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Performance Evaluation of a Pilot-Scale Constructed Wetland with *Typha latifolia* for Remediation of Domestic Wastewater in Zimbabwe

Richwell Alufasi [1], Wilson Parawira [2], Cuthbert J. Zvidzai [1], Alexandros I. Stefanakis [3], Nancy Musili [4], Phiyani Lebea [5], Ereck Chakauya [6,7] and Walter Chingwaru [8,*]

[1] Biological Sciences Department, Bindura University of Science Education, Private Bag, Bindura 1020, Zimbabwe; alufasirichie@gmail.com (R.A.); czvidzai@buse.ac.zw (C.J.Z.)
[2] Zimbabwe Council of Higher Education, 21 J, M. Nkomo Road, Hatfield, Harare P.O. Box H100, Zimbabwe; parawiradr@yahoo.co.uk
[3] School of Environmental Engineering, Technical University of Crete, TUC Campus, 73100 Chania, Greece; astefanakis@tuc.gr
[4] Medlico Research and Training Centre, 4 Lanark Road, Belgravia, Harare, Zimbabwe; nancymusili@aims.com
[5] Tokabio, No. 4 Edenvale Street, Wieda Park, Centurion, South Africa; phiyani@tokabio.com
[6] AUDA-NEPAD Southern Africa Network for Biosciences, Council for Scientific and Industrial Research (CSIR) Advanced Agriculture and Food, Pretoria, South Africa
[7] Food Security and Safety Focus Area, Faculty of Natural and Agricultural Sciences, North-West University, Mmabatho 2735, South Africa
[8] National Biotechnology Authority of Zimbabwe, No. 21 Princess Drive, Newlands, Harare, Zimbabwe
* Correspondence: wchingwaru@yahoo.co.uk

Abstract: The management of wastewater remains a challenge, particularly in developing countries. The potential use of constructed wetlands to treat wastewater is promising but their contaminant removal efficiencies, particularly in a tropical country such as Zimbabwe, are not fully understood. A pilot-scale study was undertaken in Zimbabwe to evaluate the efficiency of vertical-flow constructed wetlands planted with *Typha latifolia* in the treatment of domestic wastewater. Four pilot subsurface vertical-flow constructed wetland units (measuring 1 m × 1 m × 1.1 m) were built from concrete. The units were filled with waste rock from a nickel mine. Three units were planted with *Typha latifolia* while the fourth one was left unplanted, acting as the control. Each unit was loaded with wastewater at a rate of 220 dm^3/day. Physico-chemical and bacteriological parameters were analyzed during the winter season. Physico-chemical and bacterial contaminant concentrations were significantly lower in the effluent than in the influent, and the system achieved maximum removals for BOD_5, COD, TDS, TSS, nitrates, phosphates, phosphate pentoxide, phosphorus, and *E. coli* of 56.01%, 82.87%, 30.61%, 90.40% 17.26%, 35.80%, 36.19%, 40.64%, and 90.28%, respectively. The study shows that constructed wetland systems can be successfully established for the removal of physical, chemical, and microbial contaminants from domestic wastewater.

Keywords: constructed wetlands; removal efficiency; *Typha latifolia*; domestic wastewater

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1. Introduction

The depletion and scarcity of freshwater resources have become a global crisis as available resources can no longer meet the increasing demands of water usage [1–6]. Studies have shown that water scarcity results from an interplay of factors, including increased per-capita water use, over-abstraction of groundwater, climate change, and the extended contamination of water sources [6]. Makopondo et al. [7] report that in the year 2020, 1.1 billion people had no access to clean drinking water and 2.6 billion had no adequate sanitation. It is projected that 16% of sub-Saharan Africa's population and 1.8 billion people across the globe will have no access to fresh and drinking water by 2025 [3].

A more severe and disturbing crisis is that of poor sanitation, which is mainly affecting developing countries. Although sub-Saharan Africa and southern Asia account for the vast majority of the world's population, their sanitation coverage remains below 50% [4]. The lack of access to better sanitation facilities and ingestion of contaminated water have caused illnesses in people, and millions of people, including 3900 children, die from diarrheal diseases every year [8].

In order to address water shortages, many countries have resorted to water recovery methods, such as wastewater reuse or reclamation, as a valuable source of water, especially for non-potable requirements, e.g., agriculture and urban irrigation [5,9–12]. However, management of wastewater remains a challenge, particularly in developing countries, as evidenced by higher levels of contaminants in water systems.

Over the years, wastewater treatment approaches (physical, chemical, and bioremediation techniques) have been developed and implemented across the globe. In addition, several studies have investigated the efficiency of these technologies for contaminant removal from various forms of wastewater [9,12,13]. However, the technologies, which were mainly developed in Europe, are machine- and labor-intensive and expensive to install and operate. Most developing countries adopted these technologies with little or no modifications to suit their climatic conditions. As such, the installed technologies are, in most cases, dysfunctional and fail to cope with the unprecedented large volumes of wastewater discharge generated in most ever-growing cities [14,15]. This has resulted in the release of untreated or partially treated wastewater into surface water bodies, adversely affecting the quality of the water [2,16]. The presence of contaminants in water sources poses serious health hazards, necessitating the implementation of mitigation measures to reduce their occurrence and associated public and environmental health risks. There is therefore a need for natural, robust, and both economically and environmentally sustainable wastewater treatment technologies to recover water, such as constructed wetlands (CWs).

In recent years, research evidence has widely recognized the application of CWs as a solution to both treating and preserving water resources [5,17]. Constructed wetlands are engineered wastewater treatment systems that are designed to mimic processes that occur in natural wetlands [1,18,19]. The technology utilizes wetland macrophytes, substrate materials, and their microbial communities to remove contaminants from wastewater through the interaction of physical (e.g., sedimentation, filtration, and volatilization), chemical (e.g., precipitation), and biological processes, as well as operational conditions [1,16,19–21]. These interactions are capable of converting various pollutants into non-toxic by-products in the wastewater [18]. Compared to conventional wastewater treatment systems, CWs are less expensive to install and easy to operate and maintain [9,10,12]. Under different sanitation typologies, they have proved to be an efficient wastewater treatment method, even in rural communities [12,22].

This technology has been implemented in many parts of the world since the 1960s [23–25] and has shown great potential in treating various types of wastewater (e.g., municipal, domestic, agricultural, and industrial wastewater) [10,17,26].

