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Long-term investigation of pollutant removal efficiency in two constructed wetlands for wastewater treatment and reuse in urban areas

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Constructed wetland (CW) systems offer many advantages for wastewater treatment in urban areas and are increasingly seen as sustainable solutions. However, their pollutant removal capacity can vary significantly, influenced by weather conditions and the specific plant species used. This paper presents a long-term study conducted on two pilot-scale horizontal subsurface flow (HSSF) CWs located in two different towns of Sicily (Italy). The main aims were to compare the pollutant removal efficiency (RE) of two HSSF CWs treating urban wastewater and to assess the effect of treated wastewater (TWW) reuse on bermudagrass [*Cynodon dactylon* (L.) Pers.] traits and soil characteristics. The two CWs had comparable surface areas and were each planted with a different species, resulting in monoculture systems. Two experimental fields of bermudagrass were set up, one for each HSSF CW. The effects of 3 years and two sources of irrigation water [TWW and freshwater (FW) as control] were assessed using a split-plot design for two-factor experiments. Results highlight that removal efficiencies up to 83% were achieved for an inlet of 55 ± 14 mg COD L^{-1}, 81% for an inlet of 31 ± 5 mg BOD_5 L^{-1}, 66% for an inlet of 20.6 ± 3.5 mg total nitrogen L^{-1}, and 50% for an inlet of 7.9 ± 0.8 mg total phosphorus L^{-1}. Both CW systems demonstrated effective long-term performance in the removal of physico-chemical and microbiological contaminants. Bermudagrass had higher above-ground biomass production (1,358.74 kg ha^{-1}) in TWW-irrigated plots than those plots irrigated with FW (1,005.98 kg ha^{-1}), on average. The highest biomass yields were recorded during the second and third years of the study. Visual turf quality ratings were consistently similar across years and irrigation treatments. No significant variations in soil pH were observed between FW- and TWW-irrigated soils. However, soils irrigated with TWW showed higher salinity, organic matter, macronutrients, and sodium levels, on average.

KEYWORDS

bermudagrass, climatic conditions, nature-based solution, phytoremediation performance, soil, water management

1 Introduction

Constructed wetland (CW) systems are engineered systems designed for wastewater (WW) treatment and reuse, and since the last century they have been increasingly recognized to play a strategic role in delivering a range of ecosystem services (Wu et al., 2015; Wang et al., 2021; Agaton and Guila, 2023). They are designed to remove contaminants from various types of wastewaters in a controlled environment by replicating the natural processes that occur in wetlands (Vymazal, 2014; Hassan et al., 2021; Gebru and Werkneh, 2024; Addo-Bankas et al., 2024). Microorganisms, plants, and substrates are key components of CWs, and their synergistic interaction enables these systems to achieve higher treatment efficiency compared to natural wetlands (Ji et al., 2022; Kushwaha et al., 2024). Treated wastewater (TWW) and plant biomass are the primary outputs of CWs, and their use can greatly increase the multi-functionality of these systems (Masi et al., 2017; Takavakoglou et al., 2022). In particular, in the semi-arid environment, TWW is an effective alternative water supply, and its use forms one of the most promising strategies for sustainable water management, due to its potential environmental benefits for soil health and water quality (Stefanakis, 2019; Franci Gonçalves et al., 2021; Licata et al., 2019; de Campos and Soto, 2024). As widely supported by existing research (Licata et al., 2019; Shtull-Trauring et al., 2022; Muscarella et al., 2024), TWW contains inorganic and organic nutrients that can be incorporated into the soil and exploited by crops, thereby supporting soil fertility and reducing the need for use of mineral fertilizers. Other studies (Hashem and Qi, 2021; Ofori et al., 2021; Hajjar et al., 2025) have also reported supplemental benefits provided by the application of TWW irrigation. In addition to this, the harvested biomass from CWs can be used as a fertilizer or soil conditioner or repurposed as a renewable energy source through combustion, bioethanol production, or biogas generation (Avellan et al., 2007; Rodriguez-Dominguez et al., 2021; Pereira et al., 2022). The reuse of both TWW and plant biomass depends mainly on the pollutant removal efficiency (RE) of the system. As reported by Huang et al. (2013), pollutant removal significantly affects TWW quality, and it is strongly correlated with climatic conditions and nutrient dynamics within the system. This implies that the performance of a given constructed wetland may vary under different environmental conditions (Wang et al., 2021). In particular, rainfall patterns, solar radiation intensity, and temperature trends significantly influence plant growth and evapotranspiration (ET) rates, thereby exerting a notable impact on pollutant removal efficiency, as supported by previous studies (Headley et al., 2012; Beebe et al., 2014; Tuttolomondo et al., 2016). Garfi et al. (2012) emphasized that climatic seasonality can significantly influence the performance of CWs, noting that pollutant removal efficiency is generally higher in tropical regions than in temperate zones. This is primarily attributed to the positive effects of prolonged exposure to warm conditions on plant growth and microbial activity. Several authors (Aktatos and Tsihrintzis, 2007; Zhu et al., 2018; Wang et al., 2023) agreed that increasing temperatures promote pollutant RE by promoting plant growth and microbial activity. Ávila et al. (2013) and Mittal et al. (2023) affirmed that evapotranspiration significantly affects pollutant RE by influencing the redox conditions within the system.

Some studies (Bialowiec et al., 2014; Tuttolomondo et al., 2016) have reported a correlation between evapotranspiration and pollutant RE for organic compounds, noting that as ET exceeds certain thresholds, an increase in organic matter content may be observed in the effluent. However, it is also evident that the effects of seasonal climatic conditions on pollutant RE should be analyzed by taking into account the plant species and cropping system used in the CW. From this point of view, few studies have reported that the choice of plant species and cropping systems can have a greater influence on the pollutant RE of CWs than other design features. It is well documented (Vymazal, 2011; Zhang et al., 2012; Ahmed and Kareem, 2024) that plant roots promote various physical effects and influence the hydraulic properties of the substrate, either increasing or decreasing the hydraulic retention time (HRT). This suggests that increased root density can impede wastewater flow within the substrate, effectively prolonging hydraulic retention time and leading to improved pollutant RE. Plant species vary in their capacity to remove pollutants from wastewater, with some demonstrating more consistent performance in purification across seasons compared to others (Toscano et al., 2015; Kulshreshtha et al., 2022).

Regarding cropping systems, although there is ongoing debate among researchers about the consistency of their effects, it is well established that the use of monoculture in comparison to polyculture systems leads to differing impacts on pollutant RE across seasons (Marín-Muñiz et al., 2020). According to Calheiros et al. (2015) and Carrillo et al. (2023), CWs using polyculture systems may achieve higher pollutant RE as the more diverse root distribution provides a favorable habitat for a wider range of microorganisms. On the contrary, other authors argued that CWs with monoculture systems produce greater above-ground biomass and exhibit greater stability compared to polyculture systems, particularly concerning competition levels for nutrients among plants and climate responses (Zhang et al., 2007; Liang et al., 2011). In Sicily (Italy), CWs have been used to treat different types of wastewater for several decades (Cirelli et al., 2006; Barbera et al., 2009; Toscano et al., 2015; Licata et al., 2019). Furthermore, in-depth investigations into the technical and vegetative aspects of these systems have been conducted, yielding valuable insights. However, no studies to date have compared the performance of CWs under different climatic conditions within Sicily. With this in mind, the main aims of this study were to assess i) the long-term pollutant RE of two horizontal subsurface flow system (HSSF) CWs in Sicily, ii) the medium-term effects of TWW irrigation on yield parameters of bermudagrass [*Cynodon dactylon* (L.) Pers.] plants, and iii) the medium-term effects of TWW irrigation on soil characteristics.

2 Materials and methods

2.1 Pilot-scale constructed wetland systems

2.1.1 CW 1

CW 1 was located in an urban park in Raffadali, a small town in the West of Sicily (Italy) (37°24′N–1°05′E, 440 m a.s.l.). The park was

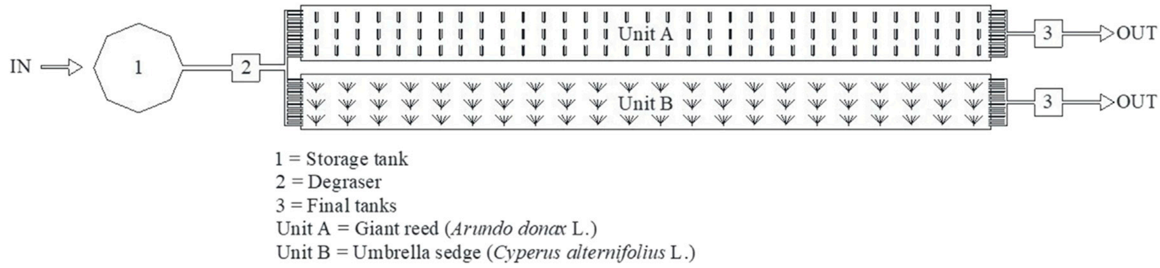

FIGURE 1
Layout of CW 1.

planted with aromatic and medicinal plants, turfgrass species, orchards, and olive orchards, covering an area of 13 ha. Located in a hilly area, Raffadali has a temperate–warm climate, with mild winters and dry summers in accordance with the Köppen–Geiger climate classification (Kottek et al., 2006). Based on the 2000–2021 time-series data, provided by the SIAS (2024), the area has an average annual temperature of 17.1°C, with average minimum and maximum temperatures of 11.3°C and 23.5°C, respectively. The average annual rainfall is 508 mm.

The pilot-scale CW 1 was used for tertiary treatment of pretreated urban wastewater supplied by the town's activated sludge wastewater treatment plant. The CW plant was built in 2000 and subsequently upgraded through modifications to the substrate size and plant species. It consisted of two parallel and independent units (A and B), covering a total surface area of 100 m^2 (Figure 1). Each unit was 50 m long and 1 m wide and was constructed with concrete walls. The floor of each unit was leveled with fine sand. Both units were then filled with a uniformly graded substrate consisting of 20 mm silica quartz river gravel (Si 30.02%; Al 5.11%; Fe 6.10%; Ca 2.65%; Mg 1.05%, according to supplier specification). Each unit was provided with a slope of 1% and was lined with high-density polyethylene (HDPE) geomembrane sheets, covered with a layer of nonwoven fabric. The depth of the units was 0.50 m. Unit A was planted with giant reed (*Arundo donax* L.) at a density of 4 rhizomes m^{-2}, and Unit B, with umbrella sedge (*Cyperus alternifolius* L.) at a density of 5 stems m^{-1}. The two species were selected based on their characteristics such as rapid growth, low maintenance requirements, and high efficiency in pollutant removal. The urban wastewater was initially stored in a 15 m^3 waterproof, concrete storage tank. The tank was equipped with a submerged pump and timer to control the flow rate and distribution of the wastewater into units A and B. A degreaser was also incorporated into the system to introduce an additional stage of wastewater treatment. The wastewater was then pumped into each of the two units through a 1 m wide perforated polyvinylchloride pipe. In each unit, the inlet pipe was placed 10 cm above the surface of the substrate. Wastewater was pumped continuously throughout the day without variations in time. Finally, for each planted unit, the TWW was fed into a 5 m^3 storage tank, which was connected to a sprinkler irrigation system. The CW was fitted with a submerged pump to recirculate the TWW from the bottom to the top of the CW plant. The wastewater inflow rate (Q_i) was kept at a constant of 6 $m^3 day^{-1}$ during the trials. The units were operated using a hydraulic loading rate (HLR) of 6 cm day^{-1} and a hydraulic retention time (HRT) of 8.3 days.

2.1.2 CW 2

CW 2 was located downhill from the municipal sewage plant in Piana degli Albanesi, a small town in the West of Sicily (37°59′56″40 N–13°16′50″16 E, 740 m a.s.l). Piana degli Albanesi is located in a mountainous area in Sicily and has a temperate–humid climate, with cold winters and mild summers in accordance with the Köppen-Geiger climate classification (Kottek et al., 2006). Based on the 2000–2021 time-series data, provided by the SIAS (2024), the area has an average annual temperature of 15.3°C, an average maximum temperature of 19.2°C, and an average minimum temperature of 11.6°C. The average annual rainfall is 650 mm.

The pilot-scale CW 2 was used for tertiary treatment of pretreated urban wastewater directly pumped from the town's activated sludge wastewater treatment plant. It was designed and built in 2004 and consists of two parallel, independent units (A and B), each with a surface area of 33 m^2 (Figure 2). Each concrete unit was 33 m long and 1 m wide and filled with an evenly sized substrate of 20 mm silica quartz river gravel (Si 30.32%; Al 5.23%; Fe 6.87%; Ca 2.79%; Mg 1.01%, according to the supplier specification). Each unit had a slope of 1.5% and a depth of 0.60 m. HDPE geomembrane sheets covered with a layer of nonwoven fabric were used to line the CW units. Unit A was planted with *C. alternifolius* L., and Unit B was planted with reedmace (*Typha latifolia* L.). Plant density was 4 rhizomes m^{-2} for reedmace and 5 stems m^{-1} for umbrella sedge. Urban wastewater from the outflow tank of the activated sludge sewage plant was initially fed into a concrete storage tank. It was then pumped into the two wetland units through a perforated feed pipe system to ensure even distribution of the wastewater. The outflow tanks, located downhill from the two units, were fitted with a grid to promote additional filtration and prevent potential clogging of the substrate. The TWW flowed downhill into two 64 m^3 storage tanks, each of which was connected to a sprinkler irrigation system. The two units were operated using an HLR of 8 cm day^{-1} and an HRT of 7.4 days.

2.2 Sampling and wastewater analysis

Wastewater samples were collected from 2014 to 2018. Sampling was conducted on a monthly basis across all seasons: spring,

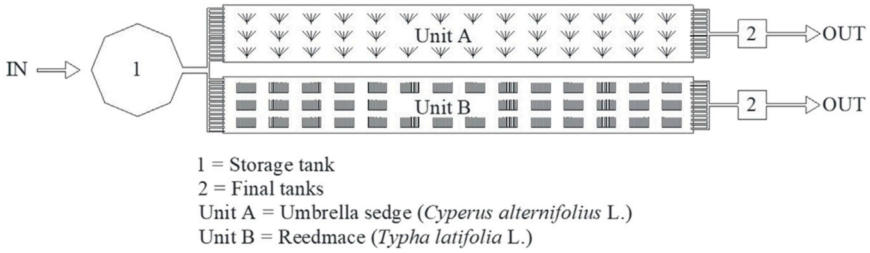

FIGURE 2
Layout of CW 2.

summer, autumn, and winter. At each sampling event, 1 L of wastewater was collected at both the inflow and outflow points of the CW units. Regarding WW analyses, total suspended solids (TSS), biochemical oxygen demand (BOD_5), chemical oxygen demand (COD), total nitrogen (TN), and total phosphorus (TP) were determined in a laboratory using Italian water analytical methods (APAT, 2003). Total coliform (TC) and *Escherichia coli* (*E. coli*) levels were determined using membrane filter methods, based on standard methods for water testing (American Public Health Association, 1998). The pollutant RE of the two CWs was calculated based on Equation 1 provided by the International Water Association (Kadlec et al., 2000):

$$RE = \frac{(C_i - C_0)}{C_i}. \tag{1}$$

The pH and electrical conductivity (EC) were determined at the time of sample collection using a portable Universal meter (Multiline WTW P4).

