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Efficiency and effectiveness of systems for the treatment of domestic wastewater based on subsurface flow constructed wetlands in Jarabacoa, Dominican Republic

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Abstract

Constructed wetlands (CW) are well known nature-based systems for water treatment. This study evaluated the efficiency and effectiveness of seven domestic wastewater treatment systems based on horizontal flow CWs in Jarabacoa, the Dominican Republic. The results showed that the CWs were efficient in reducing the degree of contamination of wastewater to levels below the Dominican wastewater discharge standards for parameters such as the 5-day biochemical oxygen demand (BOD5) and chemical oxygen demand, but not for the removal of phosphorus and fecal coliforms. In addition, a horizontal flow subsurface wetland in the peri-urban area El Dorado was evaluated in terms of the performance of wastewater treatment in tropical climatic conditions. The concentrations of heavy metals, such as zinc, copper, chromium, and iron, were found to decrease in the effluent of the wetland, and the concentrations for nickel and manganese tended to increase. The levels of heavy metals in the effluent were lower than the limit values of the Dominican wastewater discharge standards. The construction cost of these facilities was around 200 USD per population equivalent, similar to the cost in other countries in the same region. This study suggested some solutions to the improved performance of CWs: selection of a microbial flora that guarantees the reduction of nitrates and nitrites to molecular nitrogen, use of endemic plants that bioaccumulate heavy metals, combination of constructed wetlands with filtration on activated carbon, and inclusion of water purification processes that allow to evaluate the reuse of treated water.

Keywords: Domestic wastewater; Constructed wetland; BOD; COD; Pathogens; Heavy metals

1. Introduction

The international community, through the United Nations, has proclaimed one of the Sustainable Development Goals: "Ensure the availability and sustainable management of water and sanitation for all" (United Nations, 2018). Sanitation in the Dominican Republic shows serious deficiencies. This is because public policy actions aiming at providing adequate sanitation conditions are traditionally relegated for multiple reasons (Urania Abreu, 2016). This situation is deteriorating the quality of rivers, groundwater, coasts, and natural resources, as well as the health of the population. It is estimated that only 28% of wastewater is captured through existing sewerage networks. Of the flow captured in sewerage networks, only 38% is treated, while the additional 62% remains as wastewater without treatment whose direct path is the receiving bodies of water. Until 2015, there were 79 biological

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process wastewater treatment plants (WWTPs) in the country, and only 50 were in operation (Urania Abreu, 2016). Discharges from more than one million septic tanks in the country (46% of the total) constitute the largest source of diffusive pollution. This figure does not include latrines that discharge directly into the bodies of surface and underground water. The most widespread treatment systems in the Dominican Republic are aerated lagoons, up-flow tanks with a sludge blanket, activated sludge, stabilization lagoons, biological trickling filters, and constructed wetlands (CWs) (Noyola et al., 2012). Most of these facilities are small, with flows less than 25 L/s, according to the classification established by the Pan American Center for Sanitary Engineering and Environmental Sciences (CEPIS). Therefore, it is urgent to search for and evaluate effective waste treatment systems that do not require large investments in order to mitigate the environmental problems posed by domestic waste generated in urban environments.

Conventional liquid waste treatment systems have high energy demand and high operation and maintenance costs (Temel et al., 2018). From an aesthetic point of view, they have an appearance similar to an industrial installation built with non-renewable materials such as concrete and steel and with equipment that consumes electrical energy. In addition, due to the presence of various mechanical attachments, they require specialized personnel to guarantee their operation and maintenance. These facilities are usually located outside of residential zones. Thus, the waste has to be collected and transferred to them over long distances (Stefanakis, 2020). An alternative to these systems is nature-based treatment systems (Nivala et al., 2019), such as CWs (Tao et al., 2017). In contrast to traditional systems, these systems practically consume no energy, and their operation and maintenance costs are very low (Pérez et al., 2022). They can be inserted in urban areas as they have an environmentally friendly appearance (Marín-Muñiz et al., 2020), which allows the treatment to be carried out in places close to the source of generation of the wastewater. This fact allows the reuse of certain treated water, provided that they meet the quality requirements (Vymazal et al., 2021).

On the other hand, in tropical regions where developing countries are concentrated, climatic conditions are favorable for the implementation of this type of treatment systems (Machado et al., 2017). In these areas, temperature rarely drops below 20°C during the day, while at night it drops slightly. The facts of no significant temperature variation between different seasons and moderate daily temperature changes favor the performance of these systems (Tanaka et al., 2011). Most biological processes directly depend on temperature. Thus, tropical climatic conditions favor their microbiological activity and the efficiency of the treatment (Zhang et al., 2014). CWs can be classified as surface flow wetlands and subsurface flow wetlands (Almuktar et al., 2018). Subsurface flow wetlands can be divided into horizontal flow wetlands and vertical flow wetlands. Horizontal flow wetlands are widely distributed throughout the world. Based on the accumulated experience over several decades in the construction and operation of wetlands, the most important design and operation characteristics for wetlands are: (1) the surface area is less than 2 500 m^2; (2) the length-to-width ratio is less than 3:1; (3) the depth is 0.4−1.6 m; (4) the hydraulic slope is 0.5%−1.0%; (5) the hydraulic head is less than 0.5 m/d; (6) the retention time is 2−5 d; (7) the wetland supports a natural medium or preferably an industrial by-product; (8) the porosity of the support is 30%−50%; (9) the particle size of the support is less than 20 mm, and for entrance and exit zones, it is 50−200 mm; and (10) the vegetation is preferably native species, with a 80% vegetation cover in the wetland (Almuktar et al., 2018; Kulshreshtha et al., 2022).

