size and type, and mode of sewage feeding have been investigated to assess the performance of the VSSF-CW, which has an effective volume of 0.0225 m^3, treating domestic sewage for an eight-month experiment. The results showed that the average removal efficiencies of COD, BOD, TSS, NH_4 and total phosphorus (TP) were 75, 84, 75, 32 and 22% for the planted beds compared to 29, 37, 42, 26 and 17%, respectively, for the unplanted beds. Also, *P. australis* had a significant effect ($p < 0.05$) on the removal efficiency and mass removal rate of all pollutants, except for phosphorous (Abdelhakeem et al., 2016). Four series of VSSF and HSSF wetlands as part of a hybrid system have been fed with artificial wastewater to evaluate nutrient removal and compare their performances under 0.5, 1, 2, and 4 days of HRT. The treatment efficiency of the hybrid systems showed a linear increase with retention time and the four-day HRT hybrid system was linked to the highest treatment among others considering all wastewater parameters. Organic treatment in the hybrid system was found to be very efficient even at smaller HRT with a range of mean treatment efficiency for BOD_5 between 88.5 and 92.7%. The removal efficiency of NH_4 observed in the hybrid system varied between 82.5 and 92.5%. While for NO_3 and NO_2, the treatment efficiency ranged between 30 and 47% and between 9.5 and 25%, respectively (Rasheed et al., 2014). Accordingly, the choices to go for the plant *P. australis* and the relatively long HRT in the present study were mainly based on the recommendations by Abdelhakeem et al. (2016) and Rasheed et al. (2014). Particularly *P. australis* can be found in huge quantities throughout Iraq.

Mathematical models have been developed to simulate the performance of natural treatment processes and pollutant removals to assess the behaviour of several types of set-ups (Samsó and Garcia, 2013). Modelling approaches can be divided into two categories: black-box models and process-based models (Kumar and Zhao, 2011). The first category is also known as data-driven modelling involving training and testing with data sets. The second category considers causes and effects as well as the individual processes occurring in a treatment unit (Taheriyoun and Rad, 2017). As process-based models are increasing the understanding of the processes involved in the wetland system, they may lead to improved wetland design and operation. In general, several models of varying complexity have been developed to describe the great variety of degradation and removal processes in wetlands (Langergraber, 2008). The ASM, constructed wetlands two dimensional (CW2D) model and the constructed wetlands model number 1 (CWM1) are the most recent process-driven codes that have been developed to simulate the treatment of urban wastewater. Furthermore, Samsó and Garcia (2013) used the COMSOL Multiphysics platform to implement the bio-kinetic equations of the CWM1, which describes bacteria-induced degradation and transformation processes of organic matter, nitrogen and sulphur with small changes to equations in order to include attachment and detachment of influent particulate components.

As the functioning of wetlands in terms of processes is still poorly understood and the applicability of the available models is still limited, the BIO_PORE model was built to help accelerating the development of constructed wetland models and to shed light on their internal functioning (Samsó and Garcia, 2013). Nutrient uptake and oxygen release by plant roots were also simulated. The new biofilm sub-model of the BIO_PORE model prevents unlimited growth of bacteria in areas with high substrate concentrations. It is able to reproduce the dynamic behaviour of constructed wetlands over long-term scenarios. Simulated effluent concentrations of COD and NH_4-N showed good agreement with experimental data. Water temperatures had a great impact on the model output, whereas the inclusion of plants did not cause noticeable differences.

Al-Isawi et al. (2015) has applied the Wang-Scholz Simulation Model (WSSM) for predicting removal efficiencies for municipal wastewater treated by different small laboratory-scale VSSF-CW planted with common reeds. The impact of operational and design variables was successfully evaluated.

The objectives of this paper are to (a) assess the performance of non-planted and planted pilot plant-scale VSSF-CW for municipal wastewater treatment; (b) compare the effluent water quality with international requirements to assess its suitability for irrigation; and (c) develop a numerical model using COMSOL Multiphysics 3.5a and the activated sludge model to predict the effluent characteristics of vegetated treatment systems.

2. Location, materials and methods

2.1. Site location and raw wastewater quality

The location of the pilot-scale VSSF-CW system was the Al-Rustumia Wastewater Treatment Plant (33° 17′ 15.41″ N, 44° 31′ 55.76″ E), which is situated on the eastern bank of the Tigris River, Baghdad, Iraq. The works serves the eastern zones of the Army Cannel Area including the New Baghdad District, the First and the Second Al-Sadar Neighborhood, Al-Ghadir and Al-Shaab Regions.

Both the VSSF-CW and the wastewater treatment plant were subjected to the same environmental boundary conditions and were fed with the same wastewater. Samples from both systems were assessed in the same on-site laboratory.

Raw wastewater to operate the VSSF-CW system was collected from the effluent of the primary treatment stage. The physio-chemical characteristics of raw wastewater, which fed the VSSF-CW set-up, are shown in Table 1.

2.2. Experimental design

The wetland treatment system consists of an electrical submersible pump, wastewater feeding tank, clean water feeding tank and two basins representing non-vegetated and vegetated VSSF-CW, which were designated as VFp and VF, respectively. A storage tank collecting treated wastewater was provided for subsequent recycling (irrigation in agriculture). All necessary pipe networks for feeding and distributing of wastewater, collecting the treated wastewater and aeration were fitted to operate the system appropriately.

Effluent from the primary settling tank was lifted by an electrical submersible pump of 1.5 horsepower (1119 Watt) to a polyvinyl chloride (PVC) holding tank of 1000 L, which was mounted on steel stands and elevated at a height of 2.5 m from the ground level to provide the required hydraulic head for feeding the treatment systems by gravity. Furthermore, another PVC tank of 500 L was provided to supply clean water for irrigation.

Wastewater was fed to both VSSF-CW by gravity from the feeding tank using a PVC distribution pipe network of 51-mm in diameter located along the wetland at 50 mm above the top surface layer of the wetland. The distribution pipe network consists of four PVC pipes arranged along the length of the wetland top layer and connected with each other by two lateral pipes. All distribution pipes were perforated uniformly with peripheral openings of 13-mm diameter with 100-mm spacings. The recommended opening diameter should not be less than 8 mm (Stanković, 2017). This position design allows for wastewater to be in contact with air via oxygen diffusion processes within the wetland substrate.

The two VSSF-CW had the same design parameters, except that one of them was planted with *P. australis*. Both wetlands consist of rectangular steel basins with lengths, widths and depths of 2.85 m, 1.2 m and 0.8 m, respectively. The inner basin walls were lined with fiberglass sheets to avoid direct interactions between the metals and the liquid.

Locally available inert gravel and sand were used to fill the wetland tanks. Aggregates were filled into the tanks in four layers following the design suggested by Nivala et al. (2013). The layers were organized from bottom to top of the basin as follows: 15-cm layer of coarse gravel (average diameter between 16 and 40 mm), 15-cm cover of medium-sized gravel (diameter between 8-16 mm), 20-cm layer of fine gravel (diameter of 4-8 mm), and finally 10-cm cover of coarse sand (1-3 mm in diameter). Therefore, the total depth of the multi-layered wetland bed was 60 cm, which is a typical depth for the root zone required for a *P. australis* plantation (Ellis et al., 2003). The remaining 20 cm of the upper basin were left as freeboard (Fig. 1).

During the vertical down-flow feeding of wastewater, the corresponding top water level was managed to be about 5 cm below the top surface layer to minimize water evaporation and prevent algal growth. The wastewater was gradually percolated through the wetland substrate by intermittent (batch) feeding, and the effluent was collected after three days of HRT. Twelve batch tests were conducted through the period from 15 January to 8 July 2018 with a retention time of three days and a resting time of two days. The duration between the end of any batch test and the beginning of the next one was two weeks.

