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Intensification of Constructed Wetlands for Nutrient Removal From Agricultural Runoff

Stevo Lavrnić[1] | Francesco Chioggia[1] | Salvatore Luca Gentile[2] | Giuseppe Mancuso[1] | Peyman Arjomandi Akram[1] | Francesco Cavazza[2] | Attilio Toscano[1]

[1]Department of Agricultural and Food Sciences, Alma Mater Studiorum - University of Bologna, Bologna, Italy | [2]Consorzio di Bonifica Canale Emiliano Romagnolo, Bologna, Italy

Correspondence: Giuseppe Mancuso (g.mancuso@unibo.it)

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ABSTRACT

Inspired by nature-based solutions, constructed wetlands are increasingly acknowledged as effective wastewater treatment tools, with diverse structural designs suited to various contexts. In the present work, two typologies of these systems, a full-scale farm wetland and three pilot scale systems, were tested in terms of nitrogen and phosphorus removal from agricultural runoff, as well as their potential to maintain treatment performance under intensified conditions and reduced space requirements. The farm wetland, a full-scale surface flow wetland operating for more than 20 years on an agricultural farm in Italy, was monitored for different years. It showed high water retention rates (> 60% in most monitored years and up to 100% in some years) and high nutrient retention efficiencies, with TN retention generally exceeding 65% and TP retention exceeding 46%. However, the size of this system might be an obstacle to its wider application. With that goal, three pilot systems were tested, namely surface flow (SFCW) and horizontal sub-surface flow (HFCW) constructed wetlands. The HFCWs were incorporated with two distinct compositions: one packed with gravel (HFCW-G) and another containing 10% v/v biochar in addition to gravel (HFCW-B). All CWs were monitored for 1 year, during which applied flow and nutrients concentration were studied as variables influencing their performance. The results showed that, overall, the SFCW performed the best, granting average TN reduction up to 78%, significantly higher in comparison to HFCW-B and HFCW-G (33% and 34%, respectively). In terms of TP removal, on average, the HFCW systems presented similarly high performances (43%) compared to SFCW system (33%). Seasonal fluctuations in nitrogen removal reflected a temperature and seasonal influence on the performance of the systems, particularly in the system containing biochar (HFCW-B). This study therefore confirms that comparable or higher areal nutrient removal rates can be achieved under intensified operating conditions (e.g., increased hydraulic and nutrient loading rates), suggesting that farm wetland design might consider reduced footprint while maintaining treatment efficiency.

1 | Introduction

Agricultural runoff, consisting of both agricultural drainage water and surface runoff collected within the agricultural and crop cultivation context, is a waste stream polluted by the substances used for fertilisation and control of pests and hazards. Such substances were found to pose a risk to the recipient environment, with hazards of poisoning, eutrophication and equilibria alteration (Braschi et al. 2022). Nitrogen (N) and phosphorus (P) are the two major macronutrients present in agricultural runoff that require removal (Kasak et al. 2020). Fundamental to crop productivity and health, they are regularly distributed, typically in the form of mineral fertilisers, and are susceptible to washout leading to their accumulation in water bodies and subsequent pollution (Lang et al. 2013). Agricultural runoff therefore should be intercepted and treated before reaching the ecosystems (Heyde et al. 2025). The European Union has consequently promoted several initiatives and frameworks to tackle this issue such as the Nitrates Directive, Water Framework Directive, Common Agricultural Policy and the EU Water Initiative (EUWI) (European Court of Auditors 2014; Pirrera and Pluchino 2017).

Accordingly, the scientific community has focused on the development of treatment technologies that are compatible with recipient ecosystems, technologically applicable and economically viable. The nature-based solutions (NbSs) are systems where natural mechanisms and dynamics are mimicked and applied in controlled environments to process and decontaminate resources such as water and soil (Larsen et al. 2015; van Hullebusch et al. 2021).