The various components of the CWs play significant and complementary roles in contaminant removal. The substrate material used in CWs plays a crucial role in the treatment process by providing a surface area for microbial growth and facilitating the biodegradation of organic matter. The substrate media also carry out physical processes such as filtration, adsorption, and absorption [16]. A number of materials have been tested, including soil, crushed rock, gravel, activated carbon, coal slag, crushed plastic, steel slag, etc. [9,22,27–29]. However, the nature (type, shape, and pore spaces) and chemical composition of the substrate affect the characteristics of the effluent [16].

In addition to the substrate, the wetland macrophytes increase the capacity of the CW to remove physico-chemical as well as microbial contaminants [30]. Common macrophytes used are *Phragmites*, *Typha*, *Scirpus*, *Phalaris arundinacea*, and *Iris*, and the process efficiency of CWs depends on the plant species [5]. The mechanisms employed by plants to remove contaminants from wastewater include biological, chemical, and physical processes. Mechanisms such as the absorption and accumulation [31], sedimentation, and filtration of suspended solids [32] have been validated. In addition, macrophytes provide oxygen to the rhizosphere, reduce wastewater flow, and provide time for solid particles to settle [16]. Furthermore, Vymazal [33] reported that macrophytes actively participate in carbon, nitrogen, and phosphorus cycles, as well as the production of allelochemicals or antibacterial substances. However, removal efficiency associated with different plant species is difficult to demonstrate due to inherent variations between studies and monitoring practices [34]. Nevertheless, macrophyte roots provide a large surface area for the attachment of microorganisms, which are responsible for the biological degradation of contaminants.

The microbes in CWs are responsible for the transformations of various nutrients and biosorption [21,35]. The microbes in CWs are found in the substrate matrix and are categorized according to the zones we find them, i.e., aerobic or anaerobic zones. The aerobic zones, which are reportedly rich in microorganism diversity, assist in metal oxidation while the anaerobic zone is rich in sulfate-reducing bacteria [5]. However, design features, such as the wastewater loading regime, affect the contaminant removal processes occurring in a CW [30].

There are two CW designs that have been developed for wastewater treatment, namely surface flow (SF) and subsurface flow (SSF). Surface flow CWs (SF CWs) treat wastewater by allowing it to flow over the surface of the wetland substrate [5,22] but is not preferred because it create an environmental nuisance [16]. However, studies have reported high microbial removal rates due to the disinfection of wastewater by sunlight [36]. In SSF CWs, the wastewater flows beneath the surface of the wetland. The primary removal mechanism in SSF CWs is filtration and adsorption by the wetland substrate and plant roots [31,37]. There are two types of SSF CWs, horizontal-flow (HF CWs) and vertical-flow constructed wetlands (VF CWs). Horizontal-subsurface-flow CWs are designed systems where the wastewater flows horizontally in filter substrate media. In VF CWs, wastewater is loaded on top of the bed and allowed to flow through the substrate media, to be collected by a perforated drainage pipe network at the bottom. Such a flow regime offers VF CWs advantages since it requires less space and provides higher levels of oxygen transfer into the substrate matrix [38]. They are also generally reported to perform better than the HF SSF CWs for the reduction of BOD_5 [4]. The advantages of VF CWs have seen their wide use in wastewater treatment. While there are merits for each CW design, their unique challenges have resulted in a combined wetland system called hybrid constructed wetland [18]. The system exhibits the combined advantages of both designs and studies have reported higher contaminant removal efficiencies [39].

Regardless of the several improvements to the designs, it is important to note that contaminant removal efficiency in CWs is influenced by various factors such as temperature, hydraulic loading rate, retention time, and type of pollutant [40]. Although very high contaminant removal ranges of 60–90% of nitrogen, 70–90% of phosphorous, 90–99% of biochemical oxygen demand, 80% of heavy metals, and 90–99% of total suspended solids are possible [36], it is important to note that the performances are temporal and dependent on the location of the CWs.

Although there are several studies, reviews, and books reporting the potential advantages and significant contaminant removal efficiencies of various forms of CWs, it is of great concern that their application is still lacking in developing countries. Most of the reported work is based on designs that have been developed in developed countries. The lack of information on the performance of CWs, combined with a lack of local expertise and financial capabilities, is thought to be hampering the uptake and use of this technology in developing countries [27,37]. However, the few studies conducted in Africa have shown that CWs can significantly improve the quality of various types of wastewater [9,22,27,28], despite some reporting contaminant concentrations above acceptable levels for water re-use. For southern Africa, a few studies were conducted, with much of the work performed in South Africa. However, the few studies available show inconsistencies in performance and hence there is a need for further studies to optimize and customize the designs to the operational climatic conditions. Several knowledge gaps need to be addressed so that the technology can be widely applied on a large scale, including in rural communities.

Despite the tremendous potential observed in other countries in Europe, Australia, and the United States of America, there is currently little knowledge on the application of CW systems in wastewater treatment in Zimbabwe and the southern African region in general. There are knowledge gaps with regard to CW designs and their appropriateness for operation in the climatic and social context of Zimbabwe. It is against this background that this study seeks to evaluate design considerations for the CW system and the potential application of vegetated constructed wetlands planted with *T. latifolia* as a viable option for wastewater treatment. The specific objectives are to evaluate the treatment performances of VF CWs for removing biochemical oxygen demand (BOD_5), chemical oxygen demand (COD), total suspended solids (TSS), total nitrogen (TN), total phosphorus (TP), and the bacterial indicators of fecal contamination.

2. Methods and Materials

2.1. Study Area

This study was conducted in Bindura (17°18′ S, 31°20′ E, altitude of 1100 m) [41], a mining town located in the Mashonaland central province of Zimbabwe. The town is located 89 kilometers north-east of the capital city of Zimbabwe, Harare (Figure 1). Bindura is characterized by a subtropical climate, experiencing distinct cold, hot, wet, and dry seasons. The rainy season, which lasts from November to March, is marked by drizzle and convectional thunderstorms, with January and February receiving the most rainfall [42]. The town has an average annual rainfall and temperature of 800 mm and 28 °C, respectively [42]. The hottest months are October and November, whereas the coldest months are the winter months (May to August). However, Bindura has seen the effects of climate change just like any other region in the country, with more powerful and frequent extreme weather events (including floods, droughts, and tropical storms) as well as more erratic rainfall [43].