The treated wastewater was harvested using PVC pipe networks located 15 cm above the bottom of the wetland beds, which equates to the border separated between the coarse and medium gravel layers as shown in Fig. 1b. The collecting network has a pipe system of 51-mm diameter perforated with hole diameters equal to 15 mm and spaced at distances of 100 mm from whole centre to centre. The perforated opening diameters of the collection pipe system were designated to be greater than the distribution system to avoid clogging. The collection network consisted of seven perforated PVC pipes installed along the width of the wetland bed at the bottom and connected to each other by two longitudinal manifolds. Aeration pipes of 51-mm diameter and extending above the saturated layer were provided for both wetland beds to achieve optimal oxygen transfer during the drainage stage and to establish a static and passive ventilation system.

Two PVC tanks of 500-L capacity were supplied to store the treated wastewater from each treatment system to be recycled for irrigation. The tanks were located on the concrete and steel deck with at a slope of 1% along the length of the basin to ensure flow of wastewater by gravity.

All pipes, tanks and appurtenances were purchased from a local market and characterized as heavy duty; i.e. able to resist bio-chemical attack and harsh weather conditions such as ultraviolet radiation and high temperatures as well as the heavy load of gravel. The vegetated pilot plant was planted with strong and mature *P. australis* macrophytes grown in pots and irrigated with clean tap water. The foliage was trimmed down from about 30 cm to 15 cm to encourage pollutant uptake during growth (Stefanakis et al., 2014). The planting stage of the VSSF-CW begun in September 2017 with an average density of 8 plants/m^2 (Toscano et al., 2009; Borin et al., 2013). Clean water was applied to the system until November 2017.

Thereafter, municipal wastewater was applied to both treatment units in November 2017. The VSSF-CW systems were left for two months to stabilise to allow for processes such as compaction of aggregates and development of biofilms. In January 2018, monitoring, water sampling and evaluation of the treatment wetland performance commenced.

During the operation period, the HRT was three days. In addition, two days were used as the resting time to allow for system aeration. Water samples were collected and measured twice per month until July 2018. This means that the batch tests were extended from January to July 2018. Stabilization of the plant density during the last three tests resulted in stopping the batch mode operation. The hydraulic loading rate was increased from 10.7 cm/day in the winter to 17.1 cm/day in the summer. The increase in loading rate addressed water losses due to evapotranspiration. The daily flow rate of wastewater of the Al-Rustumia Wastewater Treatment Plant (the location of the present experimental setup) increased from 90 L/capita in the winter to 145 L/capita in the summer.

2.3. Water quality evaluation

Raw and treated wastewater samples were collected in clean plastic bottles of 500-mL twice per month for inspection of their physical and bio-chemical parameters according to standard methods (APHA, 2012) unless stated otherwise.

A HACH LANG spectrophotometer was operated to analyse the wastewater samples for concentrations (mg/L) of TDS, TSS, COD, NH_4-N, NO_3-N, PO_4-P and SO_4. The EC and hydrogen ions expressed as pH value were measured using a German EC and pH meter

WTW InoLab 7110. The BOD_5 was examined using a German Lovibond BOD-System OxiDirect, which follows a manometric principle, where water dilution is not needed. The water temperature was recorded by a mercury thermometer.

2.4. Statistical evaluation

Summary statistics such as mathematical mean, standard deviation, minimum and maximum values of the measured water characteristics were calculated using Microsoft Excel. The IBM−SPSS Statistics Version 23.0 software was applied for more advanced statistical analysis of the results at 5% significance level. Normality of the data was assessed using the Shapiro-Wilk test. The independent t-test was performed to compare two independent variables when the corresponding data were normally distributed, while the Mann-Whitney U-test was used for non-normally distributed data.

2.5. Biochemical model

One of the basic assumptions made by Langergraber (2007) is that bacteria within constructed wetlands behave similar to those in activated sludge systems. Therefore, the parameters of the bio-kinetic models developed for activated sludge systems should also be applicable to processes within wetland systems. Microbial activities on organic compounds are well-captured in the functional system of the ASM versions. Therefore, the ASM1 and the ASM2 are still regarded as the state-of-the-art for the dynamic simulation of the removal of not only BOD but also the biologically degradable part of COD, biological nitrogen and phosphorus from the activated sludge systems (Langergraber and Morvannou, 2014). This means that the ASM can possibly also be used to simulate the biological strength and NH_4-N of waters in this study.

The ASM models primarily comprise the following three components: biomass, substrate and dissolved oxygen (DO). Moreover, it takes into consideration aerobic and anoxic processes that lead to the growth and decay of biomass (Giraldi et al., 2010). The kinetics or the rate equations explaining biomass growth are based on the Monod equations, whereby the biomass growth can be considered to be proportional to the biomass concentration in a first order manner, and to the substrate concentration in a mixed order manner (Langergraber et al., 2009).

The concentrations of dissolved components in the ASM are referred to as S_i and particulate components as X_i (Henze et al., 1999). Sixteen components (eight soluble and eight particulate ones) are considered in this model. The soluble components are DO (S_o, mg O_2/L), fermentable (biodegradable) soluble COD (S_F, mg COD/L), fermentation products (acetates) as COD (S_A, mg COD/L), inert soluble COD (S_I, mg COD/L), nitrite- and nitrate-nitrogen (S_{NO}, mg N/L), ammonium- and ammonia-nitrogen (S_{NH}, mg N/L), sulphate-sulphur (S_{SO4}, mg S/L), and dihydrogen-sulphide-sulphur (S_{H2S}, mg S/L). The particulate components adopted in the ASM model comprise slowly biodegradable particulate COD (X_s, mg COD/L), inert particulate COD (X_I, mg COD/L), heterotrophic bacteria (X_H, mg COD/L), autotrophic nitrifying bacteria (X_A, mg COD/L), fermenting bacteria (X_{FB}, mg COD/L), acetotrophic methanogenic bacteria (X_{AMB}, mg COD/L), acetotrophic sulphate-reducing bacteria (X_{ASRB}, mg COD/L), and sulphide-oxidizing bacteria (X_{SOB}, mg COD/L).

Seventeen processes in the ASM are seen as the most relevant biochemical transformation and degradation processes occurring in HF and VF wetland systems. They are required to predict the effluent concentrations according to Henze et al. (1999) and Langergraber et al. (2009).The following processes are important: hydrolysis, aerobic growth of X_H on S_F, aerobic growth of X_H on S_A, anoxic growth of X_H on S_F, anoxic growth of X_H on S_A, lysis of X_H, aerobic growth of X_A on S_{NH}, lysis of X_A, growth of X_{FB}, lysis of X_{FB}, growth of X_{AMB}, lysis of X_{AMB}, growth of X_{ASRB}, lysis of X_{ASRB}, aerobic growth of X_{SOB} on S_{H2S}, anoxic growth of X_{SOB} on S_{H2S}, and lysis of X_{SOB}. The stoichiometric matrix of the ASM, the process rates, the kinetic parameters and the stoichiometric as well as composition parameters are discussed elsewhere (Henze et al., 2000; Langergraber et al. 2009). The stoichiometric matrix and some of the above parameter are required as essential input for the COMSOL reaction lab package. The output data of the ASM-COMSOL model are the concentrations of COD and ammonia-nitrogen.

3. Results and discussion

3.1. Monitoring of *Phragmites australis*

Initially, the height of the plants was approximately 0.3 m. They were trimmed to a height of 0.15 m and planted in the experimental units to ensure rapid growth. The density of *P. australis* in the vegetated treatment unit increased considerably from 22 plants/m^2 at the beginning of experiment to reach 108 plants/m^2 after six months with an average height not less than 1.5 m. However, some of the foliage suffered (dry yellow leaves) due to a temperature of about 50°C during the summer season. The trimming of these stems and returning them to the VSSF-CW as organic matter to be degraded (mimicking natural processes) could have been practiced (Stefanakis et al., 2014).