2.3 Bermudagrass experimental fields and main cultivation practices

Bermudagrass was tested over a 3-year period through the establishment of two separate experimental fields at Raffadali (EXF 1) and Piana degli Albanesi (EXF 2). The Tifway variety was used for the bermudagrass tests. At EXF 1, the soil was classified as clay loam (40% sand, 21% silt, and 39% clay) and identified as Regosols (typic Xerorthents), according to the United States Department of Agriculture (USDA, 1999). At EXF 2, the soil type was classified as sandy clay loam (54% sand, 23% silt, and 23% clay) and identified as Aric Regosol by USDA (1999).

Experimental plots were arranged in a split-plot design (Gomez and Gomez, 1984) to evaluate the effects of two factors, with three replications over three growing seasons. The main plot factor was year (Y), with three treatment levels: Y_1 (2016), Y_2 (2017), and Y_3 (2018). The sub-plot factor was irrigation water (IW), with two treatment levels: IW_1 [freshwater (FW), as control] and IW_2 and IW_3 (TWW from planted units). At EXF 1, IW_2 and IW_3 were produced by giant reed- and umbrella sedge-planted units, respectively. At EXF 2, IW_2 was produced by the umbrella sedge-planted unit, while IW_3 was obtained from the reedmace-planted unit. In both experimental fields, each plot measured 2.25 m^2 (1.5 m × 1.5 m) and was spaced 40 cm apart. A conventional herbicide [N-(phosphonomethyl)glycine] was used for weed control, applying a standard dose of 4 kg ha^{-1} $year^{-1}$. Both fields were managed with conventional tillage and fertilization. Each field was equipped with three sprinkler irrigation systems, one for each source of irrigation water in the study. Irrigation was applied from April to September, with events scheduled three times per week. On average, 80 m^3 ha^{-1} of water was applied during each irrigation event. The water requirement of bermudagrass was estimated by calculating the difference between ET losses and rainfall using meteorological data from a local weather station. Equation 2 was used to calculate crop evapotranspiration (ET_c):

$$ET_c = ET_0 \times K_c, \tag{2}$$

where ET_0 is the reference evapotranspiration and K_c is the crop coefficient of the species. The irrigation volume was calculated according to Equation 3:

$$V = 10,000 \times (FC - WP) \times \rho \times H, \tag{3}$$

where 10,000 is the area of 1 ha, FC is the soil water content at field capacity, WP is the soil water content at wilting point, ρ is the bulk density of soil, and H is the height of the soil layer from wet, equivalent to the rooting depth of the species.

Fertilization was applied from April to September, with each plot receiving monthly rates of 50 kg ha^{-1} N, 10 kg ha^{-1} P, and 20 kg ha^{-1} K. Mowing was carried out twice per week on average using a helicoidal mower, maintaining a cutting height of 3 cm. In July and August, the mowing frequency increased due to greater vegetation growth. No fungicide and insecticide treatments were carried out.

2.4 Plant characteristics

Plant trait measurements included both morphological and productive traits. Leaf texture was assessed monthly by randomly collecting 100 flattened leaves per sub-plot. Leaf width was measured 1 cm above the ligule, as described by Leto et al. (2008). In June and September, shoot density was assessed by counting the number of shoots within a 50 cm^2 core sample for each sub-plot, following the method described by Croce et al. (2002). Above-ground dry biomass was calculated by removing all plant tissues from the core top and drying the collected material in an oven at 60°C to constant weight

(Magni et al., 2014). A grass sample was taken randomly from each sub-plot of each irrigation treatment in June and September. Regarding qualitative parameters, visual turfgrass quality was based on a 1 (= poorest or dead) to 9 (= outstanding or ideal) visual rating scale (Leto et al., 2008).

2.5 Soil characteristics

Measurements were carried out on the topsoil (0.30 m). Before transplanting, three soil samples were randomly collected from each replication and analyzed. At the end of the study, one soil sample per subplot was collected for each replication and analyzed. Soil samples were air-dried, ground, and sieved through a 2 mm mesh screen prior to analysis of their chemical and physical traits. The samples were analyzed for pH and EC in the ratio of 1:2 dry soil: water extract, according to official methods for chemical soil analysis provided by Italian Ministry of Agricultural Policies (1999). Total organic carbon (TOC) was determined using the Walkley and Black method (Nelson and Sommers, 1996) (±0.01%), total Kjeldahl nitrogen (TKN) was determined using the Kjeldahl procedure (Nelson and Sommers, 1998) (±0.02, g kg^{-1}), and the content of assimilable phosphorus as $H_2PO_4^-$ (P) was determined using the Olsen method (Pansu and Gautheyrou, 2006) (±0.02, ppm). The Na content (±0.09, ppm) was determined by using an atomic absorption spectrophotometer. All the analyses were carried out at the CoRiSSIA Research Center in Palermo (Italy).

2.6 Weather data

Weather data for the two experimental sites were obtained from two meteorological stations, operated by the SIAS (2024). The stations were equipped with an MTX data logger (model WST1800) and various sensors. Sensors provided data on maximum and minimum air temperatures, 10-day cumulative rainfall, relative humidity (RH), total radiation (TR), leaf wetness (LW), and reference evapotranspiration (EVP). Monthly data for 5 years are presented in Supplementary Tables S1, S2.

2.7 Statistical analyses

Statistical analysis was performed using the Minitab software (Minitab, Ltd., Coventry, United Kingdom: version Release 19). Data on the treatment performance of the CWs were analyzed using mean ± standard deviation. Two-way analysis of variance (ANOVA) was applied to analyze the data for plant and soil characteristics, and mean comparisons were performed using Tukey's test ($p \leq 0.05$).

3 Results and discussion

3.1 Performance of the experimental systems CW 1 and CW 2

In the case of CW 1, based on data from the meteorological station, the minimum temperature recorded during the study period was 4.2°C, and the maximum temperature recorded was 35.0°C. The average rainfall was 46.8 mm, and minimum and maximum RH were 21% and 96.7%, respectively. The mean leaf wetness was 472 min, and mean evapotranspiration was 99 mm. Table 1 and Figure 3 present the results of the physicochemical analysis conducted at the inlet and outlet of units A and B, planted with *A. donax* and *C. alternifolius,* respectively, over multiple years and across different seasons.

At the outlets of constructed wetland units 1 and 2 maximum and minimum COD concentrations ranged from 11 to 31 mg L^{-1}, corresponding to inlet values, which varied between 38 and 76 mg L^{-1}. The maximum and minimum BOD_5 concentrations at the outlet ranged from 5 to 23 mg L^{-1}, corresponding to inlet values between 23 and 43 mg L^{-1}. The BOD/COD ratio varied from 0.3 to 0.8. The TSS concentration at the inlet varied between 24 and 66 mg L^{-1}, and that at the outlet ranged from 6 to 22 mg L^{-1}. TN and TP values at the inlet varied between 12.7 and 29.8 mg L^{-1} and 6.6 and 10.3 mg L^{-1}, respectively, and at the outlet, TN values ranged from 6.1 to 17.7, while those for TP varied between 3.5 and 7.1 mg L^{-1}. The average pH of the wastewater at the inlet was 7.2 ± 0.2, while pH values for outlets at units 1 and 2 ranged between 6.3 and 7.5. The inlet average conductivity was 543 ± 149 µS cm^{-1}, while conductivity at the outlets of units 1 and 2 ranged between 321 and 976 µS cm^{-1}, respectively. At the inlets of CW units A and B, total coliform counts varied between 2×10^4 and 4×10^4 CFU 100 mL^{-1}, while total coliform counts at the outlet varied between 1×10^3 and 6×10^3 CFU 100 mL^{-1}. Removal efficiency for both units was consistent throughout the year, 88% ± 4% and 87% ± 4%, respectively. At the inlets of CW units 1 and 2, *E. coli* concentrations varied between 7×10^2 and 3×10^3 CFU 100 mL^{-1}, and outlet concentrations varied between 7×10^1 and 2×10^2 CFU 100 mL^{-1}. Removal efficiency remained constant throughout the year, averaging 88% ± 4% for Unit 1 and 87% ± 4% for Unit 2. As shown in Figure 4, the organic mass loadings for units A and B varied between 23 and 45 kg ha^{-1} d^{-1}, while maximum COD mass removal values ranged between 10 and 34 kg ha^{-1} d^{-1}.

Concerning CW 2, based on data from the meteorological station, the minimum temperature recorded during the study was 1.9°C, and the maximum temperature recorded was 32.9°C. The average rainfall was 97.7 mm, and minimum and maximum RHs were 20% and 100%, respectively. The mean total radiation was 16 MJ mq^{-1}; mean leaf wetness was 276 min, and mean evapotranspiration was 9 mm. The results of the physicochemical analysis corresponding to the inlet and outlet of units A (*C. alternifolius*) and B (*T. latifolia*) over multiple years and seasons are presented in Table 2 and Figure 5.

At the outlets of constructed wetland units 1 and 2, COD concentrations ranged from 8 to 45 mg L^{-1}, corresponding to inlet values between 32 and 99 mg L^{-1}. The BOD_5 values at the outlet ranged from 6 to 23 mg L^{-1}, corresponding to inlet values between 19 and 48 mg L^{-1}. The BOD/COD ratio varied from 0.3 to 0.8. For influent TSS concentrations ranging from 25 to 73 mg L^{-1}, the CW units produced effluent concentrations between 8 and 43 mg L^{-1}. TN and TP concentrations at the inlet varied between 14.1 and 29.7 mg L^{-1} and 6.4 and 9.5 mg L^{-1}, respectively. Effluent concentrations ranged between 5.7 and 14.8 mg L^{-1} for TN and between 3.8 and 6.4 mg L^{-1} for TP. Average wastewater pH at the inlet was 7.5 ± 0.2, while effluent pH values for units 1 and 2 ranged

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TABLE 1 Average composition of the water at the inlet and outlet of the constructed wetland 1 (CW 1) from units A and B (UA and UB) for a hydraulic loading rate of 6 cm d^{-1} (mean ± SD).

Year	Season	BOD_5 [mg L^{-1}]			COD [mg L^{-1}]			TSS [mg L^{-1}]			TN [mg L^{-1}]			TP [mg L^{-1}]			Total coliforms [CFUs 100 mL^{-1}]			*E. coli* [CFUs 100 mL^{-1}]		
		Inlet	Outlet UA	Outlet UB	Inlet	Outlet UA	Outlet UB	Inlet	Outlet UA	Outlet UB	Inlet	Outlet UA	Outlet UB	Inlet	Outlet UA	Outlet UB	Inlet	Outlet UA	Outlet UB	Inlet	Outlet UA	Outlet UB
2014	Spring	28 ± 5	9 ± 3	11 ± 3	61 ± 15	16 ± 2	17 ± 6	40 ± 14	11 ± 4	12 ± 2	18.2 ± 6.6	11.9 ± 4.4	12.2 ± 4.8	6.7 ± 0.1	4.8 ± 0.2	5.0 ± 0.2	24180 ± 7520	2037 ± 414	2367 ± 797	1277 ± 50	122 ± 25	134 ± 22
	Summer	28 ± 4	8 ± 1	8 ± 1	47 ± 7	17 ± 5	21 ± 4	27 ± 3	7 ± 0	9 ± 0	17.2 ± 1.8	7.1 ± 0.9	7.4 ± 0.7	6.9 ± 0.2	3.6 ± 0.1	3.8 ± 0.2	34193 ± 8121	4017 ± 1879	4083 ± 2137	1320 ± 225	189 ± 47	184 ± 46
	Autumn	32 ± 2	10 ± 2	10 ± 5	51 ± 7	18 ± 2	21 ± 4	32 ± 4	11 ± 3	11 ± 4	16.8 ± 2.2	9.3 ± 1.8	9.1 ± 0.5	7.0 ± 0.5	4.1 ± 0.2	4.3 ± 0.2	20483 ± 3025	4053 ± 594	4517 ± 365	950 ± 205	186 ± 10	190 ± 26
	Winter	36 ± 2	15 ± 2	17 ± 1	54 ± 1	22 ± 1	29 ± 3	38 ± 0	15 ± 1	17 ± 1	17.5 ± 0.8	11.9 ± 0.4	10.8 ± 0.8	7.3 ± 0.1	4.5 ± 0.1	3.9 ± 0.1	23140 ± 488	4210 ± 247	4780 ± 0	980 ± 92	198 ± 5	214 ± 10
2015	Spring	27 ± 2	7 ± 1	7 ± 0	69 ± 11	19 ± 5	23 ± 5	41 ± 2	9 ± 2	10 ± 1	19.5 ± 2.1	7.4 ± 0.4	7.6 ± 1.1	7.5 ± 0.1	4.4 ± 0.3	4.7 ± 0.2	26500 ± 6144	2483 ± 401	3347 ± 740	1090 ± 243	128 ± 46	151 ± 36
	Summer	28 ± 5	7 ± 1	7 ± 1	66 ± 9	21 ± 2	20 ± 2	42 ± 1	6 ± 0	8 ± 1	20.5 ± 8.1	11.4 ± 2.0	9.9 ± 2.5	7.6 ± 0.1	4.6 ± 0.1	4.8 ± 0.1	25667 ± 9452	1607 ± 178	2400 ± 200	916 ± 141	74 ± 10	97 ± 13
	Autumn	29 ± 4	8 ± 4	10 ± 3	54 ± 1	19 ± 3	26 ± 1	50 ± 3	12 ± 4	15 ± 1	21.2 ± 0.7	10.8 ± 1.4	12.9 ± 1.7	7.6 ± 0.2	4.8 ± 0.1	5.2 ± 0.4	20050 ± 1447	2083 ± 193	2427 ± 156	842 ± 30	95 ± 3	127 ± 20
	Winter	35 ± 1	16 ± 3	18 ± 3	57 ± 2	24 ± 1	28 ± 1	56 ± 4	17 ± 0	19 ± 2	20.8 ± 0.8	12.4 ± 0.4	12.9 ± 0.4	7.9 ± 0.1	5.1 ± 0.2	5.1 ± 0.4	18450 ± 707	2340 ± 106	2560 ± 78	812 ± 23	94 ± 1	139 ± 3
2016	Spring	36 ± 3	15 ± 2	15 ± 1	69 ± 4	20 ± 1	21 ± 1	48 ± 5	13 ± 2	14 ± 1	22.3 ± 1.1	13.3 ± 1.5	13.4 ± 1.0	7.6 ± 0.2	4.5 ± 0.2	4.5 ± 0.3	27690 ± 3735	2566 ± 111	3096 ± 178	933 ± 75	101 ± 5	109 ± 2
	Summer	34 ± 3	13 ± 1	14 ± 1	50 ± 3	18 ± 1	19 ± 1	31 ± 3	8 ± 1	8 ± 0	18.9 ± 2.1	9.5 ± 1.5	9.9 ± 1.1	7.9 ± 0.1	4.5 ± 0.2	4.8 ± 0.2	28381 ± 3584	2255 ± 561	3199 ± 288	898 ± 79	93 ± 2	97 ± 3
	Autumn	37 ± 2	14 ± 2	14 ± 1	55 ± 4	21 ± 3	22 ± 4	36 ± 7	11 ± 2	12 ± 1	20.7 ± 2.6	10.1 ± 1.3	10.7 ± 0.5	8.3 ± 0.1	4.7 ± 0.2	5.0 ± 0.3	26480 ± 3968	2271 ± 281	2984 ± 29	1055 ± 81	107 ± 3	105 ± 5
	Winter	39 ± 0	18 ± 1	20 ± 3	58 ± 0	27 ± 1	29 ± 2	53 ± 7	15 ± 1	14 ± 1	21.8 ± 0.8	11.6 ± 0.4	10.7 ± 0.1	8.4 ± 0.0	5.2 ± 0.2	5.4 ± 0.1	28999 ± 165	2670 ± 71	3018 ± 22	1151 ± 122	114 ± 2	113 ± 9
2017	Spring	40 ± 3	17 ± 3	17 ± 2	61 ± 7	20 ± 1	21 ± 1	60 ± 5	11 ± 3	11 ± 3	19.8 ± 2.3	9.5 ± 1.6	10.7 ± 1.4	8.6 ± 0.1	5.1 ± 0.3	5.3 ± 0.2	27451 ± 3073	3151 ± 392	2918 ± 270	974 ± 16	124 ± 18	123 ± 19
	Summer	29 ± 1	11 ± 2	12 ± 2	50 ± 2	18 ± 4	19 ± 3	42 ± 4	8 ± 1	11 ± 1	22.8 ± 2.6	10.8 ± 1.4	11.1 ± 0.3	8.7 ± 0.2	4.6 ± 0.2	4.4 ± 0.2	30896 ± 1902	2290 ± 617	2873 ± 557	1119 ± 123	108 ± 12	109 ± 12