One of the main representatives of NbSs is constructed wetlands (CWs), where selected plants and associated microbiomes are incorporated together with specific substrates for treating water streams that derive from different sources. Such low-maintenance treatment units can improve water quality while integrating well into the existing ecosystems and providing secondary benefits such as flood protection, biodiversity promotion and soil stabilisation (Vymazal 2017; Anawar et al. 2019). CWs typically employ low-maintenance plant species, to minimise the economic and resource impact of the installation, reducing the need for irrigation, fertilisation and tilling (Kadlec and Wallace 2009). There are several types of CWs, each specifically designed to address distinct environmental challenges associated with the treatment and removal of pollutants from various types of wastewaters (Santos et al. 2024).

Different CW types operate under different principles of flow and filtration, with varying levels of efficiency in removing specific contaminants (Kushwaha et al. 2024). Recreating natural marshes, SFCWs are systems where water is collected in open areas and kept in contact with the atmosphere, thus favouring gaseous exchanges and oxygenation (Vymazal 2010). This aspect favours the presence of both aerobic and anaerobic zones, fundamental for physiochemical processes such as ammonia nitrification and denitrification (Vymazal 2017). Given the greater free volume in comparison to other CWs, SFCWs are usually characterised by longer hydraulic retention times (HRTs). Moreover, they can be useful solutions for climatic events mitigation, such as floods (Stefanakis 2019). Horizontal Flow CWs (HFCWs) convey the collected water through a substrate that occupies the majority of their volume. HFCWs have therefore shorter HRTs, but the interaction with the substrate itself is more extended in comparison to SFCWs (Anawar et al. 2019). In order to achieve high efficiency in wastewater treatment activity, the substrate choice is fundamental and different solutions are possible (Li et al. 2011). Among the active substrates, biochar is a material that has gained the research interest for contaminants removal, especially nutrients due to its high adsorption capacity. It consists of a material derived from the pyrolysis of organic matter, with structure and chemical properties that can be potentially useful in the removal of water pollutants, if applied in certain concentration ranges (Maleki Shahraki and Mao 2022). It has, in fact, presented the potential to remove nutrients from wastewater and to subsequently be potentially applicable as a soil amendment (Altland and Locke 2013).

CWs certainly represent an effective and low-cost nature-based solution for treating agricultural runoff from one farm and those systems are often called farm wetlands. They provide significant removal of nutrients (Xiong et al. 2022), but also of suspended solids and pesticides via different mechanisms (Nan et al. 2023; Gabr 2025) and its biomass has a potential to be used as a soil amendment (Chioggia et al. 2024) or as a feedstock for anaerobic digestion to produce biogas (Mancuso, Duran Lugo et al. 2026; Mancuso, Bolognini, et al. 2026). For agricultural runoff specifically, farm wetlands based on SFCW or HFCW systems are emphasised in the literature due to their proven effectiveness in managing common pollutants found in agricultural runoff (Tan et al. 2025). Indeed, selecting a specific type of farm wetland will depend on local climate conditions, space availability, the specific contaminant load and the long-term sustainability of the treatment process (Dell'Osbel et al. 2020).

However, performance of farm wetlands is highly temperature-dependent: in colder seasons, lower microbial activity, plant dormancy and ice formation impede key processes like nitrification, denitrification and adsorption (Li et al. 2024), leading to reduced pollutant removal right when runoff production might be the most intensive. Another constraint to their wider application is their substantial land requirements that are typically 1%–8% of catchment area to reach 50% of nitrogen removal (Kadlec and Wallace 2009; Nan et al. 2023). This can make application of farm wetlands impractical and farmers often resist their adoption due to, among other reasons, economic losses that can be caused by fertile land dedication to a non-productive activity (Soldo et al. 2022).

Consequently, to increase the use of farm wetlands, it is important to intensify and optimise their design, to both reduce required footprint and maintain removal efficiency, especially in winter conditions. The objectives of this study were therefore to: (1) evaluate the nutrient removal performance of a mature full-scale farm wetland treating agricultural runoff; (2) investigate, through pilot-scale experiments, alternative configurations including biochar-enhanced systems to inform the design optimisation and intensification of farm wetlands and facilitate their larger-scale application; (3) assess whether the full-scale system could be intensified and therefore its size reduced.