3.2. Electrical conductivity, pH and solids

The characteristics of raw wastewater that was used as influent to both VSSF-CW systems are shown in Table 1. During summer, the EC of the influent was between 3.5 and 3.8 mS/cm, which is significantly ($p < 0.05$) elevated. Furthermore, insignificant ($p > 0.05$) rises in EC were generally noticed for the treated wastewater from both VFp and VF (Table 2). The effluent EC of VFp was lower than the one for VF, but this finding was not statistically significant ($p > 0.05$) as shown in Table 3. According to FAO (2003), the overall EC values of the raw wastewater and effluents from both treatment systems were greater than the threshold of 3.0 mS/cm), which is seen as serious in terms of a negative impact on soil structure and the adsorption capability of macrophytes concerning required water and nutrients (Ayers and Westcot, 1994).

The monthly pH mean values were neutral. However, the statistical analysis indicated significant increases ($p < 0.05$) of pH values for March and June 2018. The overall pH value of the influent was 7.3±0.15 (mean ± standard deviation) with minimum and maximum range limits of 7.1 and 7.6, respectively (Table 1a). The monthly water quality evaluation of effluents from both VSSF-CW revealed that the increases in the measured pH were not significant ($p > 0.05$) compared to the corresponding pH values of the influent (Tables 1 and 2). The increases in pH values can be explained by aerobic biodegradation processes of organic matter in wastewater producing hydroxide and carbonate anions (Morari and Giardini, 2009). Excessive productions of these ions also increases EC, salinity, hardness and alkalinity.

Interactions among wetland components such as wastewater, organisms, media and pollutant loads could lead to unanticipated findings (Prochaska et al., 2007). Although it has been reported that the presence of plants in wetlands are responsible for buffering the pH of wastewater (Morari and Giardini, 2009), no statistically significant differences ($p > 0.05$) between the effluents of VFp and VF in terms of pH were noted (Table 3). The pH values of both raw and treated wastewater were within the threshold limits of 6.5 and 8.5 for reuse in agricultural irrigation (FAO, 2003; WHO, 2006).

The mean TSS inflow concentration was 74.7 mg/L (Table 1). Both systems exhibited significant reductions ($p < 0.05$) in TSS concentrations after treatment. Moreover, the vegetated treatment system showed a TSS removal efficiency of around 65.5%, which was significantly ($p < 0.05$) greater than the TSS removal of the unplanted system (Table 3). The main removal processes in SSF-CW are filtration and sedimentation (Vymazal, 2002).

Temperature impacts also on TSS. Suspended particles dissolve at high temperatures, and they may increase due to evapotranspiration. Hence, a strong macrophyte canopy reduces the effect of high temperatures (Zurita et al., 2009). Although TSS of the influent was high during summer, this did not impact on the corresponding removal efficiency for both treatment systems (Figs. 2a and 2b), which confirms findings by Sani et al. (2013).

The removal efficiencies of the planted beds were greater than the efficiencies of the unplanted beds with values ranging from 10 to 20%. This can be attributed to the significant role of *P. australis* roots in the filtration process (Zurita et al., 2009). Biological adsorption and assimilation reduce TSS. Collides and solids can be trapped within the biofilm located on the roots and rhizomes. Moreover, electrically charged root hairs may attract solids of opposite charge as well (Brix, 1994).

Clogging of the pilot plants was not noticed. The TSS inflow concentrations were low (Table 1). Moreover, the effluents from VFp and VF were always less than 100 mg/L, whereas there might be challenge to clogging of agricultural land, if the TSS of recycled treated wastewater is well above this threshold according to WHO (2006).

Statistically significant increases ($p < 0.05$) were observed for concentrations of TDS in the effluents of both systems indicating that there is no salinity removal for treated wastewater using constructed wetland (Table 2). A comparison between Figs. 2c and 2d only indicates a slight improvement for the vegetated wetland. Organic and inorganic dissolved solids with high bioavailability are removed by plants and microorganisms (via metabolism processes) as noted elsewhere (Hench et al., 2003; Kouki et al., 2009). The presence of high numbers of microbes is associated with the surface area of gravel, roots and rhizomes (USEPA, 1995).

Statistical analysis shows a significantly positive correlation ($p < 0.01$; $r = 1.00$) between EC and TDS for both the vegetated and non-vegetated wetlands. Elevated concentrations of TDS in the effluent are indicative of potentially high salinity damaging agricultural soil, if the water is used for irrigation. The TDS concentrations for both influent and effluents were within the range of 450 to 2000 mg/L (FAO, 2003), which indicates a slight to moderate effect on soil (Ayers and Westcot, 1994).

3.3. Oxygen demand

The removal of organic matter was characterised in terms of BOD_5 and COD before and after wastewater treatment. The influent for both systems had COD and BOD_5 concentrations of 281.3 and 145.5 mg/L, respectively (Table 1). The monthly effluent COD and BOD_5 concentrations were significantly ($p < 0.05$) lower than the corresponding values for the influent. The effluent values fluctuated between 21.9 and 49.0 mg COD/L and between 10.0 and 27.0 mg BOD_5/L for the planted wetland. In comparison, the corresponding ranges for the unplanted rig were 28.6-81.4 mg for COD/L and 13.0-40.0 mg for BOD_5/L (Table 2). These results are considerably lower than the WHO (2006) threshold of 110 mg BOD_5/L for the safe use of municipal wastewater for irrigation. Both treatment systems showed significantly ($p < 0.05$) high reductions in terms of oxygen demand. The vegetated system performed significantly ($p < 0.05$) better in the removal of oxygen demand compared to the non-vegetated one (Table 3 and Figs. 3a-d). The removal results were higher than corresponding published findings for similar systems (Brix et al., 2007; Prochaska et al. 2007; Morari and Giardini, 2009).

Organic substances within vegetated wetlands are removed due to aerobic biodegradation by microbes supported by macrophytes transferring oxygen to the water within the rhizosphere (Ciria et al., 2005; Kouki et al., 2009; Taylor et al., 2011). The statistical analysis of the wastewater treated by both systems revealed significantly positive correlations ($p < 0.01$ and $r = 0.89$ for VFp; and $p < 0.01$ and $r = 0.91$ for VF) between COD and BOD_5 due to the relatively high proportion of organic matter.

3.4. Nutrients

The overall ammonia-nitrogen (NH_4-N) concentrations of raw wastewater ranged between 79.8 and 103 mg/L. These high values indicate agricultural activity in the areas served by the Al-Rustumia treatment plant. The monthly feeding means of NH_4-N for both treatment systems were not significantly ($p > 0.05$) different throughout the experimental time period (Table 1). Subsequently, the treatment systems showed steady performances with significantly ($p < 0.05$) high reductions in NH_4-N (Figs. 3e and 3f). A significantly ($p < 0.05$) high removal of NH_4-N was observed in the vegetated compared to the non-vegetated system (Table 3). The key removal mechanism of NH_4-N in VSSF-CW is nitrification (Vymazal, 2007; Rasheed et al., 2014). The water flow regime, artificial aeration and the transfer of oxygen by the macrophyte root system into the substrate layer supported nitrifying bacteria in the oxidation of ammonium (NH_4^+) to nitrate (NO_3^-), with nitrite (NO_2^-) as an intermediate reaction (Brix, 1994; Ciria et al., 2005). Assimilation processes of macrophytes and microorganisms utilized dissolved and inorganic nitrogen within water and substrate (Kadlec and Wallace, 2009). The uptake of ammonia by aquatic organisms occurs when nitrification is restricted (Vymazal, 2007). Therefore, the significant removal of NH_4-N within the planted system can be explained by the role of *P. australis* in oxygen diffusion and provision of habitat in their root system for microorganism to develop. However, the NH_4-N content in treated wastewater from both systems was higher than the FAO (2003) limit of 5 mg/L, which has been set for reuse of treated wastewater in agricultural irrigation.