It is imperative to evaluate nature-based treatment systems and validate them as a credible alternative to conventional waste treatment methods. For some years in the Dominican Republic, several groups have been working on the design of CWs in different locations of the country (Jarabacoa, San José de las Matas, Constanza, Santiago de los Caballeros, and Santo Domingo Norte) (Vargas et al., 2021; Vásquez Guerra, 2019). However, the results obtained in the implementation of these systems have not been sufficiently evaluated. The objective of this study was to evaluate the efficiency and effectiveness of seven water treatment systems based on horizontal flow CWs, implemented in the municipality of Jarabacoa. In addition, the El Dorado wetland was selected to carry out a more detailed study of its performance in the treatment of wastewater in tropical climatic conditions (Pérez-Salazar et al., 2019), given that the bibliography on CWs in this climatic zone is still scarce. Furthermore, this wetland has been put into operation recently. Therefore, lessons learned from the construction of previous CWs were considered.

2. Materials and methods

The seven domestic WWTPs based on CWs in this study are located in the municipality of Jarabacoa in La Vega Province, the Dominican Republic. This municipality is located at 541 m above the sea level. Figure A.1 in Appendix A shows the location of the wetlands in the Jarabacoa municipality. According to the data from the meteorological station of Piedra Blanca in La Vega, the average temperature in this municipality is 22.8°C, with an average maximum temperature of 26.8°C and an average minimum temperature of 17.2°C. The studied domestic WWTPs consist of primary treatment with septic tanks (STs) and secondary treatment with horizontal subsurface flow CWs (Table 1 and Table A.1 in Appendix A).

Table A.2 in Appendix A shows the matrix of analytical determinations in this study and the years when analyses were carried out. During the sampling, samples was taken from the influent and effluent of the corresponding wetland. 2 L of each sample was collected in polyethylene plastic bottles and kept at 4°C in the absence of light until analysis. The evaluated parameters (the 5-day biological oxygen demand (BOD5), chemical oxygen demand (COD), phosphorus as orthophosphate, and fecal coliforms) were determined according to the

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Table 1
Design characteristics of CWs in Jarabacoa, Dominican Republic.

Treatment system	PE	Q_R (m^3/d)	A (square meters per PE)	Primary treatment	V_T of ST (m^3)	A_W (m^2)	Waterproofing material	Substrate and average substrate size	Macrophytes
Esc. Ambiental I	24	5.6	0.91	1 ST	4.48	21.85	Block/concrete	Gravel (7.5 cm); sand	Without plants
Esc. Ambiental II	50	10.0	0.85	1 ST	6.09	42.63	Block/concrete	Gravel (7.5 cm); sand	Without plants
El Arca	75	15.0	1.08	2 ST	10.82	61.25	Geotextile	Gravel (2.2 cm)	*Vetiveria zizanioides*
El Dorado	75	7.5	1.00	2 STs	10.84	70.00	Geomembrane (300 μm)	Gravel (2.2 cm)	*Vetiveria zizanioides*
Buenos Aires	150	30.0	0.67	2 STs	22.33	83.60	Geotextile	Gravel (2.2 cm)	*Vetiveriazizanioides*
Cristo Rey	220	38.5	1.43	2 STs	61.60	266.00	Geomembrane (1 mm)	Gravel (2.3 cm)	Without plants
La Trinchera	375	75.0	0.22	1 ST	6.75	82.08	Block/concrete	Sand gravel (2.2 cm); gravel (7.5 cm)	*Vetiveria zizanioides*

Note: PE is the population equivalent, Q_R is the residual flow, A is the treatment system area per PE, V_T is the total volume, and A_W is the wetland surface area.

standard methods for the analysis of water and wastewater (Baird et al., 2017).

The removal efficiency of pollutants (E) in wetlands was calculated with the following equation:

$$E = \frac{X_{\text{inlet}} - X_{\text{outlet}}}{X_{\text{inlet}}} \times 100\% \tag{1}$$

where X_{inlet} and X_{outlet} are the parameter values averaged over time at the inlet and outlet of the wetland, respectively. It should be noted that it does not make physical sense to calculate the removal efficiency from the measurements at the inlet and outlet at the same moment because it does not represent the real efficiency of the treatment in the wetland. The portion of residual water that enters the wetland takes a residence time to leave it under conditions of constant flow conditions. This situation should be considered. Only the samples collected at two moments at the inlet and outlet can be used to evaluate the point-to-point contaminant removal efficiency.

In addition, a more in-depth study was carried out in the El Dorado wetland for 7 d. In situ measurements were conducted, and seven composite samples were taken at the inlet and outlet of the wetland in March and August 2020. The samples were analyzed at the Instituto Tecnológico de Santo Domingo (INTEC) Environmental Analysis Laboratory. In situ measurements of temperature, salinity, electrical conductivity, total dissolved solids (TDS), and resistivity were conducted with a YSI Professional Plus multiparameter probe. Dissolved oxygen and oxygen saturation were measured with a YSI Professional 2030 probe. pH and turbidity were measured with a Thermo Scientific Orion A329 portable meter and an Orion AQ4500 portable turbidimeter, respectively.

1 L of samples was collected at intervals of 1 h for four moments each day, thus forming a 4-L sample. The samples were kept in polyethylene plastic bottles at 4°C in the absence of light until analysis. Other parameters required by Dominican wastewater discharge standards (TDS, total suspended solids (TSS), total solids (TS), BOD5, COD, and fecal coliforms) were measured with these samples (Moya-Pons et al., 2003) according to standard methods for water and wastewater analysis (Baird et al., 2017). In situ measurements were carried out at the same time of a day during sampling. A detail description of statistical analysis is detailed in Appendix A.