2 | Materials and Methods

2.1 | Study Area

The study area was a 12.5 ha experimental farm in the vicinity of Budrio village, Emilia-Romagna region, Italy. The site is characterised by a subtropical humid climate with a mean annual temperature of c.a. 13.7°C and 771 mm of mean annual rainfall, mostly occurring during spring and autumn, as confirmed by 30-year normal values of the Regional Agency for Environmental Protection (ARPAE). All the surface runoff from the farm area is collected in a single ditch through a surface drainage collection system. The farm hosts a full-scale farm wetland and different pilot CW systems (Figure 1).

The previous studies by Lavrnić et al. (Lavrnić, Nan, et al. 2020) and Buscaroli et al. (Buscaroli et al. 2024) showed

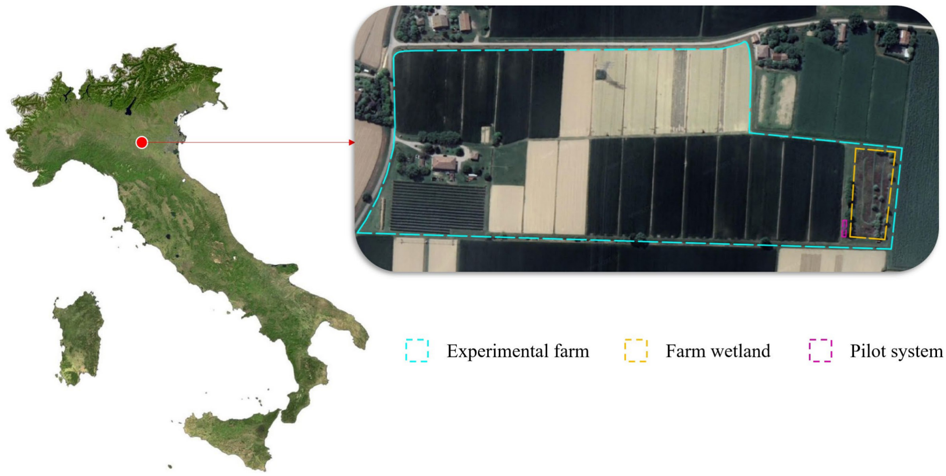

FIGURE 1 | The experimental farm and the test units (i.e., farm wetland and pilot plant).

TABLE 1 | Farm wetland and pilot systems trials.

Parameter	Farm wetland	Pilot systems
Monitoring period	2018–2023	2022–2023
Hydraulic loading rate ($cm\,day^{-1}$)	0.8–4.5	3.2–12.8
TN loading rate ($g\,m^{-2}\,day^{-1}$)	0.0–0.5	1.0–3.7
TP loading rate ($g\,m^{-2}\,day^{-1}$)	0.00–0.01	0.07–0.48

that the farm wetland can reach high removal efficiencies in terms of different contaminants, although its substrate can be considered partly clogged (Lavrnić, Alagna, et al. 2020). On the other hand, the doubts remain whether its size can be considered appropriate for the amount of agricultural runoff produced within a 12.5 ha farm. In other words, if the system could treat the same amount of water with a smaller surface, it would contribute to an optimised design of farm wetlands in the future. Since the inflow to the full-scale system depends on the amount of precipitation and therefore it cannot be regulated, the pilot systems were used to this purpose. They were studied for their capability for nutrients removal and recovery under high stress conditions (Table 1), in order to better understand the potential of the farm wetlands.

2.2 | Full-Scale Farm Wetland

A full-scale farm wetland with a SFCW configuration (Figure 2) was installed in 2000 to treat the surface runoff (Lavrnić et al. 2018). The system occupies an area of approximately 0.4 ha and it consists of four meanders that increase the water path and contribute to the treatment efficiency. The system has a maximum water level of 0.4 m and the outflow, being gravitational, is conditioned by reaching that depth. If the water level stays below 0.4 m, no outflow occurs and the inflow water can be considered retained by the system.