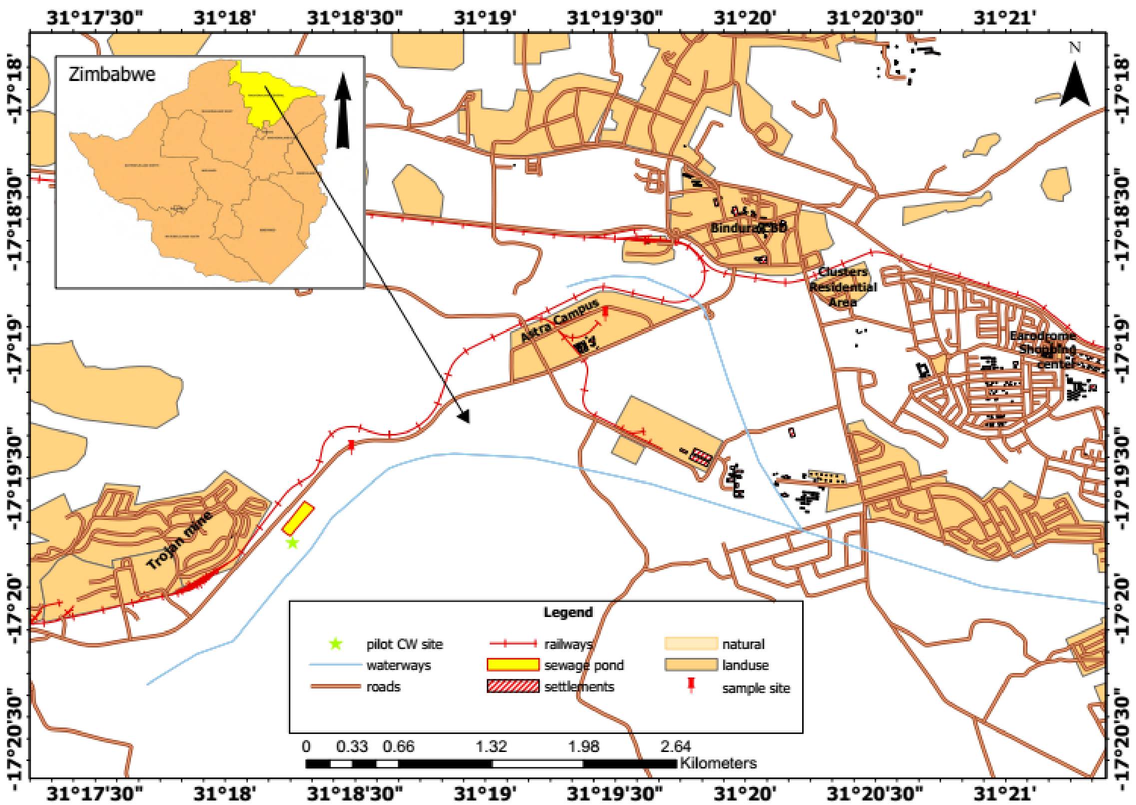

Figure 1. Map showing the study area and location of Pilot VF CWs.

2.2. Design and Installation of Pilot Vertical-Flow Constructed Wetlands

Pilot vertical-flow constructed wetlands (VF CWs) were constructed at the Trojan Mine Wastewater Sewage treatment plant in Bindura, Zimbabwe. Four pilot VF CW units were constructed with concrete and each unit had a tank volume of 1.1 m^3 and a surface area of 1 m^2, see Figure 2.

Figure 2. Layout of a VF CW. (**a**) Section of one unit showing the aeration pipe layout, (**b**) a unit being filled with rock pebbles, (**c**) four VF CWs, three planted with *T. latifolia* and one unplanted, and (**d**) a VF CW connected to reservoirs of raw sewage.

In each unit, perforated pipes were laid at the bottom of each tank and collected the water that drained vertically through the substrate. This drainage pipe was connected outside the tank with a vertical standpipe that would help adjust the water level within the unit. The drainage pipe was placed inside the bottom layer of cobbles. Within the unit, the drainage pipe was connected to a vertical aeration pipe that was opened at the top, creating passive aeration in the bed.

Each unit was filled with well-washed Nickel mine waste rock cobbles of three different sizes. Before filling the tanks, the porosity of each rock cobble size was determined using a graduated 1 L glass cylinder. Layers of substrate size and their thicknesses from top to bottom were as follows: 40 cm of fine stones (9 mm), 20 cm of medium-sized stones (mean diameter 19 mm), and 20 cm of large cobbles (mean diameter = 48 mm). Three of the VF CW units (CW1, CW2, and CW3) were planted with *T. latifolia* and the fourth one (CW4) was unplanted (control). Once planted, a commissioning phase of 3 months was allowed for the plants to grow and establish. During that period, the units were fed with a light wastewater load mixed with tap water.

After the plants had fully established, wastewater was applied in batch mode, i.e., two batches per day of equal doses of 110 dm^3, giving a hydraulic residence time of eight days. The units received raw domestic wastewater. The inflow wastewater was distributed across the entire surface of each unit and drained vertically down through the porous media and collected at the bottom by the drainage pipes. The water level inside each unit was kept at 2 cm below surface level.

2.3. Wastewater Sampling

Sampling was performed twice a month from May to August 2023. Composite samples of wastewater were aseptically collected, using sterile 500 mL bottles, from the inlet (Influent) and outlets (Effluent1, Effluent2, and Effluent4 from CW1, CW2, and CW4, respectively). The bottles were kept in an ice-cooled box (maintained at 4 °C) and transported to Bindura University of Science Education's Biological Sciences laboratory for analyses of fecal contamination, targeting indicator bacteria (*E. coli*), and Globe-Scie Laboratories for the physico-chemical characteristics of both influent and effluent samples.