3. Results and discussion

3.1. Characterization of studied WWTPs based on CWs

The CWs in this study serve small communities in Jarabacoa, the Dominican Republic. Table 1 lists the design parameters of these wetlands. The table shows that the design parameter values of these wetlands are quite different, with significant variability. According to the classification proposed by the CEPIS and used by Noyola et al. (2012) for characterization of the WWTPs in various Latin American countries, these WWTPs are considered small treatment plants, with flows less than 25 L/s (2 160 m^3/d). This size seems to be characteristic of other Latin American countries, such as Costa Rica, where about 99% of WWTPs are small and even 90% of WWTPs have flow rates less than 5 L/s (432 m^3/d) (Centeno Mora and Murillo Marín, 2019).

One disadvantage of treatment systems based on CWs is that the area required for the installation of the treatment system per population equivalent (PE) is greater than that required for conventional treatment systems. In conventional treatment systems for small communities, the rate of specific surface area per PE, which represents the area allocated to the treatment system for the waste generated by each PE of a locality, can vary between 0.13 m^2 per PE to 0.15 m^2 per PE (Freeman et al., 2019). However, for small communities in European countries, this parameter has been reported with average values of (5 ± 3) m^2 per PE in Spain (Puigagut et al., 2007) and (5 ± 2) m^2 per PE in Portugal (Duarte et al., 2010). In this study, this parameter was (0.9 ± 0.4) m^2 per PE, five times lower than that reported for European countries, although still greater than that for conventional systems. The most favorable environmental conditions for these processes existing in the Caribbean allow to significantly reduce the rate of specific surface area per PE.

Small community treatment systems typically have high costs per PE due to the scale factor, collection characteristics (variability of flow and loads), and the complexity of control, maintenance, and monitoring. Freeman et al. (2019) estimated that the costs of construction and assembly of a conventional installation for the treatment of waste generated by 1 000 PE in England were 403 USD per PE. Puigagut et al. (2007)

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reported that the investment costs of facilities based on CWs in Spain were (300 ± 100) USD per and Temel et al. (2018) reported 301 USD per PE in Turkey. In the Latin American region, Centeno Mora and Murillo Marín (2020) estimated that the costs for this type of installation in Colombia were 117−225 USD per PE. In the Dominican Republic, the average investment costs obtained in this study were (200 ± 100) USD per PE, which agreed with those obtained by Centeno Mora and Murillo Marín (2020), and represented 50% of the investment costs of a conventional installation for small communities.

3.2. Efficiency and effectiveness of wastewater treatment

3.2.1. Organic matter removal

Puigagut et al. (2007) surveyed horizontal flow CWs in Spain and noted that when BOD5 organic load rates applied to wetlands were above 5 g/($m^2 \cdot d$), low-quality effluents were generally produced (with BOD concentrations above 25 mg/L, the green line in Fig. 1(a)). As shown in Fig. A.2 in Appendix A, the average organic loads in the studied wetlands were 19−54 g/($m^2 \cdot d$). Although all the applied loads were four to 11 times greater than the aforementioned value (5 g/($m^2 \cdot d$)), most of the BOD5 concentrations of the effluents were below the value of the wastewater discharge standards (50 mg/L) and close to the aforementioned value (25 mg/L). An exception was the Cristo Rey wetland, where the measured BOD organic load at the inlet ranged from 27 g/($m^2 \cdot d$) to 40 g/($m^2 \cdot d$). However, the measured BOD5 concentration at the outlet varied between 17 mg/L and 68 mg/L. In this case, only two samples were studied, resulting in a significantly amplified standard deviation. Li et al. (2018b) carried out in a study in 61 horizontal flow CWs in China and reported an average BOD organic load of (34 ± 8) g/($m^2 \cdot d$) at the inlets, which agreed with this study ((32 ± 12) g/($m^2 \cdot d$)).

Biodegradation in CWs is mainly due to the action of the communities of microorganisms present on the filter support or in the rhizosphere of macrophytes (Brisson and Chazarenc, 2009; Tanner, 2001). Fig. 1(b) shows that the BOD5 removal efficiency ranged from 67% to 95%, with an average value of 78%. These values were 9% less than those reported by Vymazal (2019), who studied 102 horizontal flow CWs with an average BOD5 organic load of 6.05 g/($m^2 \cdot d$). However, the BOD removal efficiencies were in the range of the values reported in the literature for wetlands with efficient operation (66%−95%) (Grinberga, 2020; Rousso et al., 2019; Vidanage et al., 2020). The Kruskal−Wallis H test and the Games−Howell post-hocs test did not show statistically significant differences in the mean BOD5 removal efficiencies of the studied wetlands.