The full-scale farm wetland was monitored between 2018 and 2023, for the whole 6 years. The inlet and outlet water volumes were constantly measured with the flow-metres and the samples were periodically taken through automatic samplers and analysed for nutrient content (Lavrnić et al. 2018). They were stored within a refrigerated unit of the samplers and analysed within 24 h.

A factor called water retention rate was calculated for every year of monitoring as a difference between inlet and outlet yearly volume indicating therefore the part of the inlet volume that was absorbed by the system (e.g., vegetation consumption, infiltration, evaporation) and never exited it. It is a good indicator of the potential of the farm wetland to manage incoming volumes, reducing flood risk and it was calculated by means of Equation (1):

$$Water\ retention\ (\%) = \frac{V_{in} - V_{out}}{V_{in}} \cdot 100\,\% \tag{1}$$

where V_{in} ($m^3\ year^{-1}$) is the yearly water inlet, whereas V_{out} ($m^3\ year^{-1}$) is the yearly water outlet from the system.

2.3 | Pilot-Scale Systems

The use of both full-scale and pilot-scale systems allowed for a complementary assessment of performance. The full-scale farm wetland provided insight into long-term behaviour under real operating conditions (e.g., hydrological variability). In contrast, the pilot-scale systems enabled controlled experiment under higher loading conditions, allowing the evaluation of system intensification.

The pilot plant (Figure 2) was built within the framework of the H2020 WATERAGRI project at the same site where the full-scale farm wetland is located. A particular interest in the study was placed on biochar, an innovative recycled substrate whose potential for nutrient removal was assessed.

The pilot plant consists of:

- 2 m^2 HFCW filled with gravel up to 90 cm with a void volume of 0.72 m^3 (HFCW-G);

FIGURE 2 | The studied full-scale farm wetland (left) and pilot systems (right).

TABLE 2 | Sampling periods of the pilot plant with respective details.

Sampling period	Date	Average air temperature T (°C)	Flow rate ($L\,d^{-1}$)	Number of samples per system
Startup	Sep 2022–Jan 2023	10.7	128	9
Spring	March 2023	12.1	64	4
Early Summer	Jun 2023–Aug 2023	25.7	192	9
Late Summer	Oct 2023	22.5	192	4
Early Autumn	Nov 2023	8.2	320	7
Late Autumn	Dec 2023	3.6	320	4

- $2\,m^2$ HFCW filled with gravel and 10% v/v of biochar (c.a. 37 kg), up to 90 cm with a void volume of $0.72\,m^3$ (HFCW-B);
- $2\,m^2$ SFCW with a gravel layer of 30 cm with a void volume of $1.38\,m^3$;

The biochar applied in the HFCW-B systems was provided by CharLine (Riedlingsdorf, Austria). It consisted of pyrolyzed untreated wood chips, with a final dry surface area of $292\,m^2\,g^{-1}$ and a bulk density of $218\,kg\,m^{-3}$. No treatment was applied to the biochar prior to the experiments. All the systems were planted with *Phragmites australis*, a predominant species in the full-scale farm wetland.

The pilot scale systems were fed with a synthetic working solution. Although this approach does not consider different pollutants normally present in agricultural runoff (e.g., solids, pesticides), it ensured controlled and reproducible conditions. The solution was applied throughout the experiment by filling a 500 L tank with normal irrigation water and by adding HNO_3 and H_3PO_4 in proportions relative to the desired final concentration. In the beginning, the concentrations were chosen to reflect the levels found in the agricultural runoff of the same experimental farm area (Lavrnić, Nan, et al. 2020), but they were deliberately increased to put systems under a higher stress, representative of peak runoff events, in line with the objective of exploring possible system intensification. For the tests, the working solution was mixed with irrigation water through a fertigation pump and the irrigation water flow was modulated in function of the target inlet flow rate. Each pilot-scale system was supplied through four drippers with a nominal flow rate of $2\,L\,h^{-1}$. A pump conveyed water from the tank to the drippers and operated for a predetermined period to deliver the desired water volume to each system. The outlet flow, on the other hand, was regulated by gravity, whereas a constant water level in each system was maintained by the outlet pipe.