2.4. Physico-Chemical and Microbial Analysis

Laboratory analysis procedures used to determine pH, electrical conductivity (EC), total dissolved solids (TDS), total suspended solids (TSS), total nitrogen (TN), total phosphates (TP), chemical oxygen demand (COD), and biological oxygen demand (BOD_5) followed APHA 2017 methods [44]. The physico-chemical parameters of dissolved oxygen (DO), electrical conductivity (EC), and potential of hydrogen (pH) were measured using a multi-parameter probe from HANNA Instruments, Smithfield, RI, USA. Nitrates, phosphorus, nitrogen, and TSS were measured using an HT100-type Photometer machine, WagTech Projects Company, London, UK. Chemical Oxygen Demand was measured by HT Hydrotest COD machine, WagTech Project Company, London, UK. To determine the BOD_5, the Warburg method was used. Briefly, initial DO levels were measured on day one using the multi-parameter probe. The wastewater samples were placed in a respirometer and incubated at 20 °C in the dark for five days. Final levels of DO were then measured and BOD was calculated using the following formula: (Initial DO-Final DO) × Dilution factor. COD was analyzed according to the standard dichromate method [44]. Briefly, samples were mixed with potassium dichromate, which acts as an oxidizing agent. Samples were refluxed for two hours and the reduction in dichromate ions caused a change in color. The change in color was measured using a spectrophotometer and absorbance was correlated to COD concentrations using a calibration curve. Nitrates in wastewater samples were analyzed using ultraviolet spectrophotometric screening, standard method 4500-NO_3^- B [44]. A 50 mL wastewater sample was thoroughly mixed with 1 mL HCl solution to prevent interference from hydroxide or carbonate concentrations up to 1000 mg $CaCO_3$/L. Standard curves were prepared using NO_3^- calibration standards in the range of 0 to 7 mg NO_3^- N/L. Absorbance or transmittance was read against distilled water set at zero absorbance or 100% transmittance. A wavelength of 220 nm was used to obtain NO_3^- readings and a wavelength of 275 nm was used to determine interference due to dissolved organic matter. Using the corrected sample absorbance, the sample concentrations were obtained directly from the standard curve. To determine the TSS in mg/L, a well-mixed sample was filtered through a weighed standard glass fiber filter. The residue retained on the filter was dried to a constant weight at 105 °C. The gain in the weight of the fiber filter was the suspended solids. The phosphate concentration was determined using the spectrophotometric method at a wavelength of 370 nm. Ortho-phosphate reacts with ammonium molybdate to form molybdo-phosphoric acid. This is reduced by ascorbic acid to the blue complex known as molybdenum blue. The intensity of the color is proportional to the concentration of the phosphate ion in the sample and this is measured using a spectrophotometer.

The removal rates for each physico-chemical pollutant under study were calculated as follows:

$$R = 100 \times \frac{\left(C_{In} - C_{Ef}\right)}{C_{In}} \quad (1)$$

where R is the percentage removal of the contaminant (%), C_{In} is the mean contaminant concentration influent, and C_{Ef} is the mean contaminant concentration effluent.

2.5. Enumeration of *E. coli* in Influent and Effluent Wastewater

The concentration of *E. coli* in the influent and effluent was determined bi-weekly for the period. The collected sample wastewater was serially diluted, by a factor of 10, to determine the concentration of *E. coli* according to a modified protocol of [45]. Briefly, aliquots (100 µL) of the serially diluted samples were spread on MacConkey agar (Himedia Laboratories, India) in four replicates. The plates were incubated at 37 °C for 24–48 h and enumeration was conducted by counting donut-shape pink-to-dark-pink colonies on MacConkey agar. The percentage removal rate of the contaminant was determined using Equation (1).

2.6. Statistical Analysis

Data cleaning, coding, and data analysis were performed using R software version 4.1.3 (10 March 2022). The level of statistical significance was set at 5% and all tests were two-sided. The first step in analyzing data was to perform exploratory data analysis. Exploratory data analysis was performed to obtain an overview of the data structure and to highlight some of the features of the data. It involved the use of descriptive statistics and graphical techniques. Descriptive statistics were used to summarize and understand the central tendency and variability of physico-chemical properties and *E. coli* removal. To determine the efficiency of the selected macrophytes in removing enteropathogens from water comparisons between the influent, effluent 1, effluent 2, and control effluent 4 were analyzed using the ANOVA test.

3. Results

3.1. Physico-Chemical Parameters

The performance of the VF CW was assessed using the following physico-chemical parameters, pH, EC, TDS, TSS, TN, TP, COD, and BOD, and the results are presented in boxplots in Figures 3 and 4. The mean concentrations of analyzed parameters in the influent and effluent varied considerably during the study period, as shown in Figure 3.

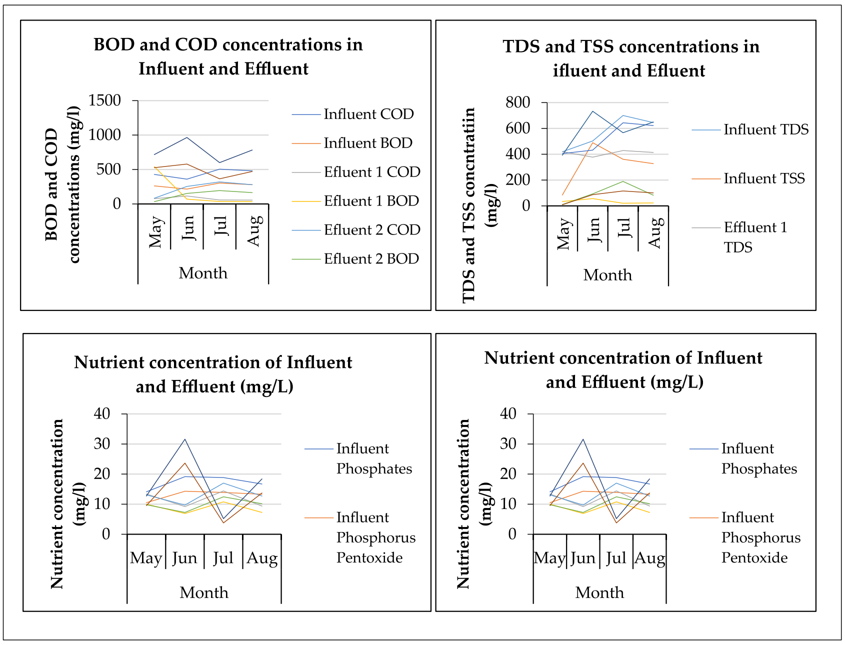

Figure 3. Mean monthly concentrations of chemical parameters of influent and effluent samples.