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TABLE 1 (*Continued*) Average composition of the water at the inlet and outlet of the constructed wetland 1 (CW 1) from units A and B (UA and UB) for a hydraulic loading rate of 6 cm d^{-1} (mean ± SD).

Year	Season	BOD_5 [mg L^{-1}]			COD [mg L^{-1}]			TSS [mg L^{-1}]			TN [mg L^{-1}]			TP [mg L^{-1}]			Total coliforms [CFUs 100 mL^{-1}]			*E. coli* [CFUs 100 mL^{-1}]		
		Inlet	Outlet UA	Outlet UB	Inlet	Outlet UA	Outlet UB	Inlet	Outlet UA	Outlet UB	Inlet	Outlet UA	Outlet UB	Inlet	Outlet UA	Outlet UB	Inlet	Outlet UA	Outlet UB	Inlet	Outlet UA	Outlet UB
	Autumn	29 ± 1	12 ± 2	12 ± 2	51 ± 4	16 ± 0	18 ± 4	33 ± 3	15 ± 2	14 ± 4	25.8 ± 1.0	12.7 ± 0.6	11.9 ± 1.4	8.8 ± 0.1	5.3 ± 0.5	5.6 ± 0.7	21970 ± 4201	2677 ± 294	2949 ± 246	1079 ± 169	112 ± 12	114 ± 28
	Winter	31 ± 1	15 ± 1	16 ± 1	53 ± 1	18 ± 1	23 ± 1	44 ± 6	18 ± 1	22 ± 2	25.6 ± 0.7	13.5 ± 0.5	14.5 ± 1.1	8.3 ± 0.4	5.4 ± 0.2	6.1 ± 0.1	18430 ± 1205	2865 ± 54	3120 ± 74	968 ± 49	124 ± 0	141 ± 3
2018	Spring	28 ± 3	10 ± 0	10 ± 1	66 ± 2	20 ± 1	19 ± 1	46 ± 12	14 ± 1	16 ± 2	22.6 ± 2.6	10.4 ± 0.7	11.0 ± 1.2	8.7 ± 0.1	5.2 ± 0.1	5.6 ± 0.4	31296 ± 1228	3411 ± 349	3228 ± 232	1190 ± 183	130 ± 8	139 ± 9
	Summer	29 ± 1	9 ± 1	10 ± 0	53 ± 6	20 ± 2	19 ± 1	33 ± 5	9 ± 2	12 ± 1	20.4 ± 0.7	8.8 ± 1.1	8.3 ± 0.7	9.3 ± 0.1	5.0 ± 0.3	6.1 ± 0.1	27963 ± 3066	3307 ± 548	3065 ± 107	1086 ± 129	104 ± 7	112 ± 14
	Autumn	32 ± 2	11 ± 2	13 ± 2	48 ± 2	17 ± 2	19 ± 4	30 ± 4	13 ± 3	14 ± 3	21.5 ± 0.9	11.3 ± 1.3	9.6 ± 1.0	9.7 ± 0.6	5.5 ± 0.5	6.2 ± 0.3	23113 ± 2268	3220 ± 501	2955 ± 78	933 ± 49	100 ± 5	105 ± 17
	Winter	35 ± 1	16 ± 2	16 ± 1	54 ± 4	22 ± 2	25 ± 0	35 ± 1	17 ± 1	19 ± 1	20.9 ± 0.6	9.8 ± 2.1	10.7 ± 0.6	9.7 ± 0.4	6.4 ± 0.2	6.9 ± 0.3	21340 ± 707	3240 ± 226	3019 ± 21	1024 ± 42	103 ± 2	112 ± 8

Note: Threshold values for Italian Decree 152/2006 governing the reuse of TWW in agricultural irrigation are as follows: BOD_5: 20 mg O_2 L^{-1}; COD: 100 mg O_2 L^{-1}; TSS: 10 mg L^{-1}; TN: 35 mg N L^{-1}; TP: 10 mg P L^{-1}; total coliforms: not considered by the Decree; *Escherichia coli*: 50 CFU 100 mL^{-1} (80% of samples) and 200 CFU 100 mL^{-1} (maximum value point).

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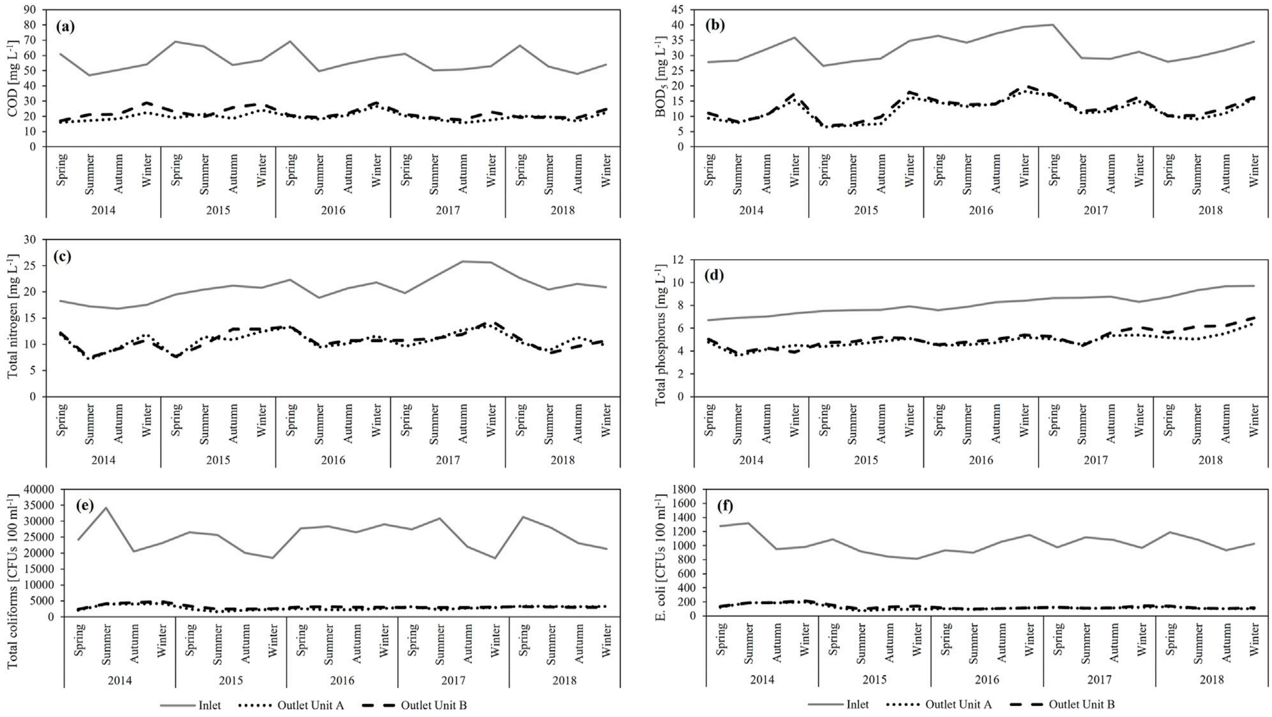

FIGURE 3
Average composition of the water concerning **(a)** chemical oxygen demand (COD), **(b)** biochemical oxygen demand (BOD5), **(c)** total nitrogen, **(d)** total phosphorus, **(e)** total coliforms and **(f)** *E. coli* at the inlet and outlet of the constructed wetland 1 (CW 1) from units A and B (UA and UB) for a hydraulic loading rate of 6 cm d^{-1}, along different seasons and years.

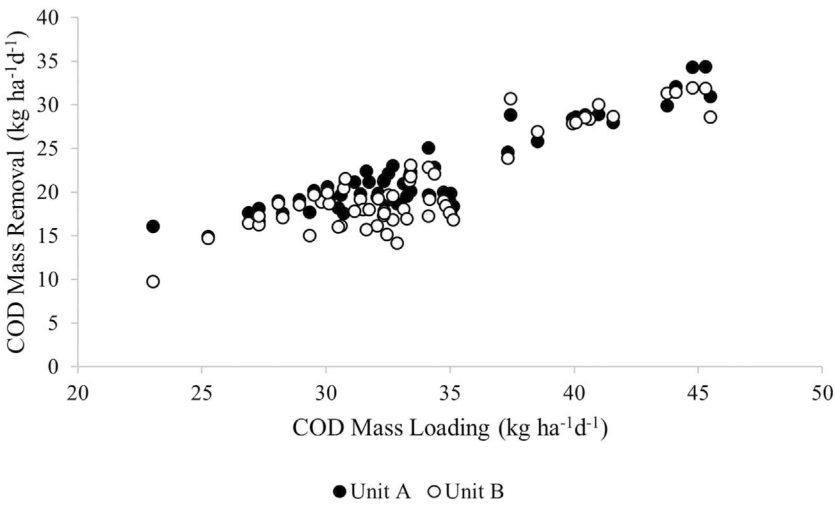

FIGURE 4
Removal of organics during system operation for the constructed wetland 1 (CW 1) from units A and B. COD mass loading vs. COD mass removal.

from 7.1 to 7.9, respectively. Concerning conductivity, the average value at the inlet was 571 ± 33 μS cm^{-1}, with outlet values for units 1 and 2 ranging between 578 and 816 μS cm^{-1}, respectively. Total coliform counts at the inlet of CW units A and B varied between 2×10^3 and 3×10^4 CFU 100 mL^{-1}, while total coliforms counts at the outlet counts varied between 2×10^3 and 9×10^3 CFU 100 mL^{-1}. Removal efficiency for both units was consistent throughout the year, averaging 83% ± 7% and 79% ± 8%, respectively. *E. coli* counts at the inlet and outlet of CW units 1 and 2 varied between 8×10^2 and 2×10^3 CFU 100 mL^{-1} and between 1×10^1 and 4×10^2 CFU 100 mL^{-1}, respectively. Removal efficiency for both units was consistent throughout the year, averaging 85% ± 5% and 83% ±

TABLE 2 Average composition of the water at the inlet and outlet of the constructed wetland 2 (CW 2) from units A and B (UA and UB) for a hydraulic loading rate of 8 cm d^{-1} (mean ± SD).