The SFCW, HFCW-G and HFCW-B were tested during five different periods (Table 2), characterised by different flow rates and air temperatures. The flow rates were chosen based on the normal flow rates to the full-scale farm wetland, but also to enable a more detailed testing of treatment performance in different seasons. Therefore, the flow rates and corresponding loading rates, both in terms of water quality and quantity, simulated high stress conditions of the farm wetland (Table 1), occurring, for example, during an extreme rain event.

2.4 | Examination Approaches and Statistical Analysis

On average, the influent and effluent samples were taken manually according to a pre-established schedule. The water quality analyses focused primarily on the determination of TN and NO_3-N, as well as TP and PO_4-P, the nutrient forms mostly present in agricultural runoff. The analysis of nutrient concentrations in the influents and effluents was performed through Hach-Lange kits and a DR3900 Hach-Lange spectrophotometer.

The concentration removal efficiency (%) of each system was calculated using Equation (2):

$$Removal\ Efficiency\ (\%) = \frac{C_{in} - C_{out}}{C_{in}} \cdot 100 \quad (2)$$

where C_{in} (mg L^{-1}) is the average inlet concentration measured during a sampling period (Table 1), whereas C_{out} (mg L^{-1}) is the single outlet concentration of the chosen parameter during the sampling day.

Also, the mass removal efficiency/retention (%) of the pilot systems or the farm wetland was estimated with Equation (3):

$$Removal\ Efficiency\ /\ Retention\ (\%) = \frac{V_{in} \cdot C_{in} - V_{out} \cdot C_{out}}{V_{in} \cdot C_{in}} \cdot 100 \quad (3)$$

where V_{in} and V_{out} are overall volumes (L $period^{-1}$) flowing in and out of a system and C_{in} and C_{out} are average concentrations (mg L^{-1}) of a particular pollutant in influent and effluent over a period considered. When considering the pilot plant, removal efficiency was the parameter used to assess its performance. On the other hand, in the case of the farm wetland, retention is a more correct term since the system is not waterproofed and therefore losses of water and nutrients to groundwater are possible.

The acquired data were processed through ANOVA and Tukey's Honest Significant Difference tests to evaluate the values distribution and if there were any significant differences ($p < 0.05$) between the pollutant removal efficiencies measured in different time periods. ANOVA was employed to test the null hypothesis that no significant variation exists among the means of pollutant removal efficiency across the different measurement periods, compared to the alternative hypothesis that at least one period significantly differs from the others (Dan et al. 2023). After obtaining a significant ANOVA result, Tukey's Honestly Significant Difference (HSD) post hoc test was applied to perform pairwise comparisons to precisely identify which periods exhibited statistically distinct differences in pollutant removal performance (Tukey 1949; Agbangba et al. 2024). Through this integrated statistical approach, the study systematically evaluated whether temporal variations and different inflow rates significantly affected the removal efficiency.

3 | Results and Discussion

The presentation and discussion of results are divided into the farm wetland as a full-scale and mature system and into experimental pilot-systems tested to evaluate the intensification potential of farm wetlands.