Almost all NH_4-N was transferred to NO_3-N due to nitrification during the treatment of municipal wastewater in both systems. The overall average concentrations of NO_3-N were increased significantly ($p < 0.05$) from 1.6±0.41 mg/L (mean ± standard deviation) of the raw wastewater to 15.9±3.63 mg/L in the planted and 13.6±3.37 mg/L in the unplanted wetland (Tables 1 and 2). The statistical analysis revealed that the presence of vegetation significantly ($p < 0.05$) increased NO_3-N in the effluents (Table 3).

A comparison between Figs. 3g and 3h shows that the presence of *P. australis* led to an increase of NO_3-N. An indirect effect of macrophyte presence was evident from reduced TSS and increased TDS concentrations. Moreover, NO_3-N correlated positively and significantly ($p = 0.002$ and $r = 0.805$) with TDS. The NO_3-N concentration of the planted system was significantly ($p < 0.05$) higher compared to the effluent of the unplanted system. Both effluents were significantly ($p < 0.05$) greater than the influents in terms of NO_3-N (Tables 1 and 3).

In general, denitrification is the common biological process to convert NO_3-N to ionic and gaseous nitrogen (Vymazal, 2007; Rasheed et al., 2014). This process takes place if organic matter is available at anaerobic or anoxic conditions. So, denitrification should be limited in the VSSF-CW, which is a well-aerated (Almuktar et al., 2018).

Macrophytes and microorganisms assimilate nitrogen for their growth (Kadlec and Wallace, 2009). Although it has been demonstrated that NH_4-N is energetically more desirable in biological respiration processes compared to NO_3-N, waters high in NO_3-N are still important nitrogen sources (Soto et al., 1999). From Table 2, it can be seen that the NO_3-N concentrations of both treatment wetland effluents complied with the range (5 to 30 mg NO_3-N/L) stated by FAO (2003), indicating a slight to moderate effect on soil and crops when the treated wastewater is recycled for agricultural irrigation (Ayers and Westcot, 1994).

The vegetated treatment system was significantly ($p < 0.05$) more efficient in PO_4-P removal compared to the non-vegetated one (Table 3; Figs. 3i and 3j). Despite its role as a vital nutrient for plants, excessive phosphorus concentrations in aquatic ecosystems can be the main source for eutrophication and algal blooms (Chung et al., 2008; Rasheed et al., 2014). Ortho-phosphate-phosphorus reductions in the current study exceeded the expected performance based on literature values that are commonly between 20 and 30% (Brix et al., 2007). This decrease in the phosphate concentration can attribute to the adsorption capacity of the media as well as the plant and microorganism uptake. The removal mechanisms for PO_4-P are commonly adsorption, precipitation as well as plant and microorganism uptake (Vymazal, 2007; Rasheed et al., 2014). According to FAO (2003), the total phosphorus allowed in wastewater to be reused safely for irrigation is below 2 mg/L. In comparison, USEPA (2004) limits the total phosphorus at 5.0 mg/L for irrigation of crops. Furthermore, WHO (2006) states that a total phosphorus concentration between 6 and 20 mg/L in municipal wastewater has no reported negative effect, if reused for irrigation.

In this study, SO_4 was unsuccessfully treated by VSSF-CW compared to other parameters such as NH_4-N and PO_4-P, as shown in Figs. 3k and 3l. Concentrations of SO_4 raised from 549.3 mg/L in the raw wastewater to 557.0 mg/L and 588.0 mg/L for the effluents of the planted and unplanted systems, respectively (Tables 1 and 2). Organically-bound sulphur present in the water column is transformed to inorganic sulphur compounds. Three likely processes responsible for this are the conversions of sulphide to sulphate by sulphur bacteria, sulphide to sulphur by sulphur bacteria, and sulphur to sulphate by oxidation (Mehdizadeh and Shaigan, 2003).

In addition, sulphur might be released from dead macrophytes located on top of the surface of the planted system (Kadlec and Wallace, 2009). The SO_4 concentration has been limited by FAO (2003) for reclaimed municipal wastewater to be below 20 mg/L for safe reuse in crop irrigation.

3.5. Evaluation of the biochemical model

The simulation study was performed to validate the bio-kinetic parameter set using a load series with twelve individual loads. The targets of fitting were measured COD and NH_4-N concentration series of the effluent. The possibly relevant parameters based on the present mathematical formulations of the bio-kinetic model (ASM2) developed in conjunction with COMSOL Multiphysics 3.5a were anticipated to be (a) µH that is the maximum aerobic growth rate of X_H on readily biodegradable COD (X_F) (Samsó, 2014); (b) bH, which is the rate constant of lysis for X_H; (c) rate_O2: re-aeration rate; (d) Y_H that is the yield coefficient for X_H (heterotrophs), X_{FB} and X_{SOB}; and (e) X_F:S_A:X_I, which are the fractionation of the COD load to readily and slowly biodegradable as well as inert substances.

Calibration of the studied model can be carried out by comparing the observed and simulated data with each other, and subsequently trying to minimise the differences between them. Manual or mathematical calibration can be carried out for the model using visualization or statistical measures, respectively. In this study, manual calibration of the model has been adopted, while kinetic stoichiometric and other parameters, which are linked to the biochemical processing within the wetland, were taken into consideration (Keskinler and Yildiz, 2005). Adjustment of the values of model parameters continued until the best match between observed and simulated data was found.

Table 4 summaries the bio-kinetic parameter set adopted from Samsó (2014) and their values to achieve the acceptable agreement between the predicted and measured values of the target pollutants. Figure 4 presents some results of representative simulations showing time variations of COD and ammonia-nitrogen during a simulated filling and empting cycle of a wetland. The findings indicate that there is a good fit between the measured and predicted values of COD and NH_4-N. Hence, the calibrated model can be used successfully for assessing the behaviour of pollutants (e.g., occurrence of concentration maxima) for each cycle.

4. Conclusions and recommendations

The vegetated and non-vegetated treatment wetlands showed significant removal performances for all of the considered water quality parameters except for NO_3-N, sulfate and TDS. The presence of *P. australis* significantly improved the removal efficiencies of COD (86.0%), BOD_5 (85.6%), NH_4-N (82.1%), PO_4-P (59.6%) and TSS (65.5%). The density of *P. australis* in terms of number of stems and their corresponding heights increased during the experiment.

The effluents of both wetlands were suitable for irrigation in terms of pH, TSS, TDS, COD, BOD_5, NO_3-N and PO_4-P according to international guidelines and legislation published by organisations such as the Food and Agriculture Organization of the United Nations and the World Health Organization. However, the values of EC, NH_4-N and SO_4 obtained in this study were above commonly recommended limits.

COMSOL Multiphysics 3.5a was successfully used in combination with ASM2 for predicting COD and NH_4-N effluent concentration series. The calibrated model can be used for further optimization of the design and operation of the VSSF-CW unit by application of different (shorter) HRT and other initial concentrations of key water quality parameters. The results revealed that VSSF-CW are adequate for the treatment of municipal wastewater as a secondary treatment unit in the hot climate of central Iraq.

The performance of the vegetated treatment system was sufficient to allow for its effluent to be recycled for irrigation in agriculture. Nevertheless, the authors recommend to introduce a hybrid treatment system comprising a horizontal SSF-CW treating the effluent of a vegetated VSSF-CW to reduce TSS, EC, TDS and NO_3-N further.