On the other hand, the efficiency of wastewater treatment in terms of COD varied between 73% and 95% (Fig. 2(b)), with an average value of 82%. This value was similar to that of BOD5. As shown in Fig. 2(b), the effluents of all wetlands met the discharge standards for COD. These effluent COD concentrations were also in the range of those reported for both horizontal flow subsurface wetlands and hybrid wetlands (71%−96%) (Mello et al., 2019). It should be noted that

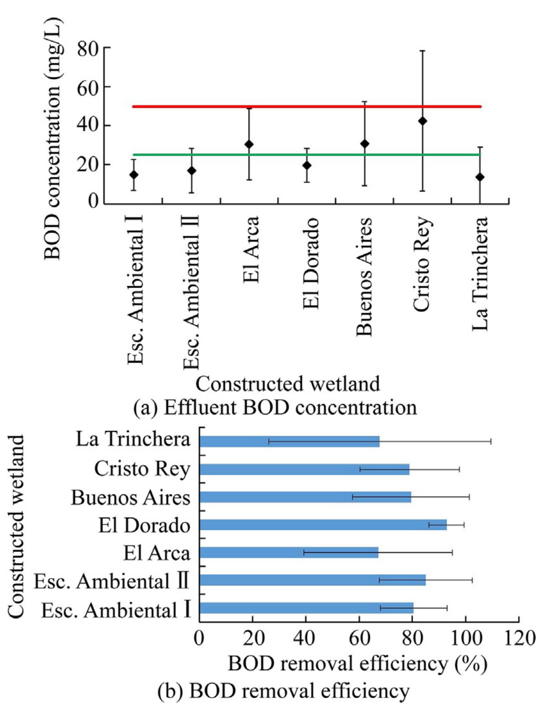

Fig. 1. Effluent BOD5 concentrations and BOD removal efficiencies in evaluated CWs (with the red line representing the discharge standard value (Moya-Pons et al., 2003) and the green line denoting the limit of good-quality effluents according to Puigagut et al. (2007)).

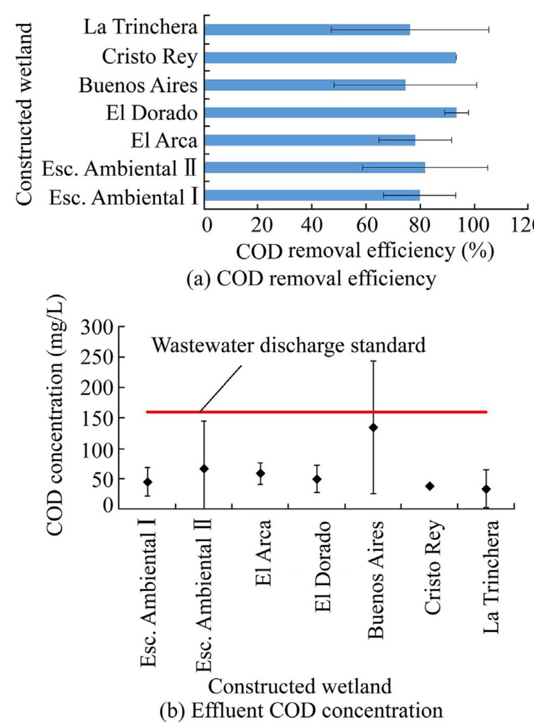

Fig. 2. COD removal efficiencies and effluent COD concentrations in evaluated CWs.

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although the CWs were designed for a maximum COD load of 740 mg/L, the systems were able to respond efficiently and effectively to higher loads up to 1 408 mg/L. Therefore, the systems responded efficiently to the removal of organic matter. The Kruskal−Wallis H test and the Games−Howell post-hoc test showed statistically significant differences between the El Arca and El Dorado wetlands in terms of the COD removal efficiency. There was no significant difference between the rest of the wetlands.

3.2.2. Phosphate removal

Phosphorus is important to plant growth. Thus, its level in water must be controlled to avoid eutrophication (Brisson and Chazarenc, 2009). The elimination of phosphate in CWs is accomplished by the absorption with plants and transformation by microorganisms (Khalifa et al., 2020). The removal of phosphate in CWs varies between 30% and 70%, depending on the characteristics of the wastewater and the used media (Baldovi et al., 2021; Ruan et al., 2021), and can reach very high efficiencies of over 90% in hybrid wetlands (Saeed and Khan, 2019). In the studied wetlands, the removal efficiency varied between 15% and 49% (Fig. 3(a)). Fig. 3(b) shows that the effluents of all the wetlands had phosphate concentrations above the values of the wastewater discharge standards, indicating the inefficiency of these systems for phosphate removal. In a study of 149 wetlands, Vymazal (2007) found that total phosphorus removal in horizontal flow wetlands reached an average efficiency of 41%, slightly greater than the average value in this study (35%). It is known that CWs do not have efficient mechanisms for phosphorus removal (Song et al., 2015), except for some macrophytes that use small amounts of nutrient for their biological functions (Burgos et al., 2017). A solution to increase the elimination of phosphorus could choose adsorbent substrates, such as zeolite, dolomite, limestone, and apatite (Arteaga-Cortez et al., 2019) or the combination of wetlands with biochar filters (Li et al., 2018a).

3.2.3. Fecal bacteria removal

In CWs, pathogen removal is achieved through a combination of physical, chemical, and biological processes. Some techniques are the filtration and adsorption of microorganisms in the roots of plants and the filter support, oxidation by dissolved oxygen, and exposure to biocides excreted by some plants (Khalifa et al., 2020). The available data indicated removal efficiencies of fecal coliforms of approximately 7%−25% in terms of the maximum possible number (MPN) of fecal coliforms per 100 mL, with an average value of 17% (Fig. 4). These values were in the range reported by Khalifa et al. (2020) and Ramprasad et al. (2017).

As shown in Fig. 4(b), the concentrations of fecal coliforms in the influents and effluents of the evaluated wetlands exceeded the national discharge standards for the dumping of effluents into rivers. Fecal and total coliforms are eliminated by chlorination and ultraviolet radiation, systems that require high costs of installation, maintenance, and operation (Nguyen et al., 2019), although another alternative is filtration through infiltration systems.