Year	Season	BOD_5 [mg L^{-1}]			COD [mg L^{-1}]			TSS [mg L^{-1}]			TN [mg L^{-1}]			TP [mg L^{-1}]			Total coliforms [CFUs 100 mL^{-1}]			*E. coli* [CFUs 100 mL^{-1}]		
		Inlet	Outlet UA	Outlet UB	Inlet	Outlet UA	Outlet UB	Inlet	Outlet UA	Outlet UB	Inlet	Outlet UA	Outlet UB	Inlet	Outlet UA	Outlet UB	Inlet	Outlet UA	Outlet UB	Inlet	Outlet UA	Outlet UB
2014	Spring	30 ± 3	10 ± 3	11 ± 3	82 ± 10	15 ± 3	19 ± 6	36 ± 2	11 ± 2	12 ± 1	24.0 ± 2.9	9.6 ± 1.7	11.0 ± 2.2	8.5 ± 0.9	4.8 ± 0.9	5.8 ± 0.6	22536 ± 2892	2942 ± 1408	4927 ± 651	1651 ± 213	131 ± 22	166 ± 10
	Summer	24 ± 2	7 ± 1	7 ± 1	50 ± 15	10 ± 3	12 ± 2	31 ± 3	9 ± 1	10 ± 1	17.7 ± 3.0	8.0 ± 0.3	9.6 ± 0.6	7.8 ± 0.2	4.2 ± 0.1	4.5 ± 0.5	21223 ± 3808	1890 ± 283	2886 ± 547	1125 ± 109	85 ± 11	144 ± 9
	Autumn	24 ± 4	11 ± 4	12 ± 5	40 ± 6	15 ± 4	17 ± 6	29 ± 4	11 ± 3	12 ± 4	18.6 ± 5.9	10.5 ± 2.8	11.3 ± 3.0	7.6 ± 0.7	4.6 ± 0.6	4.8 ± 0.7	26893 ± 6047	3045 ± 814	3947 ± 911	1155 ± 240	138 ± 44	173 ± 27
	Winter	27 ± 0	13 ± 0	16 ± 1	79 ± 23	23 ± 3	25 ± 1	39 ± 5	15 ± 1	17 ± 1	23.4 ± 1.3	11.8 ± 1.1	12.1 ± 1.9	7.1 ± 0.8	4.9 ± 0.2	4.8 ± 0.4	18457 ± 10606	3498 ± 28	5124 ± 314	1257 ± 63	198 ± 20	204 ± 5
2015	Spring	31 ± 3	11 ± 4	13 ± 5	74 ± 11	17 ± 7	20 ± 7	36 ± 5	13 ± 5	15 ± 5	21.1 ± 1.4	9.5 ± 0.9	10.2 ± 0.6	7.9 ± 0.3	4.8 ± 0.3	5.1 ± 0.2	22128 ± 1056	2859 ± 1479	3900 ± 1253	1431 ± 138	185 ± 68	205 ± 64
	Summer	24 ± 3	7 ± 1	7 ± 1	38 ± 5	9 ± 2	10 ± 1	30 ± 3	9 ± 1	10 ± 1	17.3 ± 3.0	8.2 ± 0.7	9.7 ± 0.2	7.4 ± 0.3	4.6 ± 0.1	4.7 ± 0.4	21091 ± 3193	1807 ± 236	2633 ± 277	1063 ± 179	94 ± 10	135 ± 23
	Autumn	24 ± 5	10 ± 3	10 ± 3	40 ± 3	12 ± 2	13 ± 1	33 ± 6	12 ± 3	16 ± 6	19.1 ± 5.2	9.8 ± 1.6	10.3 ± 0.9	8.2 ± 1.0	5.0 ± 0.6	4.9 ± 0.3	29033 ± 6197	4641 ± 2573	5411 ± 2984	1063 ± 267	225 ± 125	282 ± 159
	Winter	29 ± 0	13 ± 1	15 ± 2	39 ± 1	19 ± 4	23 ± 7	43 ± 3	17 ± 1	20 ± 2	23.1 ± 1.0	11.4 ± 0.2	12.4 ± 1.8	7.5 ± 0.8	4.9 ± 0.4	4.6 ± 0.4	22679 ± 8546	5689 ± 778	6213 ± 879	1346 ± 104	367 ± 44	386 ± 10
2016	Spring	42 ± 2	14 ± 3	15 ± 6	70 ± 1	18 ± 6	26 ± 6	66 ± 6	23 ± 2	27 ± 7	23.1 ± 3.3	10.6 ± 1.5	11.1 ± 1.1	8.1 ± 0.7	5.1 ± 0.5	5.3 ± 0.8	30751 ± 1537	4329 ± 208	5017 ± 661	1470 ± 253	201 ± 13	213 ± 12
	Summer	29 ± 3	10 ± 1	11 ± 1	52 ± 8	11 ± 1	18 ± 2	47 ± 5	19 ± 3	19 ± 1	18.1 ± 2.5	6.8 ± 1.2	7.2 ± 1.1	8.0 ± 0.4	4.4 ± 0.1	4.5 ± 0.6	23160 ± 1135	3228 ± 246	3562 ± 302	1245 ± 96	147 ± 28	143 ± 28
	Autumn	30 ± 1	11 ± 1	12 ± 1	43 ± 4	10 ± 2	14 ± 2	66 ± 3	25 ± 5	27 ± 6	19.1 ± 3.1	7.9 ± 2.6	8.1 ± 2.5	7.9 ± 0.4	5.3 ± 0.5	5.0 ± 0.3	31161 ± 2679	4661 ± 1850	6220 ± 1337	1269 ± 72	202 ± 30	195 ± 32
	Winter	30 ± 1	12 ± 2	11 ± 0	43 ± 3	12 ± 0	13 ± 2	70 ± 3	38 ± 6	32 ± 1	23.1 ± 0.6	11.2 ± 0.3	10.4 ± 0.4	7.9 ± 0.1	5.6 ± 0.0	5.3 ± 0.1	31903 ± 1255	6456 ± 235	8018 ± 195	1310 ± 87	245 ± 8	210 ± 6
2017	Spring	31 ± 2	11 ± 1	11 ± 1	60 ± 2	17 ± 3	19 ± 2	64 ± 4	28 ± 4	28 ± 2	19.8 ± 2.3	9.6 ± 0.9	10.6 ± 0.6	7.1 ± 0.2	4.7 ± 0.1	4.8 ± 0.4	26041 ± 2551	4753 ± 946	5299 ± 1065	1021 ± 48	134 ± 10	149 ± 9
	Summer	29 ± 1	8 ± 1	9 ± 1	51 ± 7	11 ± 1	17 ± 1	56 ± 6	20 ± 5	24 ± 4	19.9 ± 4.0	7.5 ± 1.1	8.4 ± 1.3	7.6 ± 0.6	4.3 ± 0.2	4.5 ± 0.4	25370 ± 5079	3336 ± 501	4106 ± 1058	944 ± 51	99 ± 2	106 ± 4
	Autumn	26 ± 4	9 ± 0	10 ± 1	41 ± 3	14 ± 3	18 ± 1	41 ± 5	18 ± 1	20 ± 2	17.0 ± 1.2	6.8 ± 1.1	7.4 ± 0.7	7.7 ± 0.4	4.9 ± 0.6	4.5 ± 0.3	28231 ± 3027	3923 ± 1533	7490 ± 1438	938 ± 66	134 ± 60	179 ± 35

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TABLE 2 (*Continued*) Average composition of the water at the inlet and outlet of the constructed wetland 2 (CW 2) from units A and B (UA and UB) for a hydraulic loading rate of 8 cm d^{-1} (mean ± SD).

Year	Season	BOD_5 [mg L^{-1}]			COD [mg L^{-1}]			TSS [mg L^{-1}]			TN [mg L^{-1}]			TP [mg L^{-1}]			Total coliforms [CFUs 100 mL^{-1}]			*E. coli* [CFUs 100 mL^{-1}]		
		Inlet	Outlet UA	Outlet UB	Inlet	Outlet UA	Outlet UB	Inlet	Outlet UA	Outlet UB	Inlet	Outlet UA	Outlet UB	Inlet	Outlet UA	Outlet UB	Inlet	Outlet UA	Outlet UB	Inlet	Outlet UA	Outlet UB
	Winter	24 ± 1	11 ± 2	10 ± 0	41 ± 2	16 ± 0	17 ± 1	40 ± 2	23 ± 4	25 ± 2	22.4 ± 3.0	11.3 ± 2.4	12.6 ± 3.1	8.2 ± 0.0	5.6 ± 0.3	5.1 ± 0.4	27561 ± 1945	6452 ± 547	8856 ± 126	1219 ± 146	245 ± 30	268 ± 35
2018	Spring	32 ± 2	11 ± 1	12 ± 1	83 ± 3	33 ± 4	38 ± 7	41 ± 5	17 ± 2	19 ± 3	27.1 ± 2.4	10.4 ± 1.2	12.0 ± 1.1	7.9 ± 0.3	5.1 ± 0.4	4.8 ± 0.3	25230 ± 3017	4363 ± 241	5160 ± 505	1342 ± 81	205 ± 45	273 ± 46
	Summer	41 ± 10	10 ± 3	13 ± 2	48 ± 5	14 ± 4	21 ± 7	38 ± 6	10 ± 1	14 ± 2	17.8 ± 1.9	7.3 ± 0.6	6.9 ± 0.3	7.7 ± 0.4	4.6 ± 0.5	5.0 ± 0.4	23903 ± 4212	3828 ± 547	4421 ± 809	1255 ± 244	167 ± 57	159 ± 46
	Autumn	26 ± 4	9 ± 0	10 ± 1	39 ± 3	12 ± 1	12 ± 1	35 ± 3	12 ± 2	15 ± 2	16.9 ± 0.6	7.6 ± 1.1	7.4 ± 1.9	7.8 ± 0.4	4.5 ± 0.3	5.4 ± 0.2	29456 ± 4271	5604 ± 2651	6250 ± 2601	963 ± 64	170 ± 61	177 ± 64
	Winter	28 ± 2	10 ± 1	10 ± 1	68 ± 18	22 ± 7	27 ± 10	30 ± 2	16 ± 2	17 ± 0	21.3 ± 3.6	11.3 ± 1.7	12.4 ± 2.0	7.8 ± 0.2	4.2 ± 0.4	4.6 ± 0.5	27679 ± 2592	8112 ± 322	8981 ± 78	1169 ± 110	226 ± 11	246 ± 20

Note: Threshold values for Italian Decree 152/2006 governing the reuse of TWW, in agricultural irrigation are as follows: BOD_5: 20 mg O_2 L^{-1}; COD: 100 mg O_2 L^{-1}; TSS: 10 mg L^{-1}; TN: 35 mg N L^{-1}; TP: 10 mg P L^{-1}; total coliforms: not considered by the Decree; *Escherichia coli*: 50 CFU 100 mL^{-1} (80% of samples) and 200 CFU 100 mL^{-1} (maximum value point).

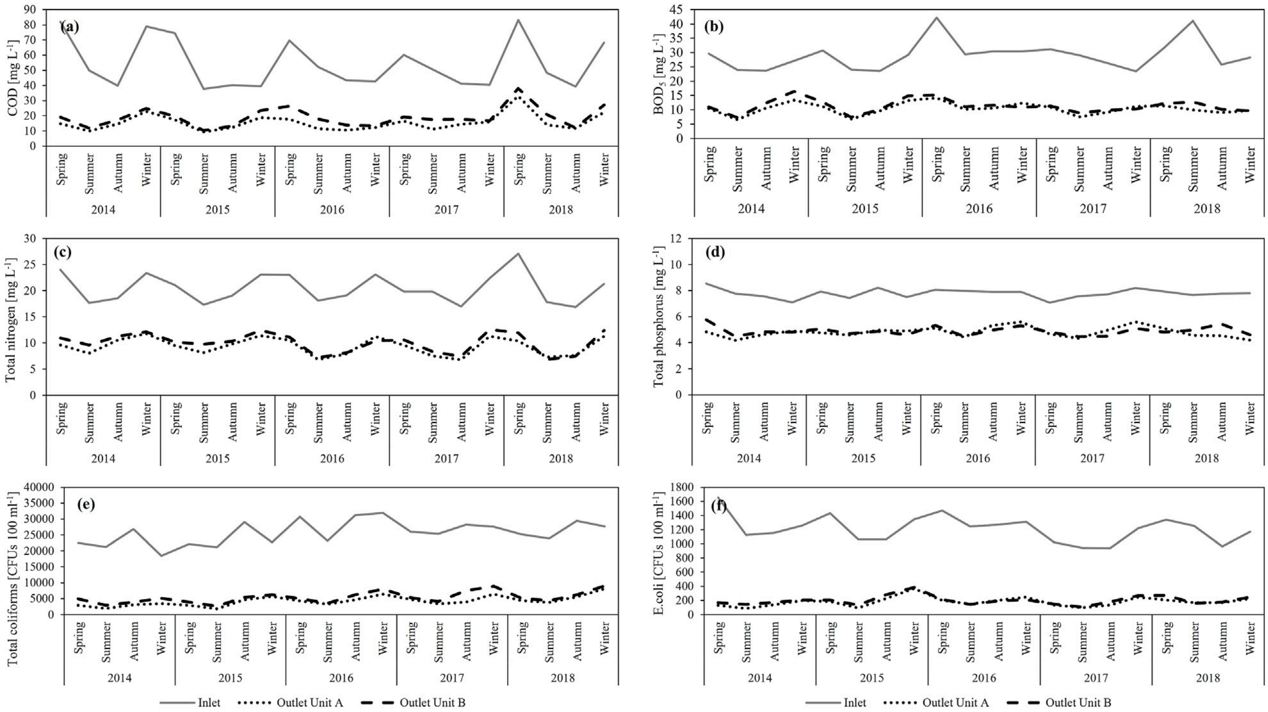

FIGURE 5
Average composition of the water concerning **(a)** chemical oxygen demand (COD), **(b)** biochemical oxygen demand (BOD5), **(c)** total nitrogen, **(d)** total phosphorus, **(e)** total coliforms and **(f)** *E. coli* at the inlet and outlet of the constructed wetland 2 (CW 2) from units A and B (UA and UB) for a hydraulic loading rate of 8 cm d^{-1}, along different seasons and years.

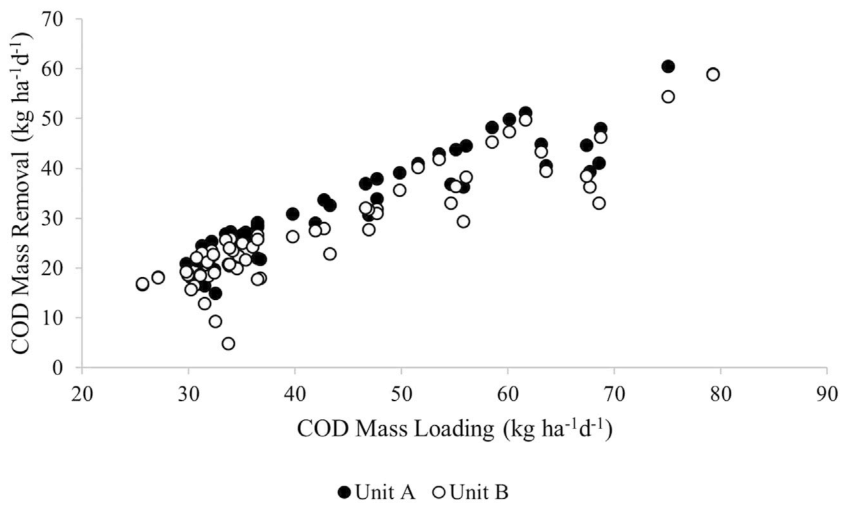

FIGURE 6
Removal of organics during system operation for the constructed wetland 2 (CW 2) from units A and B. COD mass loading vs. COD mass removal.