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Table 1

Raw wastewater quality parameters for the whole experiment period (January 2018 to July 2018).

Monthly influent quality (mean value)								
Parameter	Unit	January	February	March	April	May	June	July
Chemical oxygen demand	mg/L	146.0	274.5	234.0	310.0	320.5	318.5	315.0
Biochemical oxygen demand	mg/L	77.0	163.0	104.0	139.5	172.5	171.0	169.0
Ammonia-nitrogen	mg/L	87.0	87.5	92.0	99.4	84.6	89.7	84.8
Nitrate-nitrogen	mg/L	1.2	1.4	2.0	1.7	1.8	1.6	1.5
Ortho-phosphate-phosphorus	mg/L	4.2	3.9	3.5	4.1	3.9	4.2	3.6
Sulphate	mg/L	480.0	572.5	515.5	595.5	508.5	559.0	609.0
Total suspended solids	mg/L	30.0	60.0	74.5	71.0	103.5	83.0	91.0
Electrical conductivity	mS/cm	2.6	3.0	3.1	3.3	2.8	3.5	3.8
Total dissolved solids	mg/L	1290.0	1510.0	1540.0	1623.0	1378.0	1738.0	1910.0
pH	-	7.1	7.2	7.4	7.2	7.2	7.4	7.2

Overall influent quality (mean value)						
Parameter	Unit	Number	Mean	Standard deviation	Minimum	Maximum
Chemical oxygen demand	mg/L	12	281.3	60.21	146.0	346.0
Biochemical oxygen demand	mg/L	12	145.5	35.71	77.0	177.0
Ammonia-nitrogen	mg/L	12	89.9	7.56	79.8	103.0
Nitrate-nitrogen	mg/L	12	1.6	0.41	0.9	2.3
Ortho-phosphate-phosphorus	mg/L	12	3.9	0.67	2.9	4.9
Sulphate	mg/L	12	549.3	48.57	480.0	619.0
Total suspended solids	mg/L	12	74.7	22.89	30.0	126.0
Electrical conductivity	mS/cm	12	3.1	0.40	2.5	3.8
Total dissolved solids	mg/L	12	1564.6	198.5	1240.0	1910.0

pH	-	12	7.3	0.15	7.1	7.6
Water temperature	^{o}C	12	24.2	6.00	14.4	33.8
Air temperature (maximum)	^{o}C	12	30.4	9.84	18.3	48.3
Air temperature (minimum)	^{o}C	12	15.3	8.32	3.3	30.0

Table 2

Effluent water quality parameters for the whole experiment period (January 2018 to July 2018).

Monthly effluent quality (mean value)								
Parameter	Unit	January	February	March	April	May	June	July
Vegetated system								
Chemical oxygen demand	mg/L	31.0	44.5	35.5	48.6	32.2	38.0	32.0
Biochemical oxygen demand	mg/L	16.0	26.5	16.0	22.0	17.5	20.5	17.0
Ammonia-nitrogen	mg/L	19.0	14.0	17.4	15.5	12.2	16.3	23.0
Nitrate-nitrogen	mg/L	11.1	13.6	16.0	13.9	15.4	21.5	18.8
Ortho-phosphate-phosphorus	mg/L	1.5	1.6	1.4	1.7	1.6	1.9	1.4
Sulphate	mg/L	521.0	594.5	491.5	599.0	519.5	571.0	621.0
Total suspended solids	mg/L	12.0	20.5	20.5	28.0	40.5	25.5	27.0
Electrical conductivity	mS/cm	2.4	3.0	3.3	3.3	3.1	3.6	3.9
Total dissolved solids	mg/L	1205.0	1510.0	1622.5	1660.0	1550.0	1805.0	1955.0
pH	-	7.3	7.4	7.6	7.4	7.3	7.7	7.3
Non-vegetated system								
Chemical oxygen demand	mg/L	37.0	58.9	45.8	72.1	46.9	52.9	48.0
Biochemical oxygen demand	mg/L	19.0	35.0	20.5	32.5	25.0	28.5	26.0
Ammonia-nitrogen	mg/L	28.0	22.1	25.2	22.9	19.0	17.6	29.0
Nitrate-nitrogen	mg/L	9.7	10.8	14.8	11.4	13.0	18.5	16.7
Ortho-phosphate-phosphorus	mg/L	2.1	2.3	2.3	2.9	2.5	2.7	2.5
Sulphate	mg/L	610.0	650.5	537.0	612.0	536.0	577.0	630.0
Total suspended solids	mg/L	18.0	24.0	27.5	34.5	49.0	32.9	31.0
Electrical conductivity	mS/cm	2.6	3.2	3.2	3.4	3.1	3.7	4.0
Total dissolved solids	mg/L	1315.0	1582.5	1602.5	1680.0	1540.0	1842.5	1995.0

pH	-	7.4	7.6	7.67	7.4	7.48	7.7	7.5

Overall effluent quality (mean value)

Parameter	Unit	Number	Mean	Minimum	Maximum	Standard deviation	Removal %
Vegetated system							
Chemical oxygen demand	mg/L	12	38.4	21.9	49.0	9.68	86.0
Biochemical oxygen demand	mg/L	12	19.8	10.0	27.0	5.08	85.9
Ammonia-nitrogen	mg/L	12	16.0	10.2	23.0	3.53	82.1
Nitrate-nitrogen	mg/L	12	15.9	10.9	22.6	3.63	-928.6
Ortho-phosphate-phosphorus	mg/L	12	1.6	1.1	2.1	0.36	59.6
Sulphate	mg/L	12	557.0	465.0	640.0	52.94	-1.4
Total suspended solids	mg/L	12	25.8	12.0	54.0	10.13	65.5
Electrical conductivity	mS/cm	12	3.2	2.4	3.9	0.40	na
Total dissolved solids	mg/L	12	1621.3	1205.0	1955.0	199.98	-3.9
pH	-	12	7.4	7.3	7.8	0.18	na
Non-vegetated system							
Chemical oxygen demand	mg/L	12	53.2	28.6	81.4	14.86	80.8
Biochemical oxygen demand	mg/L	12	27.3	13.0	40.0	7.61	80.9
Ammonia-nitrogen	mg/L	12	22.5	14.3	32.0	5.20	74.9
Nitrate-nitrogen	mg/L	12	13.6	8.9	19.0	3.37	-775.0
Ortho-phosphate-phosphorus	mg/L	12	2.5	1.7	3.6	0.51	36.0
Sulphate	mg/L	12	588.8	517.0	700.0	51.75	-7.4
Total suspended solids	mg/L	12	32.1	18.0	67.0	12.29	56.4
Electrical conductivity	mS/cm	12	3.3	2.6	4.0	0.38	na
Total dissolved solids	mg/L	12	1650.4	1315.0	1995.0	189.53	-5.7
pH	-	12	7.6	7.4	7.8	0.15	na

Note: na, not applicable.

Table 3

Assessment of the statistically significant differences between the overall effluent water quality for vegetated and non-vegetated systems

Parameter (mean value)			
Parameter	Shapiro-Wilk test (p-value)[a]	Statistical test	p-value[b]
Chemical oxygen demand	0.678	I-T	0.008
Biochemical oxygen demand	0.944	I-T	0.010
Ammonia-nitrogen	0.374	I-T	0.002
Nitrate-nitrogen	0.811	I-T	0.125
Ortho-phosphate-phosphorus	0.466	I-T	<0.001
Sulphate	0.831	I-T	0.152
Total suspended solids	0.000	M-W-U	0.049
Electrical conductivity	0.614	I-T	0.717
Total dissolved solids	0.614	I-T	0.717
pH	0.071	I-T	0.092
Removal efficiency of pollutant			
Parameter	Shapiro-Wilk test (p-value)[a]	Statistical test	p-values[b]
Chemical oxygen demand	0.678	I-T	0.006
Biochemical oxygen demand	0.593	I-T	0.006
Ammonia-nitrogen	0.691	I-T	0.001
Ortho-phosphate-phosphorus	0.028	M-W-U	<0.001
Total suspended solids	0.041	M-W-U	0.011

[a] Test of normality (if p-value > 0.05, data are normally distributed; if p-value < 0.05, data are not normally distributed.