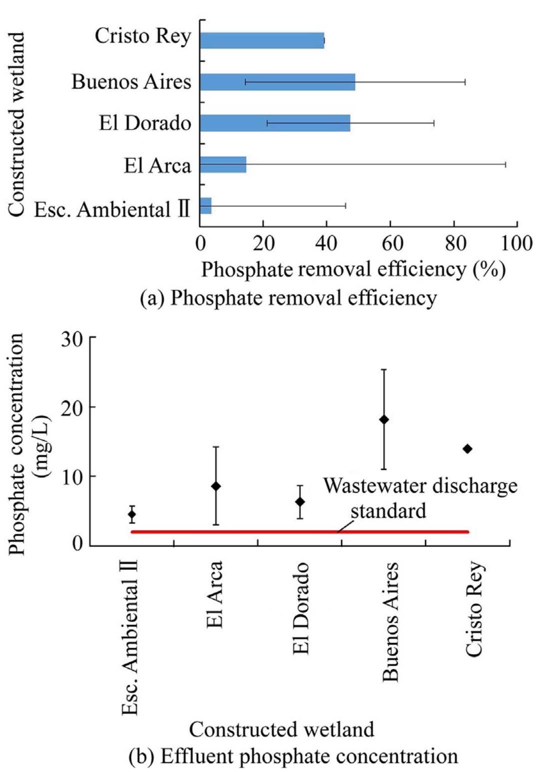

Fig. 3. Phosphate removal efficiencies and effluent phosphate concentrations in evaluated CWs.

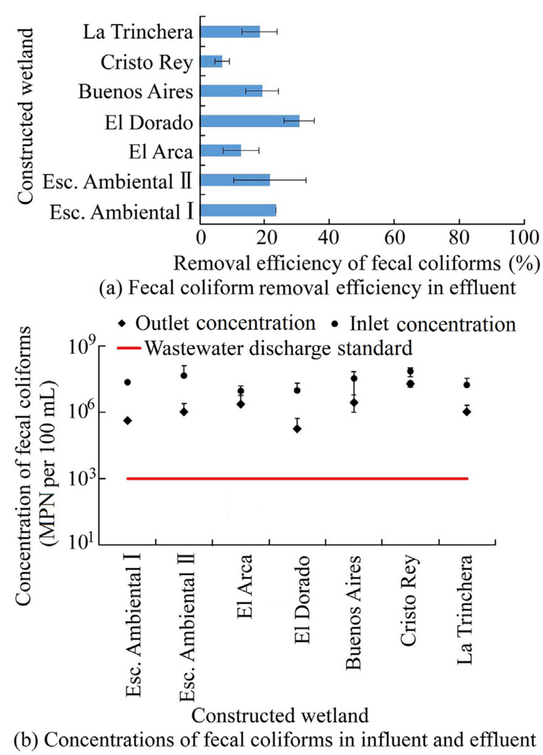

Fig. 4. Fecal coliform removal efficiency in effluents and concentrations of fecal coliforms in influents and effluents in evaluated wetlands.

3.3. Statistical analysis

This study showed a high multicollinearity between the predictor variables (Table A.3 in Appendix A). The collinearity between the hydraulic loading rate and the absence of plants with the rest of the variables was significant. Table A.4 in Appendix A shows the correlation coefficients between wetland design and operation parameters and the removal efficiency of organic pollutants from domestic wastewater. The parameters that showed statistically significant correlations with the removal efficiency were the ratio of wetland area to PE, which showed a positive correlation, and the hydraulic loading rate, with a negative correlation.

3.4. Performance of El Dorado wetland

As previously stated, the El Dorado treatment system was selected to carry out a more in-depth evaluation of the quality of the wastewater treatment that is achieved through CWs.

3.4.1. Flow variability in El Dorado wetland

In sampling days, the flow varied between 0.4 m^3/d and 8.5 m^3/d. The measured maximum flow was slightly higher than the design value (7.5 m^3/d). However, the measured average flow was 2.5 m^3/d, causing a hydraulic residence time of 6.6 d.

3.4.2. Physicochemical characteristics of wetland influent and effluent

Temperature and pH significantly affect the development of microorganisms that contribute to the degradation of pollutants in CWs. In fact, the difference in the performance of CWs in different climatic zones due to the influence of temperature has been found (Zhang et al., 2015). Temperature measurement was conducted in March to August in the Dominican Republic where it is the tropical climatic zone. The measured temperature was above 20°C at the inlet and outlet of the wetland. Fig. 5(a) shows that the wastewater was subject to a slight cooling process as it passed through the wetland. In the wetland, the wastewater to be treated has pH values of 6.1−8.8, with a median of 6.5 (Fig. 5(b)). As it passed through the CW, pH was reduced to a range of 6.5−7.2, with an average value of 6.9 in the effluent. This phenomenon was positive to wastewater treatment because neutral or slightly acidic pH values favor the removal of sulfates (Marín-Muñiz et al., 2020). The measured temperature and pH were within the limits of the Dominican regulations for wastewater discharge to surface water bodies (Moya-Pons et al., 2003).