5%, respectively. Organic mass loadings for units A and B (Figure 6) varied between 26 and 79 kg ha^{-1} d^{-1} for COD, with maximum mass removals of 60 kg COD ha^{-1} d^{-1}.

The efficiency of the CW units operating under different hydraulic conditions (CW 1 and CW 2) was monitored, and a summary of the wastewater characteristics collected from the inlet and outlet of each unit is presented in Table 3. In terms of organic matter removal (COD and BOD_5), both HSSF CWs exhibited similar trends in inlet concentrations and removal efficiencies. Despite differences in set up conditions, including the vegetation used, performance of the two systems followed similar trends. High removal rates of TSS were achieved, reaching up to 88%. Reductions in concentrations of TP (up to 50% from an average inlet concentration of 7.9 ± 0.8 mg L^{-1}) and TN (up to 66%, from an

TABLE 3 Summary of constructed wetland treatment performance for different hydraulic retention times (HRTs) (mean ± SD) (n = 50).

Parameter	HRT (d)	Concentration (mean ± SD) [mg L^{-1}]	Concentration (mean ± SD) [mg L^{-1}]	Removal (mean ± SD) [%]	Concentration (mean ± SD) [mg L^{-1}]	Removal (mean ± SD) [%]
		Inlet	Outlet unit A		Outlet unit B	
COD	8.3	56 ± 8	20 ± 3	65 ± 6	22 ± 4	60 ± 8
[mg L^{-1}]	7.4	54 ± 17	16 ± 6	71 ± 8	19 ± 8	64 ± 12
BOD_5	8.3	32 ± 5	12 ± 4	62 ± 12	13 ± 4	60 ± 9
[mg L^{-1}]	7.4	29 ± 6	10 ± 3	64 ± 8	11 ± 3	61 ± 9
TSS	8.3	41 ± 10	12 ± 4	70 ± 10	13 ± 4	66 ± 10
[mg L^{-1}]	7.4	43 ± 14	17 ± 8	61 ± 9	19 ± 7	57 ± 8
TN	8.3	20.7 ± 3.3	10.7 ± 2.2	48 ± 9	10.7 ± 2.1	48 ± 9
[mg L^{-1}]	7.4	20.5 ± 3.7	9.4 ± 2.0	54 ± 7	10.0 ± 2.2	51 ± 8
TP	8.3	8.1 ± 0.9	4.9 ± 0.6	40 ± 5	5.2 ± 0.8	37 ± 7
[mg L^{-1}]	7.4	7.8 ± 0.6	4.8 ± 0.5	38 ± 6	4.9 ± 0.5	37 ± 5
Total coliforms	8.3	$3 \times 10^4 \pm 6 \times 10^3$	$3 \times 10^3 \pm 8 \times 10^2$	88 ± 4	$3 \times 10^3 \pm 8 \times 10^2$	87 ± 4
[CFU 100 mL^{-1}]	7.4	$3 \times 10^4 \pm 5 \times 10^3$	$4 \times 10^3 \pm 2 \times 10^3$	83 ± 7	$5 \times 10^3 \pm 2 \times 10^3$	79 ± 8
Escherichia coli	8.3	$1 \times 10^3 \pm 2 \times 10^2$	$1 \times 10^2 \pm 4 \times 10^1$	88 ± 4	$1 \times 10^2 \pm 4 \times 10^1$	87 ± 4
[CFU 100 mL^{-1}]	7.4	$1 \times 10^3 \pm 2 \times 10^2$	$2 \times 10^2 \pm 7 \times 10^1$	85 ± 5	$2 \times 10^2 \pm 6 \times 10^1$	83 ± 5

average inlet concentration of 20.6 ± 3.5 mg L^{-1}) were detected. However, these values were lower than those observed for COD and BOD_5 (up to 83%). The removal efficiency of total coliform and *E. coli* was consistent throughout the year and for both CWs.

Observational trends from the monitoring data suggest a stable performance across seasons for both CW 1 and CW 2. The removal efficiencies remained consistent despite some seasonal variations in temperature, rainfall, and evapotranspiration. This suggests the resilience of the system design and the adaptability of the macrophyte species used. Future studies may explore seasonal performance in more detail using statistical modeling to identify potential optimization opportunities.

In CWs, the effectiveness of wastewater treatment is highly dependent on the selection of system design, flow regime, plant species, and substrate composition. The selection of plant species should take into account the location of the CW, the water regime, and wastewater characteristics (Calheiros et al., 2015). The plants selected for the present study were successfully established and are wetland plants (*Cyperus* spp. and *Typha* spp.) typically used in CW systems (Vymazal and Kröpfelová, 2008). *Arundo* sp., for example, demonstrated good capacity to absorb nutrients from highly contaminated tannery water (Calheiros et al., 2012); however, its growth is dominant and vigorous and should be carefully monitored to prevent its spread to the surroundings. The wastewater flowing into both HSSF CWs was subjected to secondary treatment, which reduced pollutant concentrations to levels lower than those typically found in low-strength, untreated wastewater (Metcalf and EddyInc, 2014). In the present study, the organic loads varied between 23 and 79 kg COD ha^{-1} d^{-1}, although higher loadings (17–579 kg COD ha^{-1} d^{-1} and 10–143 kg BOD_5 ha^{-1} d^{-1}) for secondary treatments are unknown (Calheiros et al., 2015). Wastewater at the inlet of both CWs (CW 1 and CW 2) can be considered suitable for simple biological treatments, based on the BOD/COD ratio (Metcalf and EddyInc, 2014). These findings are consistent with those reported in the literature regarding the operation of CW systems (Calheiros et al., 2019). Zurita et al. (2009) reported that the treatment of domestic wastewater in a horizontal subsurface system planted with multiple species (*Strelitzia reginae*, *Anthurium andreanum*, and *Agapanthus africanus*) enhanced BOD, COD, TSS, and TP removal. However, in that study, inlet COD and BOD concentrations were higher than those in the present study, and removal efficiency was also slightly higher. Concerning TN and TP, the inlet concentrations were similar to those reported by Zurita et al. (2009), as were the removal efficiencies, placing the performance within the range reported in the literature. The pH and EC were similar for both CWs, with the outlet water classified as slightly saline. According to Rhoades et al. (1992), this level of salinity can be considered suitable for use in irrigation. The removal rates of total coliform and *E. coli* were consistent with values reported in previous studies. Several authors have reported on the effectiveness of CWs in removing pathogens from wastewater (Stefanakis and Akratos, 2016; Wu et al., 2016; Shingare et al., 2019; Singh et al., 2023). The reduction of bacteria of anthropogenic origin is recognized as a complex process involving physical, chemical, and biological factors, which are influenced to varying degrees by operational parameters such as hydraulic regime, retention time, vegetation, seasonal fluctuations, and water composition (Vymazal and Kröpfelová, 2008; Wu et al., 2016; Rahman et al., 2020).

In our study, the average values of chemical and microbiological parameters at the outflows of CW 1 and CW

TABLE 4 Morphological, productive, and qualitative traits of bermudagrass at EXF 1 affected by the medium-term application of FW and TWW irrigation.

Source of variation	Leaf width [mm]	Shoot density [n cm^{-2}]	Visual quality [1–9 scale]	Above-ground dry biomass [kg ha^{-1}]
Year (Y)				
Y_1	1.50 b	1.97 a	7.21 b	1168.27 b
Y_2	1.51 b	1.98 a	7.24 ab	1231.81 a
Y_3	1.55 a	1.99 a	7.27 a	1250.69 a
Irrigation water (IW)				
IW_1	1.42 c	1.82 c	7.25 a	950.37 c
IW_2	1.56 b	2.05 b	7.23 a	1322.49 b
IW_3	1.58 a	2.07 a	7.25 a	1377.92 a
Interaction Y × IW				
$Y_1 \times IW_1$	1.43 d	1.81 c	7.23 a	921.11 a
$Y_1 \times IW_2$	1.53 c	2.03 b	7.20 a	1264.8 a
$Y_1 \times IW_3$	1.55 bc	2.07 ab	7.21 a	1318.91 a
$Y_2 \times IW_1$	1.41 d	1.82 c	7.24 a	966.46 a
$Y_2 \times IW_2$	1.56 bc	2.09 a	7.23 a	1339.73 a
$Y_2 \times IW_3$	1.56 bc	2.03 b	7.26 a	1389.23 a
$Y_3 \times IW_1$	1.42 d	1.83 c	7.28 a	963.54 a
$Y_3 \times IW_2$	1.57 b	2.03 b	7.25 a	1362.93 a
$Y_3 \times IW_3$	1.65 a	2.10 a	7.28 a	1425.62 a
p-value				
Y	0.000	0.089	0.002	0.000
IW	0.000	0.000	0.142	0.000
Y × IW	0.000	0.000	0.713	0.463

Means are shown. Values followed by the same letters are not statistically different at $p \leq 0.05$, according to Tukey's test. Y_1 = 2014; Y_2 = 2015; Y_3 = 2016; IW_1 = freshwater; IW_2 = treated wastewater with giant reed from unit A; IW_3 = treated wastewater with umbrella sedge from unit B.

2 were not all within the threshold values of the Italian Decree 156/2006, concerning the reuse of TWW for irrigation purposes. In particular, the concentration level of TSS at the outflow of planted units was higher than 10 mg L^{-1} in all seasons, except in summer. A possible solution to improve the TSS removal efficiency would be to enhance the level of WW pretreatment and further reduce the concentration of organic matter or recirculate TWW several times in each CW to obtain higher filtration of TSS. During the test period, the microbiological parameter data obtained for *E. coli* were not always found to be within these legislative limits (50 CFU 100 mL^{-1} in 80% of the samples and 200 CFU 100 mL^{-1} as maximum value point). However, a high *E. coli* removal efficiency was observed in both CW 1 and CW 2 (80%–85%). A combined HSSF–VSSF system could be useful to remove *E. coli* with higher efficiency, as demonstrated in other Mediterranean areas (Abidi et al., 2009; Avila et al., 2013). The different retention times in the two systems would provide changes in the aerobic and anaerobic conditions of the substrate, producing higher pathogenic removal rates.

3.2 Effect of TWW irrigation application on bermudagrass characteristics

The main morphological, productive, and qualitative traits of bermudagrass at experimental fields EXF 1 and EXF 2, as influenced by the medium-term irrigation with FW and TWW, are shown in Tables 4, 5.

At EXF 1, the year had a significant effect on leaf width, dry above-ground weight, and visual turf quality, while irrigation water significantly affected all traits, except visual turf quality. The year-by-irrigation water interactions were significant for leaf width and shoot density (Table 4). In the case of EXF 2, ANOVA showed that the year had significant effects on all traits, except shoot density, while irrigation water significantly influenced all parameters in the study. The year-by-irrigation water interactions were significant for shoot density and visual turf quality performance (Table 5).

Regarding morphological traits, both experimental fields showed greater leaf width and shoot density in the TWW-irrigated plots than in the FW-irrigated plots. For EXF 1, leaf width measured an average of 1.52 cm, ranging from 1.42 (FW

TABLE 5 Morphological, productive, and qualitative traits of bermudagrass at EXF 2 affected by the medium-term application of FW and TWW irrigation.

Source of variation	Leaf width [mm]	Shoot density [n cm^{-2}]	Visual quality [1–9 scale]	Above-ground dry biomass [kg ha^{-1}]
Year (Y)				
Y_1	1.58 b	1.93 a	6.33 b	1265.36 ab
Y_2	1.57 b	1.93 a	6.38 a	1289.58 a
Y_3	1.64 a	1.93 a	6.38 a	1241.23 b
Irrigation water (IW)				
IW_1	1.56 b	1.86 b	6.36 b	1061.60 c
IW_2	1.60 a	1.98 a	6.39 a	1338.09 b
IW_3	1.63 a	1.97 a	6.35 b	1396.47 a
Interaction Y × IW				
$Y_1 \times IW_1$	1.56 a	1.85 c	6.33 bc	1048.92 a
$Y_1 \times IW_2$	1.60 a	1.96 ab	6.38 abc	1354.64 a
$Y_1 \times IW_3$	1.59 a	1.98 ab	6.29 c	1392.54 a
$Y_2 \times IW_1$	1.53 a	1.86 c	6.36 abc	1118.45 a
$Y_2 \times IW_2$	1.58 a	2.00 a	6.43 a	1342.47 a
$Y_2 \times IW_3$	1.59 a	1.94 b	6.36 abc	1407.82 a
$Y_3 \times IW_1$	1.59 a	1.86 c	6.38 ab	1017.44 a
$Y_3 \times IW_2$	1.63 a	1.96 ab	6.37 abc	1317.18 a
$Y_3 \times IW_3$	1.70 a	1.98 ab	6.40 ab	1389.07 a
p-value				
Y	0.000	0.940	0.003	0.012
IW	0.001	0.000	0.015	0.000
Y × IW	0.158	0.009	0.035	0.106

Means are shown. Values followed by the same letters are not statistically different at $p \leq 0.05$, according to Tukey's test. Y_1 = 2014; Y_2 = 2015; Y_3 = 2016; IW_1 = fresh water; IW_2 = treated wastewater with umbrella sedge from unit A; IW_3 = treated wastewater reedmace from unit B.

irrigation treatment) to 1.58 cm (TWW_2 irrigation treatment). The highest average shoot density (2.07 n cm^{-2}) was found in the TWW_2-irrigated plot, while the lowest (1.82 n cm^{-2}) was found in the FW-irrigated plot. However, at EXF 2, no significant differences in leaf width or shoot density were found between the TWW irrigation treatments. It is worth noting that, for leaf width, the highest average values were found in the last year across both experimental fields; no significant differences were observed between the first and second years. Based on the main results, the various irrigation treatments significantly affected morphological traits, and it can be concluded that the higher nutrient content in TWW had a substantial impact on increasing the leaf width and shoot density of bermudagrass. This evidence is supported by the findings of Castro et al. (2011), who highlighted that the application of TWW irrigation compared to freshwater irrigation significantly affects turfgrass growth. Miller and Dickens (1996) and Beard (1973) reported that macronutrients such as N, P, and K can strongly influence turfgrass growth, improving drought hardiness, recuperative potential, stomatal physiological mechanisms, and synthesis of carbohydrates, all processes correlated with growth. Furthermore, other nutrients also influence turfgrass growth through their roles as constituents of cell walls (e.g., Ca) or components of chlorophyll (e.g., Mg), as well documented by Turgeon (2004). The fact that no significant differences in shoot density were found across years in either experimental field can be attributed to the implementation of similar fertilization management programs across plots, which ensured a constant and adequate supply of nutrients.