3.1 | Farm Wetland Performance and Behaviour

During the 7 years of monitoring, the system exhibited considerable variability in both inflow and outflow rates (Figure 3). Since the experimental farm applies precision irrigation system, the excess water (i.e., surface runoff) is not produced during the normal irrigation practices but only in the case of precipitation events, depending on their duration and intensity, as well as a variety of other factors (e.g., temperature, soil water content, vegetation development). Furthermore, that means that the system is not functioning the whole year but only in certain periods of it, and therefore days with inflow for each year/period were used when assessing the farm wetland (Table 3). These rest periods certainly contribute to a higher water and nutrient retention,

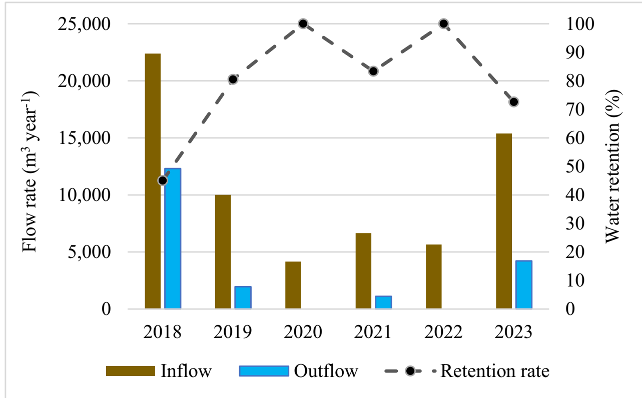

FIGURE 3 | Hydrology of the farm wetland.

but also to the fact that the system, although a mature one, still achieves high performance and does not show the signs of efficiency reduction.

The outflow of the system is a gravitational one and occurs only when the water level in the system exceeds 40 cm; otherwise, the water accumulated does not produce outflow and is subject to different dispersion processes (e.g., infiltration, evapotranspiration). For example, in 2 years, namely 2020 and 2022, there was no outflow from the system since the water level was never higher than 40 cm and the water retention rate can be considered 100%. On the other hand, a slightly lower retention rate observed for 2018 was a consequence of an intensive rain event (Table 3) that caused a constant inflow into the system that, for some weeks, kept the water level constantly above 40 cm, resulting in a high outflow and low water dispersion.

For other years, water retention was always higher than 60%, indicating that the farm wetland, if applied on a larger scale and systematically across a hydrological unit (e.g., irrigation district, watershed) can be considered as a valuable solution for flood mitigation and surface runoff quantitative management.

As for the retention performance, the farm wetland showed a rather good abatement of nutrients (Table 3). Although for 2020 and 2022 it can be considered as 100%, since there was no outlet from the system, in other years it was also very high. An exception was 2018, when TP retention was negative due to high water flow that might have disturbed the TP binding sediments, as well as due to a high concentration of TP in rain water (Lavrnić, Nan, et al. 2020). For other years, TN retention was always over 65% and for TP the minimum was 46%, underlining the importance of the farm wetland for the management of diffuse pollution and surface water protection.

When assessing the farm wetland operation, it is also important to check the loading rates, both in terms of hydraulics and nutrients. In this regard, it can be noted that these values were rather variable during the experimental period (Table 3), being the result of a difference in precipitation intensity and distribution and consequently in the inflow rates (Figure 3). Moreover, the average loading rates to the system were lower than loading rates commonly reported for SFCWs treating agricultural runoff. For example, Vymazal et al. (Vymazal

TABLE 3 | Performance of the farm wetland in the period 2018–2023.

Parameter	2018	2019	2020	2021	2022	2023	Event 2018	Event 2023
Days with inflow (day)	194	77	37	98	78	93	*82*	*23*
Hydraulic loading rate (cm day^{-1})	2.7	2.6	1.6	0.8	1.1	4.5	*5.7*	*9.6*
TN loading rate (g TN m^{-2} day^{-1})	0.40	0.48	0.43	0.01	0.16	0.44	*0.91*	*0.99*
TN retention (%)	40	78	100	65	100	87	*37*	*76*
TP loading rate (mg TP m^{-2} day^{-1})	0.56	0.28	1.01	0.47	0.64	11.75	*0.99*	*39.13*
TP retention (%)	−22	100	100	46	100	90	—	*88*

et al. 2020) reported the range of 0.003–3.087 g TN m^{-2} d^{-1}, with an average value of 0.35 g TN m^{-2} d^{-1}. As for TP, Johannesson et al. (Johannesson et al. 2015) found the values in the range of 0.01–0.41 g TP m^{-2} d^{-1} for SFCWs in Southern Sweden, the values much higher than those for the farm wetland. Similarly, the hydraulic loading rate was at the lower end of the range (0.36–13.30 cm day^{-1}) given by Kadlec (Kadlec 2016) for large scale SFCWs. These findings suggest that the farm wetland system may reach higher loading rates, offering potential for area and cost optimisation in future farm-scale designs. On the other hand, the current farm wetland might also serve a bigger area, contributing to the attractiveness of these systems for the farmer community.