Electrical conductivity is an important factor to establish possible reuse of treated wastewater. For example, to safely use treated water for agricultural purposes, electrical conductivity values must be below 700 μS/cm (Khan et al., 2019; Villamar et al., 2018). In this study, all electrical conductivity values in the effluent were below this limit value, indicating that the treated wastewater could be reused for certain purposes if other requirements are met (Fig. A.3 in Appendix A). The electrical conductivity of the influent was consistently higher than that of the effluent, while the resistivity was generally higher in the effluent than in the influent (Fig. A.3 in Appendix A). This behavior was expected and indicated the decrease in ions resulting from the wastewater treatment. It should be noted that the slope of the resistivity curve was inversely proportional to that of the electrical conductivity curves at all times (Fig. A.3 in Appendix A). However, an anomalous resistivity value was observed on the fifth day of measurements, and the resistivity was greater at the inlet than at the outlet of the wetland, which was not expected. In this sense, it should be noted that there is a direct relationship between electrical conductivity in wastewater and salinity, the concentration of total soluble solids (García-Ávila et al., 2019) and inverse with respect to resistivity. The conductivity values in all cases were within the limits of the Dominican regulations for wastewater discharge to surface water bodies (Moya-Pons et al., 2003).

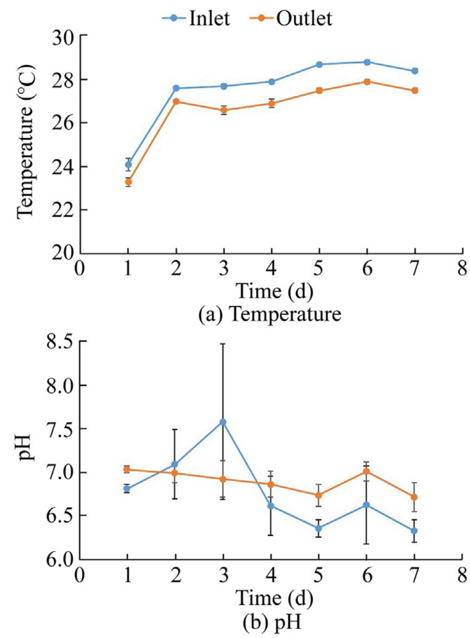

Fig. 5. Variations of temperature and pH in influent and effluent of El Dorado wetland.

In domestic wastewater, there are products such as detergents that contain ions like sodium, boron, chlorides, and fluorides, which contribute to the water salinity (Villamar et al., 2018). As the salinity of water increases, its ability to dissolve oxygen decreases. Therefore, salinity reduction is an important part of wastewater treatment. Fig. A.3 in Appendix A shows that the wetland was efficient in reducing this pollution indicator with reduction rates of 20%−60%. On the other hand, the ions that produce the salinity effect are part of TDS. Reuse of treated wastewater for irrigation purposes requires that the TDS content of the treated wastewater should be less than 450 mg/L (Norton-Brandão et al., 2013). As shown in Fig. A.3 in Appendix A, all the TDS values in the wetland effluent were below this limit value. This implied that the treated wastewater could be used for irrigation purposes if other water quality requirements are met. The average TDS removal efficiency was 39%.

Before the wastewater entered the El Dorado treatment system, it had underwent primary treatment in STs. The influent maintained TSS contents of 84−175 mg/L (Fig. 6(a)). The TSS removal efficiency of the CW was greater than 88% (Fig. 6(a)), causing TSS contents in the effluent to be lower than the norms (less than 50 mg/L) (Moya-Pons et al., 2003).

Fig. 6(b) shows the results of turbidity measurements. On the third day, turbidity showed an increase relative to that on the previous day, but the turbidity increase was not as significant as the measured TSS content. However, on the third and fourth days, the measured turbidity showed large standard deviations. Therefore, the trends of turbidity and TSS content were different. A similar situation occurred on the seventh day, in which the measured turbidity significantly dispersed. The average removal efficiencies of TSS measured in different ways were similar (93% ± 4%).

Fig. 7(a) shows the TS contents in the influent, which showed a significant variability (348−1 332 mg/L). TS represents the sum of TDS and TSS. In all cases, a reduction in the TS content in the wastewater was achieved, with an average efficiency of 47%.

The oxygen concentration in water is another important parameter to guarantee water quality. Fig. 7(b) shows that an increase of 30% in the dissolved oxygen content in the treated water was achieved in all cases. The oxygen saturation percentage at the exit of the wetland was 15%−45%, lower than the limit value of the Dominican standards for wastewater discharge to surface water bodies (70%) (Moya-Pons et al., 2003).

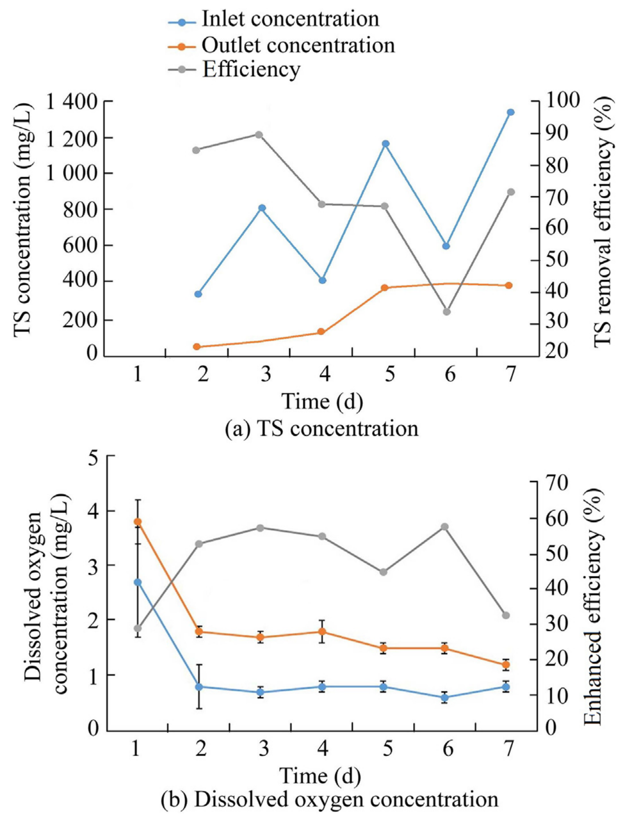

Fig. 7. Variation of TS and dissolved oxygen concentrations in influent and effluent of El Dorado wetland.