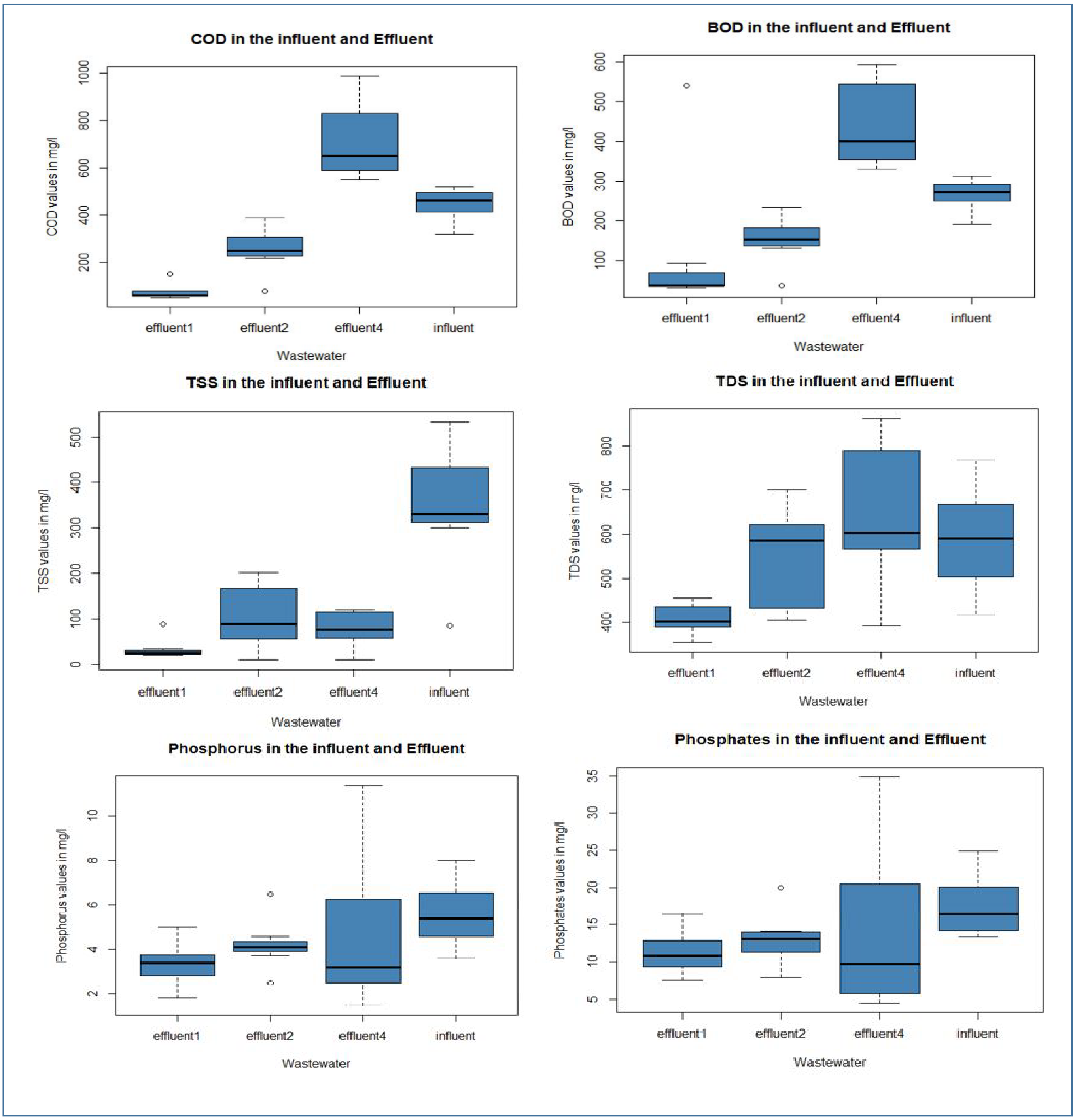

Figure 4. Boxplots diagrams for BOD, COD, TDS, TSS, Phosphorus, and Phosphate content in wastewater.

The mean concentrations of the chemical parameters for the study period are summarized in box plots in Figure 4.

The removal efficiencies of physico-chemical parameters from domestic wastewater by the CWs were generally moderate but highly variable. The concentrations of the analyzed parameters were significantly higher in the influent than in the effluent. Except for TN and TDS, all other tested parameters showed significant differences between effluents from vegetated units and the control experiment. The pilot CWs, CW1 and CW2, achieved BOD_5 reductions of 56.01% and 43.23%, respectively. The ANOVA test produced a significant p-value of <0.0001. The pairwise comparison results, using the Bonferroni correction, showed that the amount of BOD_5 in effluent1 and effluent4 differed, as well as in effluent4 and influent2 and effluent4 and the influent. Therefore, we have evidence to conclude that there are differences in the observed BOD in both the wastewater inlet and the outlets.

The vegetated pilot CWs achieved maximum COD reductions of 82.87%. The ANOVA test produced a significant p-value of <0.0001; therefore, the mean values for COD in wastewater in the inlet and the outlets differed. The pairwise comparison results using the Bonferroni correction show that all the pairwise comparisons are significant and therefore the amounts of COD in all the effluents and the influent differed.

Percentage removals of 30.61% and 7.65% were achieved in CW1 and CW2, respectively, for TDS (Figure 5). There was a significant difference in mean TDS concentrations between effluent1 and effluent4 ($p = 0.0056$). On the other hand, the p-value between Effluent1 and Influent is 0.0639, Effluent2 and Effluent4 is 0.6122, and Effluent2 and Influent is 1.000, and these values are not significant. Thus, the TDS in Effluent2 did not differ from the TDS in Effluent4 and Influent. A significant reduction in TSS concentration was observed in the effluent. The units planted with *T. latifolia* recorded TSS removals of 90.40% and 69.54% for CW1 and CW2, respectively. There was a significant difference ($p < 0.0001$) between influent and effluent TSS concentrations. Although there are differences in influent and effluent TSS amounts, the amounts of TSS in effluent1 and effluent2 and the control effluent4 were not different ($p \geq 0.05$).