[b] p-value, probability of the statistical test.

Note: Systems are statistically significantly different only if the p-value < 0.05 for the corresponding water quality parameter. M-W-U, the non-parametric Mann–Whitney U-test, and I-T, parametric Independent Sample T-test.

Table 4

Kinetic parameters adjusted for simulation by the bio-chemical model using COMSOL Multiphysics 3.5a.

	Parameter	Description	Value
Hydrolysis	K_h	Hydrolysis rate constant (1/day)	3
	K_X	Saturation/inhibition coefficient for hydrolysis (g COD_{SF}/g COD_{BM})	0.1
	η_H	Correction factor for hydrolysis by fermenting bacteria (-) Heterotrophic bacteria (aerobic growth and denitrification)	0.1
	μ_H	Maximum aerobic growth rate on S_F and S_A (1/day)	6
	η_g	Correction factor for denitrification by heterotrophs (-)	0.8
	b_H	Rate constant for lysis (1/day)	0.4
	K_{OH}	Saturation/inhibition coefficient for SO (mg O_2/L)	0.2
	K_{SF}	Saturation/inhibition coefficient for S_F (mg COD_{SF}/L)	0.0001
	K_{SA}	Saturation/inhibition coefficient for S_A (mg COD_{SA}/L)	4
	K_{NOH}	Saturation/inhibition coefficient for S_{NO} (mg N/L)	0.5
	K_{NHH}	Saturation/inhibition coefficient for S_{NH} (nutrient) (mg N/L)	0.05
	K_{H2SH}	Saturation/inhibition coefficient for S_{H2S} (mg S/L)	140
	μ_A	Maximum aerobic growth rate on S_{NH} (1/day)	1
	b_A	Rate constant for lysis (1/day)	0.4
	K_{OA}	Saturation/inhibition coefficient for SO (mg O_2/L)	0.2
	K_{NHA}	Saturation/inhibition coefficient for S_{NH} (mg N/L)	0.5
	K_{H2SA}	Saturation/inhibition coefficient for S_{H2S} (mg S/L)	140
Fermenting bacteria	μ_{AMB}	Maximum aerobic growth rate for X_{FB} (1/day)	3
	b_{AMB}	Rate constant for lysis (1/day)	0.02
	K_{SFB}	Saturation/inhibition coefficient for S_F (mg COD_{SF}/L)	2
	K_{OFB}	Saturation/inhibition coefficient for SO (mg O_2/L)	0.2
	K_{NOFB}	Saturation/inhibition coefficient for S_{NO} (mg N/L)	0.5

	K_{NHFB}	Saturation/inhibition coefficient for S_{NH} (nutrient) (mg N/L)	0.05
	K_{H2SFB}	Saturation/inhibition coefficient for S_{H2S} (mg S/L)	140
Acetotrophic methanogenic bacteria	μ_{AMB}	Maximum aerobic growth rate on for X_{AMB} (1/day)	0.085
	b_{AMB}	Rate constant for lysis (1/day)	0.008
	K_{OAMB}	Saturation/inhibition coefficient for SO (mg O_2/L)	0.0002
	K_{SAMB}	Saturation/inhibition coefficient for S_F (mg COD_{SA}/L)	56
	K_{NOAMB}	Saturation/inhibition coefficient for S_{NO} (mg N/L)	0.0005
	K_{H2SAMB}	Saturation/inhibition coefficient for S_{H2S} (mg S/L)	140
	K_{NHAMB}	Saturation/inhibition coefficient for S_{NH} (nutrient) (mg N/L)	0.01

Note: Parameters of acetotrophic sulphate reducing bacteria and sulphide oxidizing bacteria were adopted from Samsó (2014).
COD_{SF}, soluble fermentable (biodegradable) chemical oxygen demand (mg/l); COD_{BM}, biomass chemical oxygen demand (mg/L); S_F , soluble fermentable (and biodegradable) (mg/L); S_A , soluble acetate fermentation products (mg/L) ; S_O, dissolved oxygen (mg/L); S_{NO}, soluble nitrite and nitrate nitrogen (mg/L); N, nitrogen (mg/L); S_{NH} , soluble ammonium and ammonia nitrogen (mg/L); S_{H2S} , soluble dihydrogen-sulphide sulphur (mg/L); S, sulphur (mg/L).

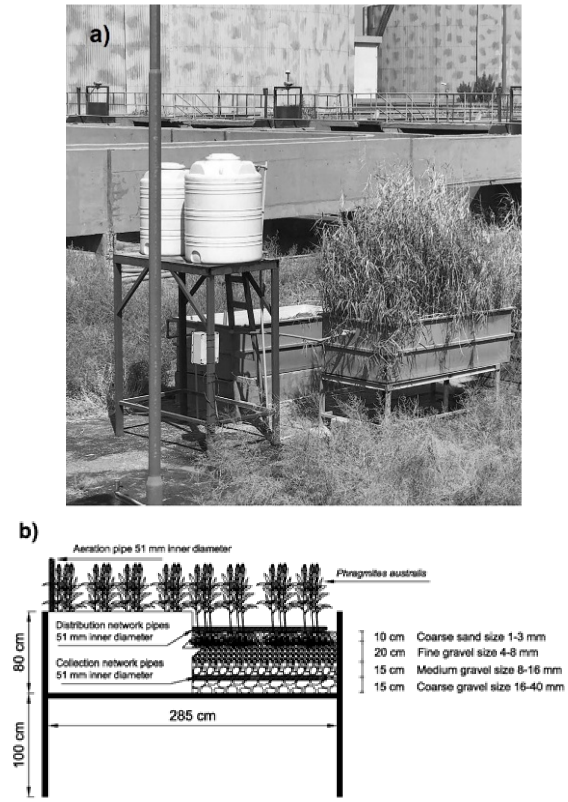


Fig. 1. Pilot-scale vertical sub-surface flow constructed wetland: (a) photograph; and (b) schematic diagram.