The above-ground biomass yield has similar trends across the two experimental fields for both year and irrigation water treatments. At EXF 1, the plot irrigated with TWW_2 yielded the highest average above-ground biomass values (1377.92 kg ha^{-1}), with a difference of 427.55 kg ha^{-1}, compared to the FW-irrigated plot. The plots yielded the best values during the second and third years. In the case of EXF 2, taking into consideration the effects of irrigation water treatments, above-ground biomass ranged from 1061.60 kg ha^{-1} (FW treatment) to 1396.47 kg ha^{-1} (TWW_2 treatment). Although ANOVA indicated significant differences across the years, observed values were relatively similar. Although agronomic management practices were identical in both

TABLE 6 Chemical characteristics of soil at EXF 1, as influenced by the application of FW and TWW irrigation.

Source of variation	pH	EC [μS cm^{-1}]	TOC [g kg^{-1}]	TKN [g kg^{-1}]	P [ppm]	Na [ppm]
Year (Y)						
Y_1	7.66 a	191.33 b	7.79 b	1.28 c	31.70 b	93.66 b
Y_2	7.64 b	200.54 a	7.83 b	1.30 b	31.48 b	92.87 b
Y_3	7.65 a	202.44 a	7.97 a	1.35 a	32.40 a	97.25 a
Irrigation water (IW)						
IW_1	7.65 a	190.78 c	7.73 b	1.27 c	31.47 b	90.83 b
IW_2	7.65 a	200.33 b	7.91 a	1.32 b	32.48 a	96.67 a
IW_3	7.65 a	203.19 a	7.95 a	1.34 a	31.63 b	96.29 a
Interaction Y × IW						
$Y_1 \times IW_1$	7.66 ab	184.43 a	7.71 e	1.25 e	31.32 cd	91.15 e
$Y_1 \times IW_2$	7.66 abc	194.22 a	7.72 de	1.27 de	32.14 ab	94.52 cd
$Y_1 \times IW_3$	7.67 a	195.33 a	7.93 c	1.31 cd	31.63 bc	95.30 c
$Y_2 \times IW_1$	7.64 c	193.11 a	7.70 e	1.24 e	31.30 cd	90.95 e
$Y_2 \times IW_2$	7.64 c	203.33 a	7.93 bc	1.32 bc	32.59 a	92.55 de
$Y_2 \times IW_3$	7.64 abc	205.18 a	7.85 cd	1.33 bc	30.54 d	95.11 cd
$Y_3 \times IW_1$	7.66 abc	194.79 a	7.77 de	1.31 c	31.79 bc	90.37 e
$Y_3 \times IW_2$	7.66 abc	203.45 a	8.09 a	1.36 ab	32.70 a	102.93 a
$Y_3 \times IW_3$	7.64 bc	209.07 a	8.06 ab	1.38 a	32.71 a	98.44 b
p-value						
Y	0.000	0.000	0.000	0.000	0.000	0.000
IW	0.683	0.000	0.000	0.000	0.000	0.000
Y × IW	0.022	0.289	0.000	0.015	0.000	0.000

Means are shown. Values followed by the same letters are not statistically different at $p \leq 0.05$, according to Tukey's test. Y_1 = 2014; Y_2 = 2015; Y_3 = 2016; IW_1 = fresh water; IW_2 = treated wastewater with giant reed from unit A; IW_3 = treated wastewater with umbrella sedge from unit B. EC, electrical conductivity; TOC, total organic carbon; TKN, total nitrogen; P, assimilable phosphorus; Na, sodium content.

experimental fields, it can be concluded that climatic conditions, such as air temperature and rainfall, and physiological processes, such as evapotranspiration, greatly affected the biomass yield of bermudagrass both in TWW- and FW-irrigated plots. In particular, during the second year of tests, higher temperatures, increased ET rates, and lower rainfall levels in both experimental fields strongly increased the demand for irrigation water, directly contributing to an enhanced biomass production. On the contrary, the lowest biomass yield was recorded in the year with lower temperatures, reduced evapotranspiration, and higher rainfall levels. Focusing on the effect of irrigation water on biomass production, it is worth noting that, in all harvests, the highest yields were obtained in TWW-irrigated plots. This indicates that TWW irrigation has a direct impact on biomass production, due to a higher content of macronutrients in comparison to FW irrigation (Licata et al., 2016). The increased accumulation of nutrients in the soil increases plant growth and biomass yields. This consideration is consistent with the findings of Castro et al. (2011), Ganjegunte et al. (2017), and Zalacáin et al. (2019), who studied the impact of TWW irrigation on various turfgrass species. However, there is no clear consensus among researchers regarding the usefulness of TWW irrigation in the short and long term. Evanylo et al. (2010) stated that depending on Na or heavy metal content in TWW, this practice could produce negative effects on biomass production and, in general, on plant growth. However, bermudagrass is known to tolerate a wide range of salt concentrations in soil and water, although large variability exists between salt-tolerant and salt-sensitive bermudagrass genotypes, as well-documented by Van Tran et al. (2019).

Regarding qualitative parameters, visual turf quality ratings were similar in both experimental fields, with minimal effects of year and irrigation water. It is interesting to highlight that, at EXF 1, irrigation water had no significant effect on visual turfgrass quality, while at EXF 2, significant differences visual turfgrass quality were found among the irrigation water treatments. In urban areas, aesthetical performance — typically assessed through leaf color and visual appearance — is a key qualitative parameter for turfgrass. However, visual turfgrass quality should also be related to biomass production as the quality of turfgrass is

TABLE 7 Chemical characteristics of soil at EXF 2, as influenced by the application of FW and TWW irrigation.

Source of variation	pH	EC [μS cm^{-1}]	TOC [g kg^{-1}]	TKN [g kg^{-1}]	P [ppm]	Na [ppm]
Year (Y)						
Y_1	7.90 a	552.92 b	7.34 a	1.72 a	18.35 b	89.95 b
Y_2	7.89 a	549.46 c	7.33 a	1.74 a	18.45 ab	90.42 ab
Y_3	7.89 a	555.34 a	7.32 a	1.74 a	18.60 a	90.64 a
Irrigation water (IW)						
IW_1	7.87 c	278.32 c	7.14 c	1.69 b	18.39 b	84.22 b
IW_2	7.92 a	705.92 a	7.44 a	1.76 a	18.42 ab	93.33 a
IW_3	7.90 b	673.48 b	7.40 b	1.75 a	18.60 a	93.46 a
Interaction Y × IW						
$Y_1 \times IW_1$	7.88 a	278.33 fg	7.13 c	1.68 a	18.29 a	83.83 a
$Y_1 \times IW_2$	7.91 a	706.23 b	7.45 a	1.74 a	18.36 a	92.82 a
$Y_1 \times IW_3$	7.91 a	674.21 d	7.44 ab	1.73 a	18.39 a	93.18 a
$Y_2 \times IW_1$	7.87 a	276.37 g	7.14 c	1.69 a	18.35 a	84.17 a
$Y_2 \times IW_2$	7.92 a	702.07 c	7.45 a	1.76 a	18.42 a	93.54 a
$Y_2 \times IW_3$	7.90 a	669.93 e	7.38 b	1.76 a	18.60 a	93.57 a
$Y_3 \times IW_1$	7.86 a	280.27 f	7.16 c	1.69 a	18.52 a	84.65 a
$Y_3 \times IW_2$	7.93 a	709.45 a	7.42 ab	1.77 a	18.47 a	93.64 a
$Y_3 \times IW_3$	7.89 a	676.30 d	7.39 b	1.76 a	18.82 a	93.63 a
p-value						
Y	0.513	0.000	0.137	0.070	0.012	0.007
IW	0.000	0.000	0.000	0.000	0.023	0.000
Y × IW	0.417	0.026	0.005	0.789	0.515	0.788

Means are shown. Values followed by the same letters are not statistically different at $p \leq 0.05$, according to Tukey's test. Y_1 = 2014; Y_2 = 2015; Y_3 = 2016; IW_1 = fresh water; IW_2 = treated wastewater with umbrella sedge from unit A; IW_3 = treated wastewater with reedmace from unit B. EC, electrical conductivity; TOC, total organic carbon; TKN, total nitrogen; P, assimilable phosphorus; Na, sodium content.

strongly dependent on plant growth. Marin et al. (2022) reported that biomass is positively correlated with the normalized difference vegetation index (NDVI), while Bell et al. (2002) stated that the NDVI is highly correlated with visual turfgrass quality. Based on this, we are able to conclude that sustainable practices that maintain turfgrass at a high-quality level may also lead to increased biomass production and thus a higher frequency of mowings per year, potentially generating higher management costs. In the present study, during the 3 years of testing, both a general increase in biomass yields and improved visual quality of bermudagrass were observed in the TWW-irrigated plots, fully confirming the findings reported in the literature. This indicates that the application of TWW irrigation in the medium term enables good maintenance of bermudagrass and provides other benefits, such as water resource savings and nutrients, compared to conventional practices. However, in the long term, considering that TWW is also a source of pathogens, health risks linked to this practice due to possible microbiological contamination of the turfgrass also need to be evaluated, as reported by Bihadassen et al. (2020).

3.3 Effect of TWW irrigation applications on soil characteristics

The effects of FW and TWW irrigation on the chemical characteristics of the soil at EXF 1 and EXF 2 are shown in Tables 6, 7.

At EXF 1, year and irrigation water had a significant effect on the main soil characteristics, except for pH. The year-by-irrigation water interactions were significant for all soil parameters, except for EC (Table 6). At experimental field EXF 2, year affected EC, TP, and the Na content, while irrigation water had significant effects on all soil parameters, except for pH. The results of ANOVA revealed that the year-by-irrigation water interactions were significant for only EC and TOC (Table 7).

Focusing on soil pH, in both experimental fields, no significant variations were observed between FW- and TWW-irrigated soils during the 3-year study. In particular, soil pH in the TWW-irrigated plots did not vary significantly during the study period, and similar values were found between soils irrigated with FW and TWW. Our findings are consistent with those of previous studies that have assessed the impact of TWW irrigation on soil pH in both the short and long term. In particular, Castro et al. (2011) studied the effect of 2 years of TWW irrigation on soil properties and turfgrass growth, reporting negligible variations in soil pH. Rusan et al. (2007), when evaluating the use of wastewater for the long-term irrigation of forage crops and subsequent effects on soil and plant quality parameters, found inconsistent variations in soil pH. The limited impact of TWW irrigation on soil pH could be due to the buffering action of the soil; this is the capacity of soil to maintain a relatively stable pH despite the presence of acidifying or alkalizing factors, as well documented by Curtin and Trolove (2013). This concept is consistent with the findings of other authors who tried to explain this phenomenon when applying TWW in different periods of time.

In the case of soil salinity, significant variations in EC were found across the various irrigation treatments and in both experimental fields over the entire study period. The 3-year application of TWW irrigation significantly increased topsoil salinity, despite differences in the clay content in the two experimental fields. Soil salinity was on average 198.10 μS cm^{-1} (EXF 1) and 552.57 μS cm^{-1} (EXF 2) during the 3-year tests. At EXF 1, the highest EC value (203.19 μS cm^{-1}) was found in soils irrigated with TWW from the umbrella sedge-planted unit, on average. At EXF 2, the highest EC value (705.92 μS cm^{-1}) was found in soils irrigated with TWW from the reedmace-planted unit, on average. The fact that the highest accumulation of total dissolved salts was detected in TWW-irrigated soil was probably due to the physical characteristics of the soils in EXF 1 and EXF 2. In fact, differences in the clay content and the quantity of soil aggregates undoubtedly influenced the relative cation exchange capacity of the soils in both experimental fields. These considerations could explain the different EC values found between sites when considering factors such as year and irrigation water. However, other aspects may help explain this finding, such as the salt concentration in TWW, climatic conditions, and agronomic soil management practices. In our study, it is worth noting that the accumulation of salts increased over time in both experimental fields. As well-explained by Rusan et al. (2007), we can thus conclude that the longer the application of TWW irrigation, the greater the increase in soil EC. This indicates that more salt could be accumulated in the topsoil in the long-term period. Therefore, a range of agronomic solutions are needed to facilitate the leaching processes, as sustained by Libutti and Monteleone (2012).

The highest TOC content at EXF 1 was found in the TWW-irrigated soils, with values ranging from 7.91 to 7.95 g kg^{-1}. It is worth noting that the organic matter content of the soil increased with the duration of TWW irrigation, attributable to higher nutrient and organic load in TWW compared to FW. On the contrary, at EXF 2, despite the best performance detected in TWW-irrigated soil, no significant variations in TOC were observed over the years. Our findings were consistent with the results of other studies as many authors (Rusan et al., 2007; Licata et al., 2017; Poustie et al., 2020) have agreed that the accumulation of organic compounds following TWW irrigation depends on two main factors: the initial concentration of organic compounds in the treated wastewater and the duration of its application. However, it is also crucial to take into account soil characteristics and soil texture in particular. In our study, the different distribution of soil texture fractions, in terms of clay, silt, and sand content, substantially affected the change in soil TOC across the years, with differences found between the two experimental fields. Nevertheless, it is worth noting that the application of TWW irrigation enriches the soil with organic matter, improving physical and chemical properties and fertility, in general.

TWW irrigation produced significant increases in N and P contents compared to FW irrigation in both experimental fields. However, in the case of TKN, substantial variations in soil N content during the 3-year period were only observed at EXF 1. EXF 2 was found to have the highest TKN, ranging from 1.69 g kg^{-1} (FW-irrigated soil) to 1.76 kg^{-1} (TWW-irrigated soil), compared to EXF 1. In contrast, over the 3-year period, the highest average TP value (31.86 ppm) was found at EXF 1, with the TWW-irrigated soil exhibiting the best performance (32.48 ppm). Although the highest level of N and P in TWW can explain their highest accumulation in the TWW-irrigated soil, it is also needed to consider the duration of irrigation and plant and microbial activities in terms of nutrient uptake and transformation (Dotaniya and Meena, 2015; Adomako et al., 2022).