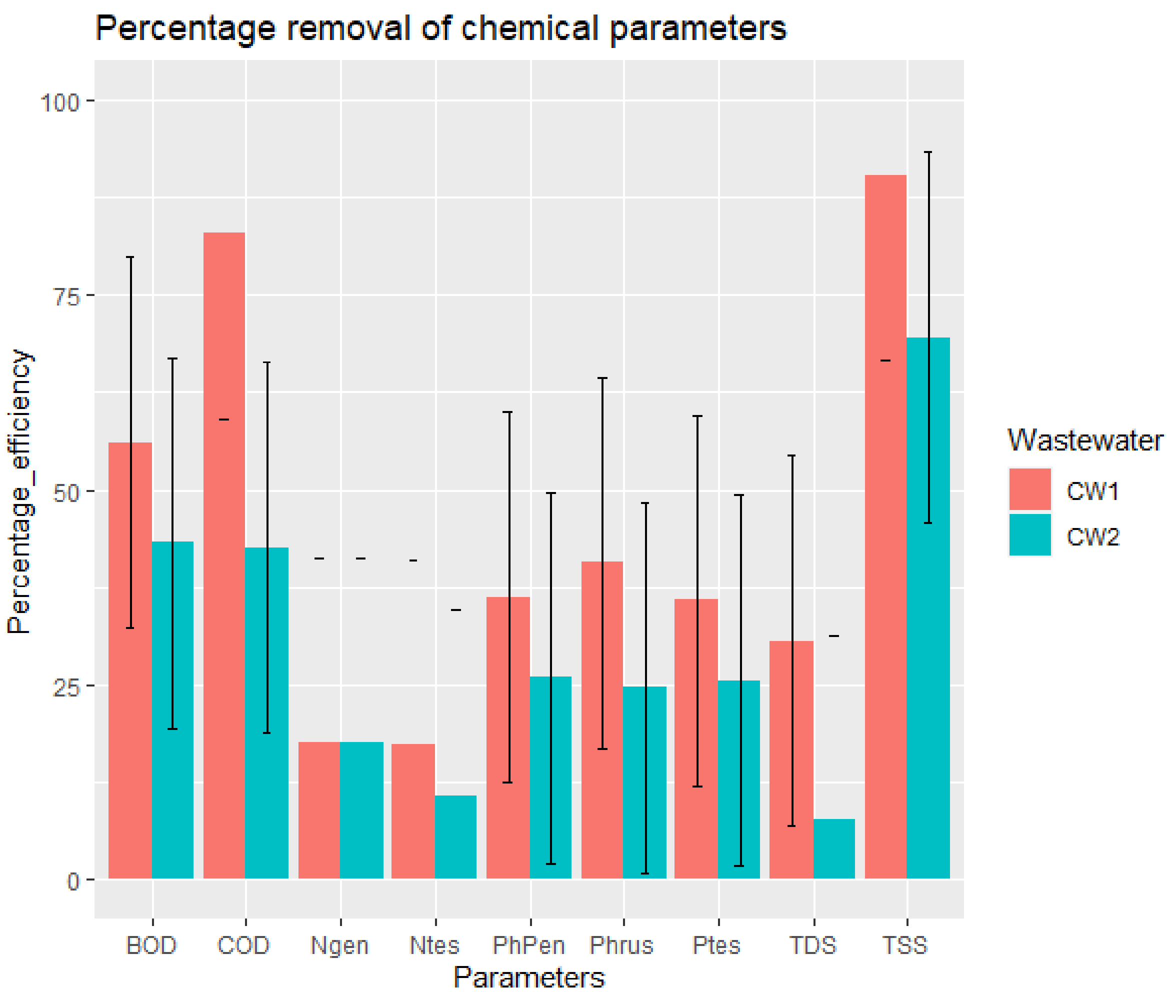

Figure 5. Percentage physico-chemical parameter removals for constructed wetlands 1 and 2 (CW1 and CW2, respectively). Ngen—nitrogen, Ntes—nitrates, PhPen—phosphorus pentoxide, Phrus—phosphorus, Ptes—phosphates.

The *p*-values for the mean values of phosphate, phosphorus pentoxide, phosphorus, and nitrates are 0.386, 0.353, 0.297, and 0.239, respectively, and are not significant. Thus, the amounts of these chemical compounds in the wastewater inlet and outlets do not differ. The *p*-value for nitrogen (0.00415) is significant. The Bonferoni *p*-values comparing nitrogen in effluent1 and effluent4 and effluent2 and effluent4 are 0.0092 and 0.0135, which are significant, meaning that the amount of nitrogen in wastewater in the outlet (effluent4, which was the control) differed from the amount of nitrogen in effluent1 and effluent2.

3.2. E. coli Removal

The *E. coli* removal data were collected in four replicates in each of the units, that is, in influent, effluent1, 2, and 4. The four replicates were averaged, which resulted in one single value for *E. coli* removal per each observation.

The concentration of *E. coli* (cfu/mL) in the influent and effluents varied in both planted (CW1 and CW2) and unplanted (CW4) units and are presented in box plot diagrams in Figure 6.