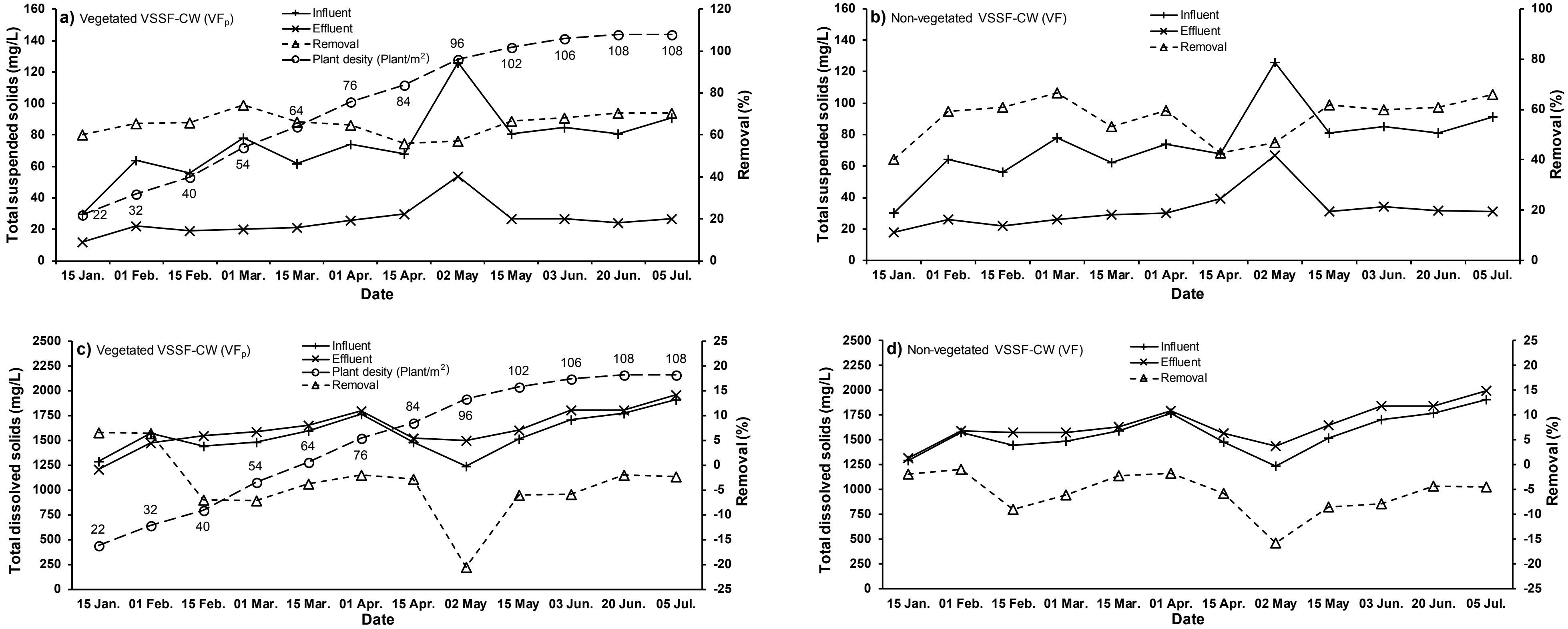
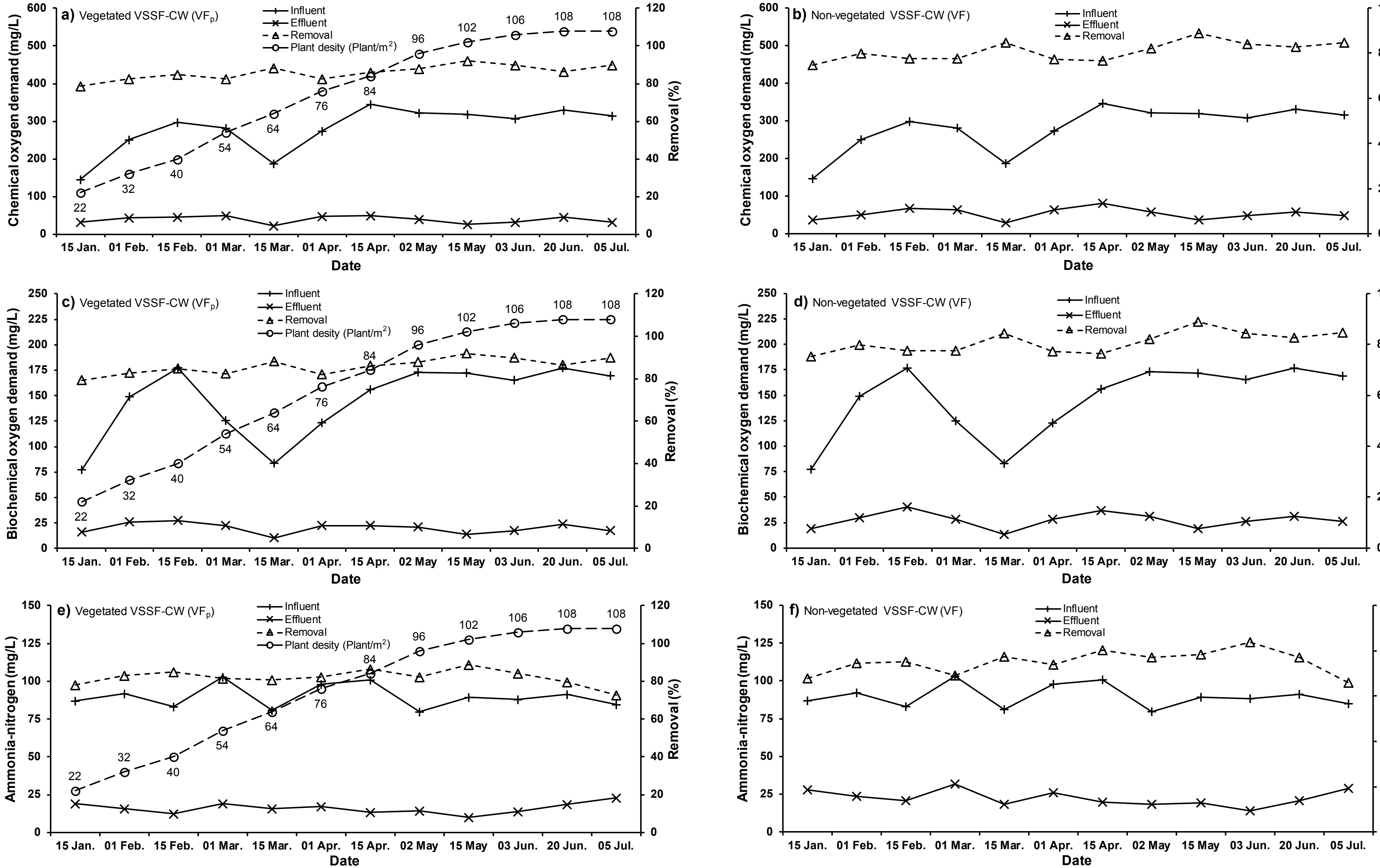


Fig. 2. Variation of pollutant concentrations and removals during the entire experimental operation time for vertical sub-surface flow constructed wetlands (VSSF-CW): (a) total suspended solids (TSS) for the planted (VF_p) system; (b) TSS or the non-planted (VF) system; (c) total dissolved solids (TDS) for VF_P; and (d) TDS for VF.



a) Vegetated VSSF-CW (VFp)
Influent
Effluent
Removal
Plant desity (Plant/m2)
Chemical oxygen demand (mg/L)
Removal (%)
Date
b) Non-vegetated VSSF-CW (VF)
c) Vegetated VSSF-CW (VFp)
Biochemical oxygen demand (mg/L)
d) Non-vegetated VSSF-CW (VF)
e) Vegetated VSSF-CW (VFp)
Ammonia-nitrogen (mg/L)
f) Non-vegetated VSSF-CW (VF)
15 Jan.
01 Feb.
15 Feb.
01 Mar.
15 Mar.
01 Apr.
15 Apr.
02 May
15 May
03 Jun.
20 Jun.
05 Jul.