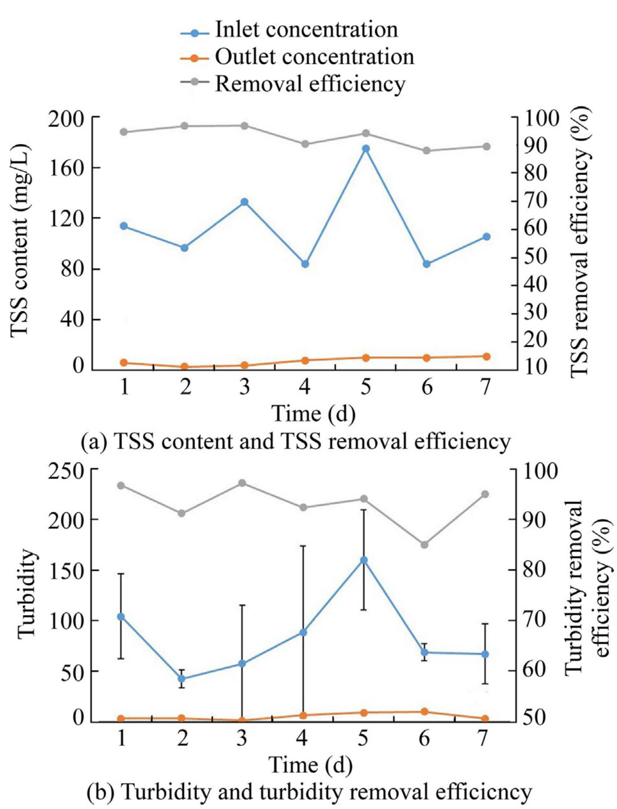

Fig. 6. Variation of TSS and turbidity in influent and effluent of El Dorado wetland.

3.4.3. Nitrogen removal efficiency

Fig. 8 shows the total nitrogen concentrations in the El Dorado wetland. The treatment system was efficient and effective in the elimination of nitrogen, with removal efficiencies of 28%−55%. In all cases, the total nitrogen concentrations were lower than the standards (Moya-Pons et al., 2003). The average total nitrogen removal efficiency was 37%, slightly lower than that reported by Vymazal (2007) as

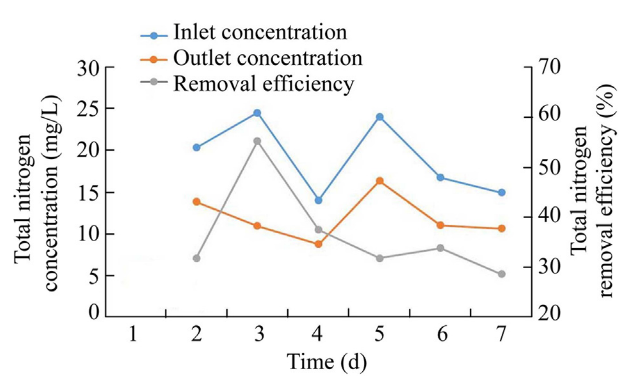

Fig. 8. Variation of total nitrogen concentration in El Dorado wetland.

an average for 137 wetlands (42%). In general, the removal of nitrogen in CWs is a challenge and a research topic, which will be the focus of further work (Ruan et al., 2021). Nevertheless, the low nitrogen removal efficiency could be related to the anaerobic conditions in the wetland, which negatively affected the rate of conversion of organic nitrogen to ammoniacal nitrogen. The pH values lower than 8 decreased the volatilization of ammoniacal nitrogen. In addition, the low dissolved oxygen concentration discouraged the oxidation of ammoniacal nitrogen to nitrate, a necessary material for the denitrification process in which nitrogen is eliminated in gaseous form (Torres Bojorges et al., 2017). One way to overcome this situation is the use of hybrid CWs.

3.4.4. Determination of heavy metal contents in influent and effluent of El Dorado wetland

Heavy metals are toxic to ecosystems, mainly cadmium (Cd), chromium (Cr), nickel (Ni), mercury (Hg), lead (Pb), manganese (Mn), copper (Cu), and zinc (Zn) (Hussain and Qureshi, 2020). These heavy metals pass into domestic wastewater from many sources, such as the wear of pipes, galvanized caps and roofs, copper covers, car washes, detergents, medicines, amalgams, and electronic waste (Singh and Srivastava, 2016). The effects of vegetation, substrate, shape, hydraulic loads, and flood regimes on the performance of CWs for removal of metals and metalloids have been known (Sharma et al., 2021). The characteristic removal processes of these pollutants are physical—chemical processes (sedimentation, flocculation, ion exchange, oxidation—reduction reactions, adsorption, and precipitation) and biological processes (microbial activity and vegetation consumption). Although metals accumulate in the sediments, plants, or water of CWs, accumulation primarily occurs in sediments (Pedescoll et al., 2015). This is fundamentally conditioned by the redox state of the wetland bed, which ultimately determines the solubility of the metals, and the pH of the medium also has an effect. Design features determine the environmental conditions within the wetland, and specifically, its redox status. Under the flood conditions of horizontal flow wetlands, the shortage of oxygen is supplied by microorganisms from nitrates and Fe and Mn oxides. In addition, a decrease in redox potential below 200 mV makes it possible for anion sulfates to turn into sulfides (Ross, 1989). Under these reducing conditions, the sulfides of various metals precipitate (Vymazal and Březinová, 2016). It should also be noted that in wetlands, depending on the redox state, removal can be positive or negative (Arroyo et al., 2010).