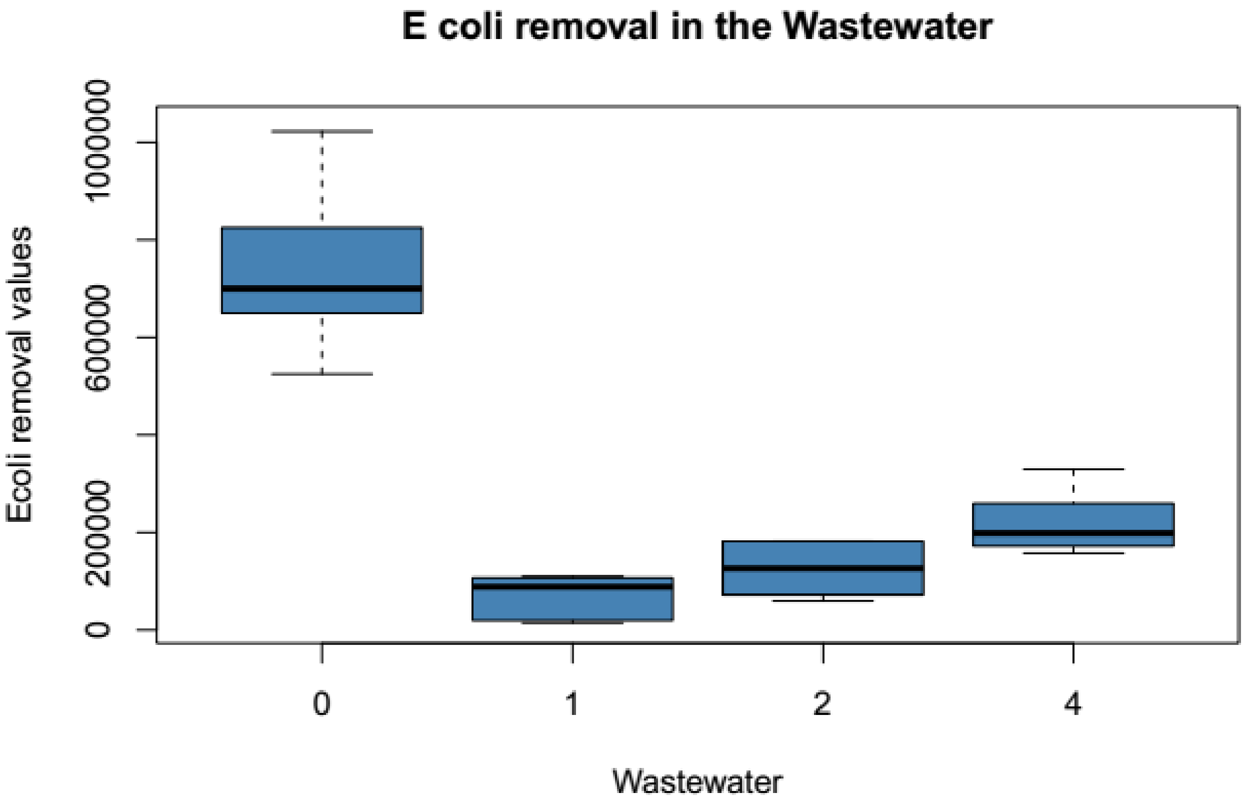

Figure 6. Boxplot diagrams for mean *E. coli* concentrations in the influent (0) and effluents (1, 2, and 4).

The concentrations of *E. coli* in wastewater were reduced from as high as 7.37×10^5 cfu/mL in the influent to as low as 7.17×10^4 cfu/mL in the effluent from planted CWs. The ANOVA test results show that there are significantly higher concentrations ($p < 0.0001$) of *E. coli* in the influent than in the effluents. Pairwise comparisons using the Bonferroni correction show that the removal of *E. coli* was different in effluent1, effluent2, and effluent4 compared to the influent ($p < 0.0001$). The vegetated CWs achieved overall *E. coli* percentage removals of 90.28% and 83.02% in units CW1 and CW2, respectively, and 70.21% in the unplanted unit CW4 (Figure 7).

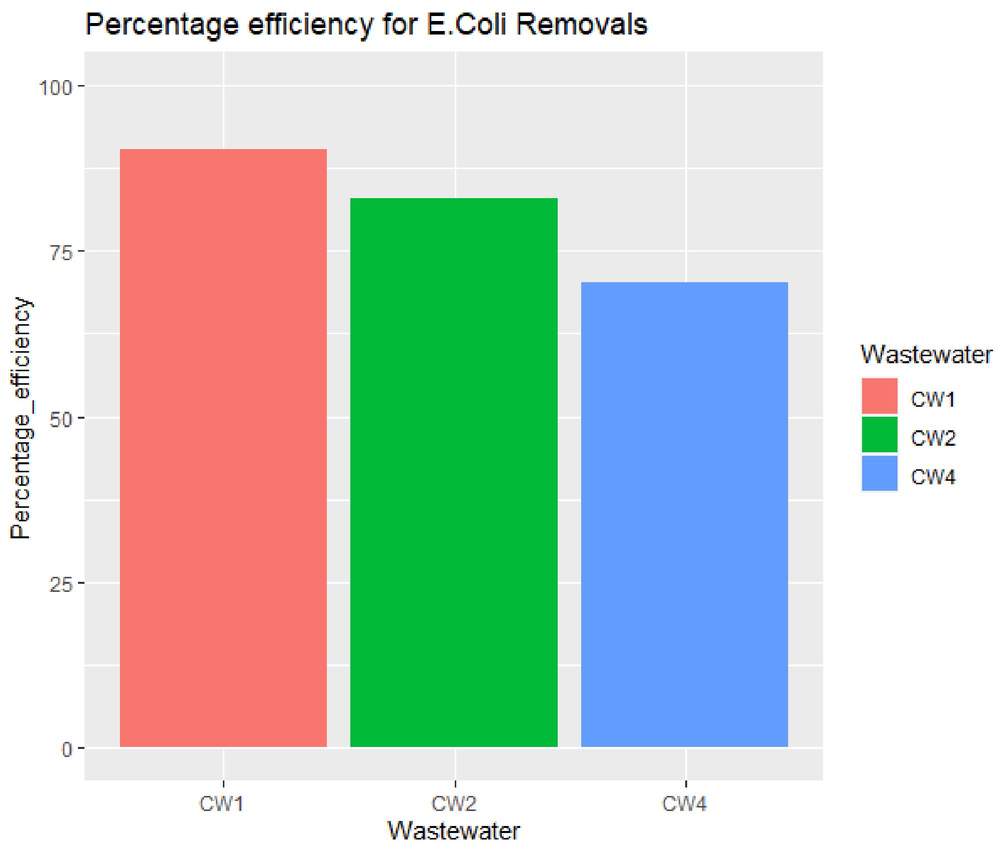

Figure 7. *E coli* removal for planted pilot VFCW CW1 and CW2.

4. Discussion

In this study, performance, in terms of contaminant removal, was evaluated in *T. latifolia*-vegetated VFCWs. Different rates of contaminant removal were observed in vegetated and un-vegetated VFCWs. Higher contaminant removal percentages were achieved in planted CW units, an indication that the presence of the macrophyte, *T. latifolia*, improved the performance of the CWs. Furthermore, the efficiencies of the studied CW system are consistent with other studies conducted under similar climatic conditions. However, there were no significant differences in the concentrations of TN and TDS between planted and unplanted (control) setups.