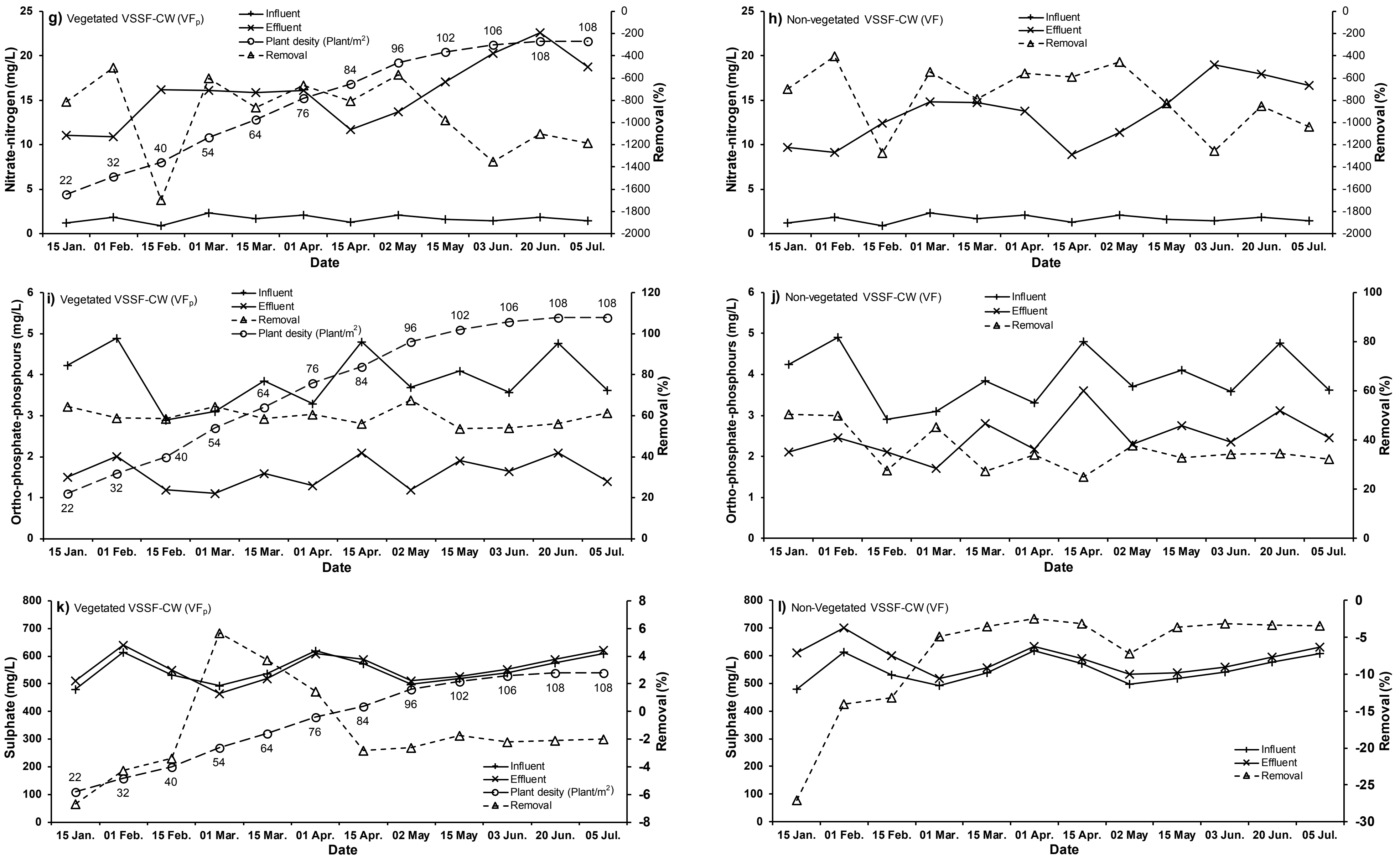

g) Vegetated VSSF-CW (VF_p)
Influent
Effluent
Plant desity (Plant/m^2)
Removal
Nitrate-nitrogen (mg/L)
Removal (%)
Date
22
32
40
54
64
76
84
96
102
106
108
108
15 Jan.
01 Feb.
15 Feb.
01 Mar.
15 Mar.
01 Apr.
15 Apr.
02 May
15 May
03 Jun.
20 Jun.
05 Jul.
h) Non-vegetated VSSF-CW (VF)
Influent
Effluent
Removal
Nitrate-nitrogen (mg/L)
Removal (%)
Date
i) Vegetated VSSF-CW (VF_p)
Influent
Effluent
Removal
Plant desity (Plant/m^2)
Ortho-phosphate-phosphours (mg/L)
Removal (%)
Date
j) Non-vegetated VSSF-CW (VF)
Influent
Effluent
Removal
Ortho-phosphate-phosphours (mg/L)
Removal (%)
Date
k) Vegetated VSSF-CW (VF_p)
Influent
Effluent
Plant desity (Plant/m^2)
Removal
Sulphate (mg/L)
Removal (%)
Date
l) Non-Vegetated VSSF-CW (VF)
Influent
Effluent
Removal
Sulphate (mg/L)
Removal (%)
Date

Fig. 3. Variation of pollutant concentrations and removals during the entire experimental operation time for vertical sub-surface flow constructed wetlands (VSSF-CW): (a) chemical oxygen demand (COD) for the planted (VF_p) system; (b) COD or the non-planted (VF) system; (c) 5-day biochemical oxygen demand (BOD_5) for VF_P; (d) BOD_5 for VF; (e) ammonia-nitrogen (NH_4-N) for VF_P; (f) NH_4-N for VF; (g) nitrate-nitrogen (NO_3-N) for VF_P; (h) NO_3-N for VF; (i) ortho-phosphate-phosphorus (PO_4-P) for VF_P; (j) PO_4-P for VF; (k) sulphate (SO_4) for VF_P; and (l) SO_4 for VF.

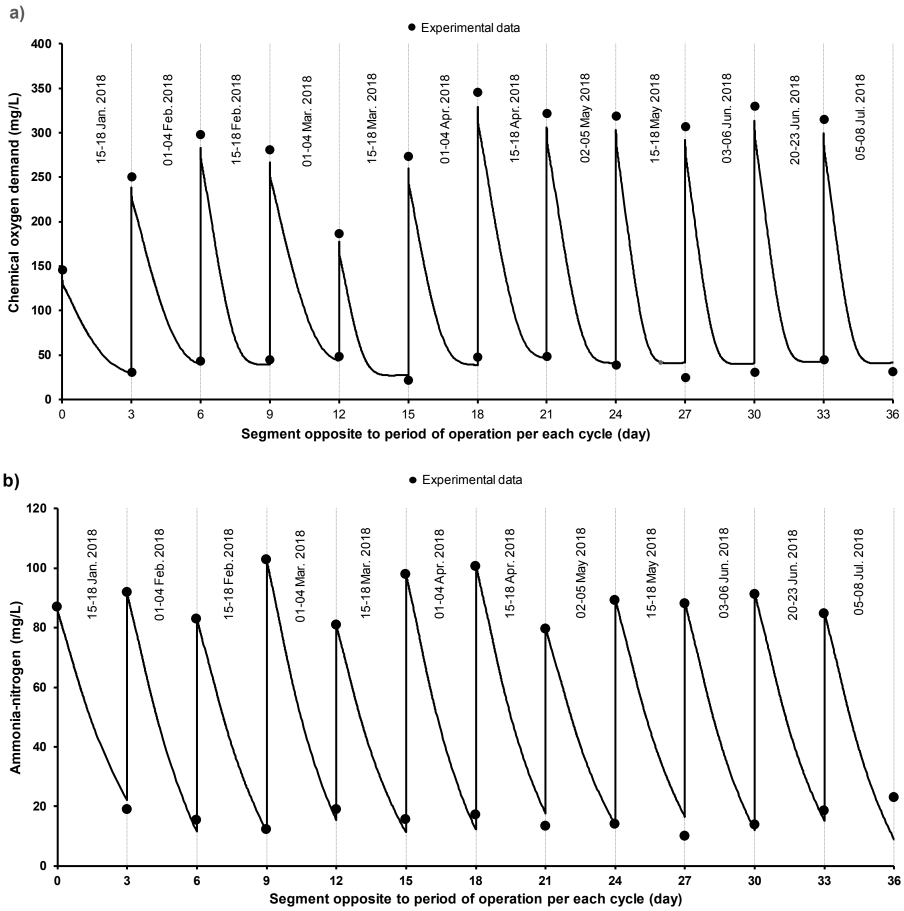


Fig. 4. Comparison of measured and predicted values for (a) chemical oxygen demand; and (b) ammonia-nitrogen during the entire experimental operation time.

Declarations of interest

None.