The application of TWW irrigation increased the soil Na content in both experimental fields, highlighting differences between the various irrigation treatments. The highest Na content was detected in TWW-irrigated soil and increased by extending the duration of TWW irrigation. It is well-established (Wakeel, 2013) that Na contributes to the deterioration of the soil structure, and excess Na content can have an adverse effect on the plant growth and soil health (Callaghan et al., 2017; Eimers et al., 2015). As a consequence, monitoring the soil over time is fundamental, especially when TWW irrigation is applied to clay soil and over the medium and long term. This concept was confirmed by Qadir et al. (2003), who studied Na removal from a calcareous saline–sodic soil through leaching and plant uptake. Furthermore, the exploitation of some agronomic practices such as the periodic application of good-quality irrigation water in soil seems necessary to avoid any risk to soil structure in the long term, leach the excess salt, and maintain a suitable sodium absorption ratio (SAR). In our study, it is worth noting that, in the 3-year period, average SAR values (data not shown) were found below the values that negatively influence soil properties (SAR > 10), according to the threshold values for Italian Decree 152/2006 governing the reuse of TWW in agricultural irrigation.

When considering the various year-by-irrigation water interactions, differences were found between EXF 1 and EXF 2. More specifically, in both experimental fields, interactions between the main factors produced evident variations in the EC and TOC contents, with the highest values found in 2018 in TWW-irrigated soil, indicating an accumulation of salts and organic compounds over time.

4 Conclusion

In this long-term study, two horizontal subsurface flow constructed wetland plants were compared to assess their pollutant treatment performance over a 5-year period. They systems were integrated with appropriate pretreatment systems and planted with different macrophytes, thus establishing monoculture systems. The two constructed wetland plants showed similar performance in terms of pollutant removal, achieving effluents of satisfactory quality. Vegetation had a significant effect on the overall treatment performance, highlighting the crucial role of plant species selection in achieving improved treatment outcomes. Seasonal variations in the main chemical and microbiological parameters were found at the outlet of the planted units, likely due to the effect of climatic conditions on plant growth and phenology. This highlights that in constructed wetlands, a monoculture system is not equally effective in pollutant treatment across seasons. The 3-year application of treated wastewater irrigation led to significant differences in the morphological and productive characteristics of bermudagrass, due to increased nutrient accumulation in the soil. However, it did not affect the qualitative characteristics of the bermudagrass plants. In the two experimental plants, all chemical soil parameters were affected over time, except for soil pH. This indicates that soil fertility can be improved by the medium-term application of treated wastewater irrigation, particularly with respect to soil organic matter and mineral content. These results confirm the benefits of using treated wastewater for irrigation in urban areas while also highlighting the need for regular monitoring of its effects on soil and vegetation.

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Author contributions

DF: Software, Methodology, Investigation, Data curation, Conceptualization, Writing – review and editing, Formal Analysis, Writing – original draft. ML: Formal Analysis, Project administration, Supervision, Validation, Methodology, Writing – review and editing, Visualization, Writing – original draft, Resources, Conceptualization. CL: Writing – original draft, Project administration, Resources, Funding acquisition. GU: Software, Writing – original draft, Data curation. FS: Data curation, Software, Writing – original draft. CS: Validation, Conceptualization, Investigation, Supervision, Writing – review and editing, Formal Analysis, Visualization, Software, Methodology, Writing – original draft, Data curation.

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The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2025.1606056/full#supplementary-material

References

Abidi, S., Kallali, H., Jedidi, N., Bouzaiane, O., and Hassen, A. (2009). Comparative pilot study of the performances of two constructed wetland wastewater treatment hybrid systems. *Desalination* 246, 370–377. doi:10.1016/j.desal.2008.03.061

Addo-Bankas, O., Zhao, Y., Wei, T., and Stefanakis, A. (2024). From past to present: tracing the evolution of treatment wetlands and prospects ahead. *J. Water Process Eng.* 60 (105151), 105151–15. doi:10.1016/j.jwpe.2024.105151

Adomako, M. O., Roiloa, S., and Yu, F.-H. (2022). Potential roles of soil microorganisms in regulating the effect of soil nutrient heterogeneity on plant performance. *Microorganisms* 10 (12), 2399–17. doi:10.3390/microorganisms10122399

Agaton, C. B., and Guila, P. M. C. (2023). Ecosystem services valuation of constructed wetland as a nature-based solution to wastewater treatment. *Earth* 4 (1), 78–92. doi:10.3390/earth4010006

Ahmed, A. M., and Kareem, S. L. (2024). Evaluating the role of hydraulic retention time (HRT) in pollutant removal efficiency using *Arundo donax* in vertical subsurface flow constructed wetlands. *Bioremediation J.*, 1–15. doi:10.1080/10889868.2024.2439829

Aktatos, C. S., and Tsihrintzis, V. A. (2007). Effect of temperature, HRT, vegetation and porous media on removal efficiency of pilot-scale horizontal subsurface flow constructed wetlands. *Ecol. Eng.* 29 (2), 173–191. doi:10.1016/j.ecoleng.2006.06.013

American Public Health Association (1998). Standard Methods for the Examination of Water and Wastewater, twentieth ed., Washington, DC: APHA: APAT

Apat, C. N. R.-IRSA (2003). *Metodi Analitici per le Acque*. 1st edn. Rome, Italy: APAT, 115–696.

Avellan, C. T., Ardakanian, R., and Gremillion, P. (2007). The role of constructed wetlands for biomass production within the water-soil-waste nexus. *Water Sci. Technol.* 75 (10), 2237–2245. doi:10.2166/wst.2017.106

Ávila, C., Reyes, C., Bayona, J. M., and García, J. (2013). Emerging organic contaminant removal depending on primary treatment and operational strategy in horizontal subsurface flow constructed wetlands: influence of redox. *Water Res.* 47 (1), 315–325. doi:10.1016/j.watres.2012.10.005

Barbera, A. C., Cirelli, G. L., Cavallaro, V., Di Silvestro, I., Pacifici, P., Castiglione, V., et al. (2009). Growth and biomass production of different plant species in two different constructed wetland systems in Sicily. *Desalination* 246, 129–136. doi:10.1016/j.desal.2008.03.046

Beard, J. B. (1973). "Warm season turfgrasses," in *Turfgrass: science and culture*. Editor J. B. Beard (Englewood Cliffs, NJ: Prentice Hall-Inc.), 133–165.

Beebe, D. A., Castle, J. W., Molz, F. J., and Rodgers, Jr., J. H. (2014). Effects of evapotranspiration on treatment performance in constructed wetlands: experimental studies and modeling. *Ecol. Eng.* 71, 394–400. doi:10.1016/j.ecoleng.2014.07.052

Bell, G. E., Martin, D. L., Wiese, S. G., Dobson, D. D., Smith, M. W., Stone, M. L., et al. (2002). Vehicle-mounted optical sensing: an objective means for evaluating turf quality. *Crop Sci.* 42, 197–201. doi:10.2135/cropsci2002.1970

Bialowiec, A., Albuquerque, A., and Randerson, P. F. (2014). The influence of evapotranspiration on vertical flow subsurface constructed wetland performance. *Ecol. Eng.* 67, 89–94. doi:10.1016/j.ecoleng.2014.03.032

Bihadassen, B., Hassi, M., Hamadi, F., Aitalla, A., Bourouache, M., El Boulani, A., et al. (2020). Irrigation of a golf course with UV-treated wastewater: effects on soil and turfgrass bacteriological quality. *Appl. Water Sci.* 10 (7), 7–10. doi:10.1007/s13201-019-1095-5

Calheiros, C. S. C., Bessa, V. S., Mesquita, B. R., Brix, H., Rangel, A. O. S. S., and Castro, P. M. L. (2015). Constructed wetland with a polyculture of ornamental plants for wastewater treatment at a rural tourism facility. *Ecol. Eng.* 79, 1–7. doi:10.1016/j.ecoleng.2015.03.001

Calheiros, C. S. C., Castro, P. M. L., Gavina, A., and Pereira, R. (2019). Toxicity abatement of wastewaters from tourism units by constructed wetlands. *Water* 11, 2623. doi:10.3390/w11122623

Calheiros, C. S. C., Quitério, P. V. B., Silva, G., Crispim, L. F. C., Brix, H., Moura, S. C., et al. (2012). Use of constructed wetland systems with *Arundo* and *Sarcocornia* for polishing high salinity tannery wastewater. *J. Environ. Manage.* 95 (1), 66–71. doi:10.1016/j.jenvman.2011.10.003

Callaghan, M. V., Head, F. A., Cey, E. E., and Bentley, L. R. (2017). Salt leaching in fine-grained, macroporous soil: negative effects of excessive matrix saturation. *Agr. Water manage.* 181, 73–84. doi:10.1016/j.agwat.2016.11.025

Carrillo, V., Casas-Ledòn, Y., Neumann, P., and Vidal, G. (2023). Environmental performance of constructed wetland planted with monocultures and polycultures for wastewater treatment. *Ecol. Eng.* 193, 107015. doi:10.1016/j.ecoleng.2023.107015

Castro, E., Mañas, M. P., and De Las Heras, J. (2011). Effects of wastewater irrigation on soil properties and turfgrass growth. *Water Sci. Technol.* 263 (8), 1678–1688. doi:10.2166/wst.2011.335

Cirelli, G. L., Consoli, S., Di Grande, V., Milani, M., and Toscano, A. (2006). "Subsurface constructed wetlands for wastewater treatment and reuse in agriculture. Five years of experiences in Sicily, Italy," in Proceedings of the 10th International Conference on Wetland Systems for Water Pollution Control, Lisbon, Portugal, 23–29 September 2006, 375–383.

Croce, P., De Luca, A., Mocioni, M., Volterrani, M., and Beard, J. B. (2002). Warm-season turfgrass species and cultivar characterization for a Mediterranean climate. *Int. Turfgrass Soc. Res. J.* 9, 855–859.

Curtin, D., and Trolove, S. (2013). Predicting pH buffering capacity of New Zealand soils from organic matter content and mineral characteristics. *Soil Res.* 51 (6), 494–502. doi:10.1071/SR13137

de Campos, S. X., and Soto, M. (2024). The use of constructed wetlands to treat effluents for water reuse. *Environments* 11 (35), 35–26. doi:10.3390/environments11020035

Dotaniya, M. L., and Meena, V. D. (2015). Rhizosphere effect on nutrient availability in soil and its uptake by plants: a review. *Proc. Natl. Acad. Sci. India, Sect. B Biol. Sci.* 85 (1), 1–12. doi:10.1007/s40011-013-0297-0

Eimers, M. C., Croucher, K. N., Raney, S. M., and Morris, M. L. (2015). Sodium accumulation in calcareous roadside soils. *Urban Ecosyst.* 18, 1213–1225. doi:10.1007/s11252-015-0454-8

Evanylo, G., Ervin, E., and Zhang, X. (2010). Reclaimed water for turfgrass irrigation. *Water* 2, 685–701. doi:10.3390/w2030685

Franci Gonçalves, R., de Oliveira Vaz, L., Peres, M., and Sarnaglia Merlo, S. (2021). Microbiological risk from non-potable reuse of greywater treated by anaerobic filters associated to vertical constructed wetlands. *J. Water Process Eng.* 39, 101751. doi:10.1016/j.jwpe.2020.101751

Ganjegunte, G., Ulery, A., Niu, G., and Wu, Y. (2017). Effects of treated municipal wastewater irrigation on soil properties, switchgrass biomass production and quality under arid climate. *Ind. Crop. Prod.* 99, 60–69. doi:10.1016/j.indcrop.2017.01.038

Garfi, M., Pedescoll, A., Bécares, E., Hijosa-Valsero, M., Sidrach-Cardona, R., and García, J. (2012). Effect of climatic conditions, season and wastewater quality on contaminant removal efficiency of two experimental constructed wetlands in different regions of Spain. *Sci. Total Environ.* 437, 61–67. doi:10.1016/j.scitotenv.2012.07.087

Gebru, S. B., and Werkneh, A. A. (2024). Applications of constructed wetlands in removing emerging micropollutants from wastewater: occurrence, public health concerns, and removal performances – a review. *S. Afr. J. Chem. Eng.* 48, 395–416. doi:10.1016/j.sajce.2024.03.004

Gomez, K. A., and Gomez, A. A. (1984). *Statistical procedures for agricultural research*. New York, NY, USA: Wiley and Sons.

Hajjar, T., Mothar, R. H., Jaoude, L. A., and Yanni, S. F. (2025). Treated wastewater reuse for irrigation in a semi-arid region. *Sci. Total Environ.* 966 (178579), 1–12. doi:10.1016/j.scitotenv.2025.178579

Hashem, M. S., and Qi, X. (2021). Treated wastewater irrigation—a review. *Water* 13 (11), 1527–1537. doi:10.3390/w13111527

Hassan, I., Chowdhury, S. R., Prihartato, P. K., and Razzak, S. A. (2021). Wastewater treatment using constructed wetland: current trends and future potential. *Processes* 9 (1917), 1917–1924. doi:10.3390/pr9111917

Headley, T. R., Davison, L., Huett, D. O., and Müller, R. (2012). Evapotranspiration from subsurface horizontal flow wetlands planted with Phragmites australis in sub-tropical Australia. *Water Res.* 46, 345–354. doi:10.1016/j.watres.2011.10.042

Huang, J., Cai, W., Zhong, Q., and Wang, S. (2013). Influence of temperature on micro-environment, plant eco-physiology and nitrogen removal effect in subsurface flow constructed wetland. *Ecol. Eng.*, 60, 242–248. doi:10.1016/j.ecoleng.2013.07.023

Italian Ministry of Agricultural Policies (Ministero delle Politiche Agricole) (1999). Metodi ufficiali di analisi chimica del suolo. Decree 13/09/1999.

Ji, Z., Tang, W., and Pei, Y. (2022). Constructed wetland substrates: a review on development, function mechanisms, and application in contaminants removal. *Chemosphere* 286 (131564), 131564–15. doi:10.1016/j.chemosphere.2021.131564

Kadlec, R. H., Knight, R. L., Vymazal, J., Brix, H., Cooper, P., and Haberl, R. (2000). *Constructed wetlands for pollution control: processes, performance, design and operation*. 1st edition. London, UK: IWA Publishing, 17–90.