The mean TSS removal efficiency of 80% (and the highest of 90.40%) observed in this study is consistent with those reported in the literature. Elfanssi et al. [22] report that TSS removal in a typical CW system ranges from 90.78% in the cold season to 98.03% in the warm season in hybrid CWs planted with *P. australis*. Although there was a significant difference ($p < 0.0001$) between influent and effluent TSS concentrations, there was no difference in TSS removal between the vegetated CWs and the control in this study. Such a result was observed in a few other studies, including Konnerup et al. [46] and Manios et al. [47]. However, this is contrary to results reported in many studies where macrophytes influenced TSS removals [27]. Nevertheless, the maximum TSS removal of 90.4% in the vegetated wetland against the control, with a removal efficiency of 77.5%, suggests that the macrophyte *T. latifolia* played a role in reducing TSS concentrations. The high removal rates observed in both the vegetated and the control setups can be attributed to the physical processes taking place within the CW, including filtration, adsorption, and sedimentation [48]. In addition, the coarse waste rock particles used could have improved TSS removal, as a study by Manios et al. [47] revealed that wetland beds with coarse gravel performed better than beds with soil, sand, and compost. Nevertheless, studies have observed that macrophytes contribute to TSS removal through the establishment of a root and rhizome system capable of stabilizing the wetland bed, increasing the interception and sedimentation of suspended particles [49]. It is important to note that effluent TSS concentrations are generally not related to effluent concentrations, as wetland processes result in TSS contribution and thus irreducible background TSS concentrations [31].

Biological oxygen demand and COD are important parameters that are used to measure organic pollution. In this study, the efficiency of the VFCW was measured using BOD_5 and COD concentrations in the influent and effluent. This study obtained mean BOD_5 and COD removals of 48.24% and 62.74%, respectively. Several studies have evaluated the performance of VFCWs, and a wide range of values have been reported (e.g., [26,50–52]). It is important to highlight that the BOD_5 and COD removal efficiency is affected by various factors, including season [22], concentration in the influent [53], macrophyte type [26,36,48], substrate composition [40], detention time [27,54], and hydraulic loading rate [31,55].

The BOD percentage removal achieved in this study is below the 57–74% range obtained by Mashauri [50] using HSSF-CW planted with *Typha latifolia*. Even higher removal efficiencies were reported in the literature, e.g., BOD_5 (93.47%) and COD (91.40%) were obtained in a hybrid CW planted with *Pragmites australis* [22]. These findings suggest that wetland design and operation parameters employed in this study may have limited its capacity to remove organic contaminants from domestic wastewater. Therefore, there is a need to optimize the factors affecting efficiency, including loading rate and hydraulic retention time, to enhance the system's efficiency.

The higher organic pollutant removals can be attributed to both physical and biological processes taking place within the wetland media matrix. The media and plant roots provide a surface for microbial attachment. The microbes are responsible for the degradation and transformation of organic contaminants [40]. Higher BOD_5 removals in vegetated CWs are attributable to the presence of *T. latifolia* roots and rhizomes providing an attachment surface for bacteria, which are responsible for the degradation of organic matter [22]. Furthermore, the waste rock particles could have allowed the percolation of organic solids, trapping them within the media matrix for a long time and thereby promoting greater biodegradation.

Although only *T. latifolia* was used in this study, several other macrophyte species have been evaluated in CWs and proven to be efficient in organic matter removal. For example, *Heliconia* spp.-vegetated CWs produced COD removals ranging from 55–70% in a tropical climate [26] and *Typha domingensis* had a COD removal of 68.7% [56]. However, these removal efficiencies were achieved in horizontal subsurface CWs.

Although the CWs could remove the organic pollutants, significantly low nutrient removal efficiencies were observed, with the highest being total phosphate at 40.64%. These results are consistent with the findings of Vymazal [57], but contrary to the findings by Calheiros et al. [53], who reported higher nutrient removals, e.g., 89% for total phosphate. Our findings also showed no significant difference in nutrient removals between vegetated and control CWs, suggesting that the macrophytes have no role in nutrient removal. However, several studies report that the macrophyte rhizosphere promotes microbial community and activity by providing a root surface for microbial growth, a carbon source, and a micro-aerobic environment [27]. Such conditions promote biogeochemical processes that effectively remove nutrients from wastewater. The relatively larger particle sizes of media used in this study provide a small surface area for microbial attachment and hence lower biogeochemical cycling of nutrients. Furthermore, the waste rock used lacked the elements Fe, Ca, and Al, which are necessary for ligand exchange reactions with phosphates. In addition, the batch loading of wastewater into the VF CWs prevented the formation of anoxic conditions within the media matrix, a condition that is important for denitrification processes [27]. Therefore, there was a small decrease in nitrates.

Microbial Contaminant Removal

Several studies have demonstrated the efficiency of VFCWs in removing microbial contaminants from wastewater (e.g., [19,58,59]). Nevertheless, the studies used fecal indicator bacteria such as *E. coli* [19] to determine microbial contaminant removal efficiencies. In this study, *E. coli* was used as a fecal contamination indicator. This study achieved 83.02–90.28%

E. coli removal from domestic wastewater, which is slightly lower than those achieved in earlier studies, i.e., 98.9% *E. coli* removal [60] and 99% coliform removal [59]. The high *E. coli* removal achieved in this study is a result of the enhanced aeration provided. A study by Headley et al. [61] has shown that an aerated CW achieved higher *E. coli* (3.3 $\log_{10}$ CFU reduction per 100 mL) reductions compared to non-aerated systems (1.4 $\log_{10}$ CFU reduction per 100 mL). However, other factors, including flow rate, hydraulic retention time, and influent concentrations, cannot be ruled out. Furthermore, the extensive and dynamic range of fecal bacterial pathogens, together with the current emergence of new bacterial strains [59], calls for more research to better understand the performance of CWs under sub-tropical climatic conditions.

Although most of the contaminant removal efficiencies reported in this study are consistent with results from similar studies, most fall short of the general permissible limits for discharge into the environment or re-use. More research is required to optimize the system through improvements to the wetland design and operational parameters, as well as incorporating hybrid systems to effectively reduce pollutant loads in effluents to permissible levels.

5. Conclusions

This study demonstrated that *T. latifolia* can be effectively established on mine waste rocks in VF CWs. This study also showed that CW systems can be successfully established for the removal of physical, chemical, and microbial contaminants from domestic wastewater. The vertical-flow constructed wetlands achieved moderate to high removal efficiencies for organic matter (BOD and COD), TSS, and TDS, as well as for the fecal indicator bacteria, *E. coli*. However, the system showed low nutrient removal, an aspect that requires some improvements. Research should therefore focus on optimizing the design and operation of the CWs to overcome these drawbacks and enhance contaminant removal efficiency.

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