In this study, the determination of heavy metal contents was carried out for the first time in a domestic wastewater treatment system based on CWs (in the El Dorado wetland) in the Dominican Republic. The studied metals were Cd, Zn, Cu, Cr, Ni, Pb, Co, Fe, and Mn. The vetiver plant species used in this wetland have been known to have a high affinity for metals (Batool and Saleh, 2019). Cd, Pb, and Ni were not detectable in the influent samples, which is common in domestic wastewater (Vymazal and Březinová, 2016)), and Co was also undetectable (Fig. 9). However, a small amount of Ni was

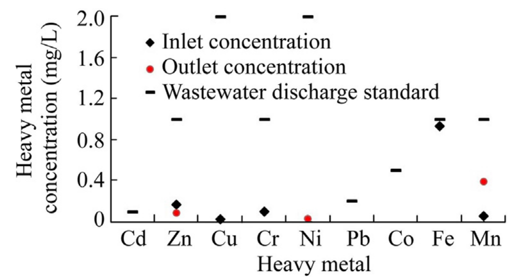

Fig. 9. Concentrations of heavy metals in wastewater.

detected in the effluent samples, indicating negative removal for this metal (Arroyo et al., 2010). A similar behavior was observed in Mn. The concentration of Mn increased significantly as it passed through the wetland, which agreed with the findings of Arroyo et al. (2010) and Jana and Kro (2009). The increase in the concentration of Mn in the water might be attributed to microbial reduction of oxides (Du Laing et al., 2007). The behavior of Ni was probably associated with that of Mn oxides because Ni was adsorbed on Mn oxides and then released during the solubilization of Ni (Du Laing et al., 2007; Vymazal and Březinová, 2016). However, Mn oxides were reduced at higher reduction potentials than Fe oxides (Du Laing et al., 2007). This could explain why Mn was solubilized and Fe was removed in the wetland. Regarding Cu and Cr, the wetland removed these elements to levels below the detectable limits. In contrast, the concentration of Zn slightly decreased. Only one influent sample showed that the Fe concentration exceeded the limit value of the Dominican standards for wastewater discharge to surface water bodies. In all other samples, including the effluent, the concentrations of heavy metals were lower than the limit of the standards (Moya-Pons et al., 2003).

3.5. Discussion

This study was carried out in a tropical area, where investigations to CWs have rarely been reported. Therefore, it is important to make some comparisons between the treatment of domestic wastewater in CWs in tropical areas and other geographical areas. The main differences can be explained by the difference in climate and environmental conditions. The first factor is related to the biodiversity of plants that are used, which is generally higher in tropical areas. This can provide a greater variety of microorganisms and increase the treatment capacity of CWs. On the other hand, in tropical areas, temperature tends to be high with intense solar radiation. This can speed up biological and chemical processes in CWs, potentially increasing the rate of pollutant removal. However, the tropics experience more pronounced rainy and dry seasons than other regions. Precipitation variability can affect the efficiency and functioning of CWs because water flow rates can vary significantly. For this reason, during the design of a CW, the diversion of runoff water must be considered to avoid the washing of the wetland during the periods of severe rains and floods. Another risk in tropical areas may be the presence of invasive plant and animal species, which could negatively affect the functioning of CWs. The control and management of these species may require additional measures compared to other regions. Finally, soils in tropical areas tend to be rich in nutrients and organic matter. This can influence the processes of absorption and retention of nutrients in CWs, as well as the production of plant biomass. Therefore, in consideration of the advantages and difficulties of CWs in tropical areas, the operation of CWs may be more efficient in tropical areas than in other geographical areas. It is therefore important to consider the variations of climate and environmental conditions when designing and managing CWs in different locations on the planet.

4. Conclusions

The wastewater treatment systems based on CWs implemented in the municipality of Jarabacoa in La Vega Province, the Dominican Republic demonstrated their efficiency and effectiveness in reducing the degree of contamination of wastewater to levels below the wastewater discharge standards for parameters such as BOD5 and COD, but not for the removal of phosphorus and fecal coliforms. These results showed that it is necessary to include tertiary treatment facilities to achieve the Dominican standards for treated wastewaters.

In the studied system based on a horizontal flow subsurface wetland (El Dorado) in a peri-urban area, the concentrations of heavy metals were determined for the first time in a CW in the Dominican Republic. Of the nine heavy metals, the presence of Cd, Ob, and Co was not detected, a decrease in Zn, Cu, Cr, and Fe concentrations in the effluent was achieved, and an increase in Ni and Mn concentrations was detected in the effluent. However, it should be noted that, in all cases, the levels of heavy metals in the effluent were lower than the limit values of the treated wastewater discharge standards.

According to this study, we can affirm that CWs constitute a viable and sustainable option for domestic wastewater treatment in urban communities in the tropics. To improve the functioning of CWs, a microbial flora that guarantees the reduction of nitrates and nitrites to molecular nitrogen should be selected to enhance the efficiency of nitrogen removal. Endemic plants that bioaccumulate heavy metals can be used. CWs can be combined with filtration under activated carbon for the removal of heavy metals and phosphate compounds. In addition, water purification processes can be included to evaluate the reuse of treated water.

Declaration of competing interest

The authors declare no conflicts of interest.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.wse.2023.08.004.

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