Kottek, M., Grieser, J., Beck, C., Rudolf, B., and Rubel, F. (2006). World Map of the Köppen-Geiger climate classification updated. *Meteorol. Z* 15, 259–263. doi:10.1127/0941-2948/2006/0130

Kulshreshtha, N. M., Verma, V., Soti, A., Brighu, U., and Gupta, A. B. (2022). Exploring the contribution of plant species in the performance of constructed wetlands for domestic wastewater treatment. *Bioresour. Technol. Rep.* 18 (101038), 101038–12. doi:10.1016/j.biteb.2022.101038

Kushwaha, A., Goswami, L., Kim, B. S., Lee, S. S., Pandey, S. K., and Kim, K. H. (2024). Constructed wetlands for the removal of organic micropollutants from wastewater: current status, progress, and challenges. *Chemosphere* 360 (142364), 142364–21. doi:10.1016/j.chemosphere.2024.142364

Leto, C., Sarno, M., Tuttolomondo, T., La Bella, S., and Licata, M. (2008). Two years of studies into native bermudagrass (*Cynodon* spp.) germplasm from Sicily (Italy) for the constitution of turf cultivars. *Acta Hort.* 783, 39–48. doi:10.17660/ActaHortic.2008.783.3

Liang, M., Zhang, C., Peng, C., Lai, Z., Chen, D., and Chen, Z. (2011). Plant growth, community structure, and nutrient removal in monoculture and mixed constructed wetlands. *Ecol. Eng.* 37, 309–316. doi:10.1016/j.ecoleng.2010.11.018

Libutti, A., and Monteleone, M. (2012). Irrigation management in Mediterranean salt affected agriculture: how leaching operates. *Ital. J. Agron.* 7 (1), e5–e35. doi:10.4081/ija.2012.e5

Licata, M., Gennaro, M. C., Tuttolomondo, T., Leto, C., and La Bella, S. (2019). Research focusing on plant performance in constructed wetlands and agronomic application of treated wastewater – a set of experimental studies in Sicily (Italy). *Plos One* 14 (7), e0219445. doi:10.1371/journal.pone.0219445

Licata, M., La Bella, S., Leto, C., Virga, G., Leone, R., Bonsangue, G., et al. (2016). Reuse of urban-treated wastewater from a pilot-scale horizontal subsurface flow system in Sicily (Italy) for irrigation of bermudagrass (*Cynodon dactylon* (L.) Pers.) turf under Mediterranean climatic conditions. *Desalin. Water Treat.* 57, 23343–23364. doi:10.1080/19443994.2016.1180479

Licata, M., Tuttolomondo, T., Leto, C., La Bella, S., and Virga, G. (2017). The use of constructed wetlands for the treatment and reuse of urban wastewater for the irrigation of two warm-season turfgrass species under Mediterranean climatic conditions. *Water Sci. Technol.* 76 (2), 459–470. doi:10.2166/wst.2017.221

Magni, S., Gaetani, M., Caturegli, L., Leto, C., Tuttolomondo, T., La Bella, S., et al. (2014). Phenotypic traits and establishment speed of 44 turf Bermudagrass accessions. *Acta Agric. Scand. Sect. B* 64 (8), 722–733. doi:10.1080/09064710.2014.955524

Marín, J., Yousfi, S., Mauri, P. V., Parra, L., Lloret, J., and Masaguer, A. (2020). RGB vegetation indices, NDVI, and biomass as indicators to evaluate C_3 and C_4 turfgrass under different water conditions. *Sustainability* 12 (6), 1–16. doi:10.3390/su12062160

Marín-Muñiz, J. L., Hernández, M. E., Gallegos-Pérez, M. P., and Amaya-Tejeda, S. I. (2020). Plant growth and pollutant removal from wastewater in domiciliary constructed wetland microcosms with monoculture and polyculture of tropical ornamental plants. *Ecol. Eng.* 147 (105658), 1–9. doi:10.1016/j.ecoleng.2019.105658

Masi, F., Rizzo, A., and Regelsberger, M. (2017). The role of constructed wetlands in a new circular economy, resource oriented, and ecosystem services paradigm. *J. Environ. Manage.* 216, 275–284. doi:10.1016/j.jenvman.2017.11.086

Metcalf and Eddy, Inc. Tchobanoglous, G., David Stensel, H., Tsuchihashi, R., and Burton, F. L. (2014). *Wastewater engineering treatment and resource recovery*. 5th edn. New York, NY, USA: McGraw-Hill.

Miller, G., and Dickens, R. (1996). Potassium fertilization related to cold resistance in bermudagrass. *Crop Sci.* 36, 1290–1295. doi:10.2135/cropsci1996.0011183x003600050036x

Mittal, Y., Tabish Noori, Md., Saeed, T., and Yadav, A. K. (2023). Influence of evapotranspiration on wastewater treatment and electricity generation performance of constructed wetland integrated microbial fuel cell. *J. Water Process Eng.* 53, 103580. doi:10.1016/j.jwpe.2023.103580

Muscarella, S. M., Alduina, R., Badalucco, L., Capri, F. C., Di Leto, Y., Gallo, G., et al. (2024). Water reuse of treated domestic wastewater in agriculture: effects on tomato plants, soil nutrient availability and microbial community structure. *Sci. Total Environ.* 928 (172259), 172259–10. doi:10.1016/j.scitotenv.2024.172259

Nelson, D. W., and Sommers, L. E. (1998). Total nitrogen analysis of soil and plant tissues. *J. Assoc. Off. Anal. Chem.* 63, 770–778. doi:10.1093/jaoac/63.4.770

Nelson, D. W., and Sommers, L. E. (1996). "Methods of soil analysis, Part 2," in *Agronomy, Am. Soc. Of agron*. Editor A. L. Page, 2nd ed. (Madison, Wisconsin: Inc.), 961–1010.

Ofori, S., Puškáčová, A., Růžičková, I., and Wanner, J. (2021). Treated wastewater reuse for irrigation: pros and cons. *Sci. Total Environ.* 760, 144026. doi:10.1016/j.scitotenv.2020.144026

Pansu, M., and Gautheyrou, J. (2006). in *Handbook of soil analysis*. Editors M. Pansu, and J. Gautheyrou 604 (Springer), 593.

Pereira, S. I. A., Castro, P. M. L., and Calheiros, C. S. C. (2022). "Biomass production and energetic valorization in constructed wetlands (chapter 7)," in *Bioenergy crops: a sustainable means of phytoremediation*. Editor T. Jos 1st edn (Boca Raton, FL: CRC Press, Taylor and Francis Group). doi:10.1201/9781003043522

Poustie, A., Yang, Y., Verburg, P., Pagilla, K., and Hanigan, D. (2020). Reclaimed wastewater as a viable water sourcefor agricultural irrigation: a review of food crop growth inhibition and promotion in the context of environmental change. *Sci. Total Environ.* 739, 139756. doi:10.1016/j.scitotenv.2020.139756

Qadir, M., Steffens, D., Yan, F., and Schubert, S. (2003). Sodium removal from a calcareous saline–sodic soil through leaching and plant uptake during phytoremediation. *Land Degrad. Dev.* 14 (3), 301–307. doi:10.1002/ldr.558

Rahman, M. E., Bin Halmi, M. I. E., Bin Abd Samad, M. Y., Uddin, M. K., Mahmud, K., Abd Shukor, M. Y., et al. (2020). Design, operation and optimization of constructed wetland for removal of pollutant. *Int. J. Environ. Res. Public Health* 17, 8339. doi:10.3390/ijerph17228339

Rhoades, J., Kandiah, A., and Mashali, A. (1992). "The use of saline waters for crop production," in *Food and agriculture organization irrigation and drainage paper No 48* (Rome, Italy: Food and Agriculture Organization), 133.

Rodriguez-Dominguez, M. A., Biller, P., Carvalho, P. N., Brix, H., and Arias, C. A. (2021). Potential use of plant biomass from treatment wetland systems for producing biofuels through a biocrude green-biorefining platform. *Energies* 14 (8157), 8157–17. doi:10.3390/en14238157

Rusan, M. J. M., Hinnawi, S., and Rousan, L. (2007). Long-term effect of wastewater irrigation of forage crops on soil and plant quality parameters. *Desalination* 215 (1–3), 143–152. doi:10.1016/j.desal.2006.10.032

Shingare, R. P., Thawale, P. R., Raghunathan, K., Mishra, A., and Kumar, S. (2019). Constructed wetland for wastewater reuse: role and efficiency in removing enteric pathogens. *J. Environ. Manage.* 246, 444–461. doi:10.1016/j.jenvman.2019.05.157

Shtull-Trauring, E., Cohen, A., Ben-Hur, M., Israeli, M., and Bernstein, M. (2022). NPK in treated wastewater irrigation: Regional scale indices to minimize environmental pollution and optimize crop nutritional supply. *Sci. Total Environ.* 806 (150387), 150387–15. doi:10.1016/j.scitotenv.2021.150387

SIAS (2024). Sicilian agro-meteorological information service. Available online at: www.sias.regione.sicilia.it (accessed on September 20, 2024).

Singh, S., Upadhyay, S., Rani, A., Sharma, P. K., Rawat, J. M., Rawat, B., et al. (2023). Assessment of pathogen removal efficiency of vertical flow constructed wetland treating septage. *Sci. Rep.* 13, 18703. doi:10.1038/s41598-023-45257-2

Stefanakis, A. (2019). The role of constructed wetlands as green infrastructure for sustainable urban water management. *Sustainability* 11 (6981), 6981–19. doi:10.3390/su11246981

Stefanakis, A., and Akratos, C. S. (2016). in *Removal of pathogenic bacteria in constructed wetlands: mechanisms and efficiency*. Editors A. Ansari, S. Gill, R. Gill, G. Lanza, and L. Newman (Cham, Switzerland: Springer International Publishing), 327–346.

Takavakoglou, V., Pana, E., and Skalkos, D. (2022). Constructed wetlands as nature-based solutions in the post-COVID agri-food supply chain: challenges and opportunities. *Sustainability* 14 (6), 3145. doi:10.3390/su14063145

Toscano, A., Marzo, A., Milani, M., Cirelli, G. L., and Barbagallo, S. (2015). Comparison of removal efficiencies in Mediterranean pilot constructed wetlands vegetated with different plant species. *Ecol. Eng.* 75, 155–160. doi:10.1016/j.ecoleng.2014.12.005

Turgeon, A. J. (2004). "Turfgrass species," in *Turfgrass management*. Editor A. J. Turgeon (Upper Saddle Rive, NJ: Pearson Prentice Hall), 59–119.

Tuttolomondo, T., Leto, C., La Bella, S., Leone, R., Virga, G., and Licata, M. (2016). Water balance and pollutant removal efficiency when considering evapotranspiration in a pilot-scale horizontal subsurface flow constructed wetland in Western Sicily (Italy). *Ecol. Eng.* 87, 295–304. doi:10.1016/j.ecoleng.2015.11.036

United States Department of Agriculture, Natural Resources Conservation Service (1999). "Soil taxonomy. Basic system of soil classification for making and interpreting soil surveys," in *Soil survey staff*. Second Edition, 1–871.

Van Tran, T., Fukai, S., Giles, H. E., and Lambrides, C. J. (2019). Salinity tolerance among a large range of bermudagrasses (*Cynodon* spp.) relative to other halophytic and non-halophytic perennial C_4 grasses. *Environ* 145, 121–129. doi:10.1016/j.envexpbot.2017.10.011

Vymazal, J. (2011). Plants used in constructed wetlands with horizontal subsurface flow: a review. *Hydrobiol.* 674 (1), 133–156. doi:10.1007/s10750-011-0738-9

Vymazal, J. (2014). Constructed wetlands for treatment of industrial wastewaters: a review. *Ecol. Eng.* 73, 724–751. doi:10.1016/j.ecoleng.2014.09.034

Vymazal, J., and Kröpfelová, L. (2008). "Wastewater treatment in constructed wetlands with horizontal sub-surface flow," in *Environmental pollution*. Editors B. J. Alloway, and J. T. Trevors (Berlin/Heidelberg, Germany: Springer).

Wakeel, A. (2013). Potassium–sodium interactions in soil and plant under saline-sodic conditions. *J. Plant Nutr. Soil Sc.* 176 (3), 344–354. doi:10.1002/jpln.201200417

Wang, H., Xu, Y., and Chai, B. (2023). Effect of temperature on microorganisms and nitrogen removal in a multi-stage surface flow constructed wetland. *Water* 15 (7), 1256. doi:10.3390/w15071256

Wang, Y., Li, Q., Zhang, W., Wang, S., and Peng, H. (2021). Pollutants removal efficiency assessment of constructed subsurface flow wetlands in lakes with numerical models. *J. Hydrol.* 598 (126289), 126289–14. doi:10.1016/j.jhydrol.2021.126289

Wu, H., Zhang, J., Ngo, H. H., Guo, W., Hu, Z., Liang, S., et al. (2015). A review on the sustainability of constructed wetlands for wastewater treatment: design and operation. *Bioresour. Technol.* 175, 594–601. doi:10.1016/j.biortech.2014.10.068

Wu, S., Carvalho, P. N., Müller, J. A., Manoj, V. R., and Dong, R. (2016). Sanitation in constructed wetlands: a review on the removal of human pathogens and fecal indicators. *Sci. Total Environ.* 541, 8–22. doi:10.1016/j.scitotenv.2015.09.047

Zalacáin, D., Martínez-Pérez, S., Bienes, R., García-Díaz, A., and Sastre-Merlín, A. (2019). Turfgrass biomass production and nutrient balance of an urban park. *Chemosphere* 237 (124481), 1–7. doi:10.1016/j.chemosphere.2019.124481

Zhang, C. B., Liu, W.-L., Wang, J., Ge, Y., Gu, B. H., and Chang, J. (2012). Effects of plant diversity and hydraulic retention time on pollutant removals in vertical flow constructed wetland mesocosms. *Ecol. Eng.* 49, 244–248. doi:10.1016/j.ecoleng.2012.08.010

Zhang, Z., Rengel, Z., and Meney, K. (2007). Growth and resource allocation of *Canna indica* and *Schoenoplectus validus* as affected by interspecific competition and nutrient availability. *Hydrobiol* 589, 235–248. doi:10.1007/s10750-007-0733-3

Zhu, H., Zhou, Q. W., Yan, B. X., Liang, Y. X., Yu, X. F., Gerchman, Y., et al. (2018). Influence of vegetation type and temperature on the performance of constructed wetlands for nutrient removal. *Water Sci. Technol.* 77 (3), 829–837. doi:10.2166/wst.2017.556

Zurita, F., De Anda, J., and Belmont, M. A. (2009). Treatment of domestic wastewater and production of commercial flowers in vertical and horizontal subsurface-flow constructed wetlands. *Ecol. Eng.* 35 (5), 861–869. doi:10.1016/j.ecoleng.2008.12.026