The annual values give a good estimation of the overall performance of the system. However, it is important to note that they are not always representative of single runoff events, but they are rather average of single flow events and rest periods. To explore better the system performance during shorter periods of intensive periods of operation, two runoff events, which occurred in February–April 2018 and May 2023, were analysed separately (Table 3).

These events, associated with precipitation events of 264 mm and 309 mm, can be characterised as episodes of constant runoff production and, consequently, continuous inflow to the farm wetland, without rest periods. Their scale is important since they represented 90% and 52% of total the annual inflows in 2018 and 2023, respectively. Such conditions of high and constant inflow in relatively short periods of time (82 and 23 days) resulted in water retention rates of 25% and 48% for the Event 2018 and Event 2023, respectively, slightly lower compared to the annual averages (Figure 3).

Moreover, it can be seen that the hydraulic and nutrient loading rates for these two events were often much higher than the annual averages, especially the TP loading rate of the Event 2023. However, in spite of the high stress under which the farm wetland was, its retention efficiencies remained rather satisfactory and higher or in line with the average yearly rates. The 2018 event had a negative retention of TP, causing the same trend also for the whole year, probably as a result of sediment resuspension and high TP concentration in rain water (Lavrnić, Nan, et al. 2020). However, this can be considered as an extreme condition, not a frequent one. On the other hand, for the 2023 event, retention above 75% for both TN and TP was recorded, suggesting that the farm wetland, even after 23 years of operation, can perform well under extreme conditions and that the hypothesis of lower surface requirements for these systems might be correct.

3.2 | Pilot System Performance

Considering the results obtained for the farm wetland, this research aimed also to explore whether similar treatment performance could be achieved in systems with a smaller surface but with higher loading rates. Therefore, different pilot-scale CWs, including systems with biochar amendments aimed at enhancing nutrient adsorption, were tested under controlled conditions and higher stress in terms of water and nutrient loading rates.

3.2.1 | Nitrogen Removal in Pilot-Scale Systems

Figures 4 and 5, respectively, report the TN and NO_3-N removal capacities of the three pilot-scale CWs throughout the experimentation, grouped in seasonal periods (Table 2). It was observed that, in terms of concentration reduction, the SFCW system reduced generally TN and NO_3-N more than the HFCW-B and HFCW-G, with overall average efficiencies of 44%, 18% and 18% for TN, and 44%, 15% and 16% for NO_3-N, respectively. Such behaviour was also observed in terms of mass balance, where the SFCW resulted to have removed about two times more of TN and NO_3-N, compared to the HFCWs.

The higher overall efficiency of the SFCW in TN (and NO_3-N) removal may be related to higher plants' uptake. In addition, the higher free-water volume and water content may have enhanced the mobility and mixing of dissolved nitrogen species within the system, increasing their interaction with plant tissues and treatment processes (Vymazal 2013). This hypothesis is consistent with literature, where SFCWs were shown as better solutions in warmer contexts. Moreover, the rapid decrease of performance with the temperature decrease, as observed towards the end of the experimentation, could have been caused by increased water viscosity and reduced plants' metabolic activity, thus lowering their capacity to process the nutrients and hindering the molecules' mixing (Stein and Hook 2005; Varma et al. 2021).

Furthermore, the HRT in HFCWs was smaller (e.g., 3.8 vs. 7.2 days in Early Summer or 2.3 vs. 4.3 days in Late Autumn), an aspect that may have provided the molecules with less contact time with the plants and as reported in the literature, shorter retention times can

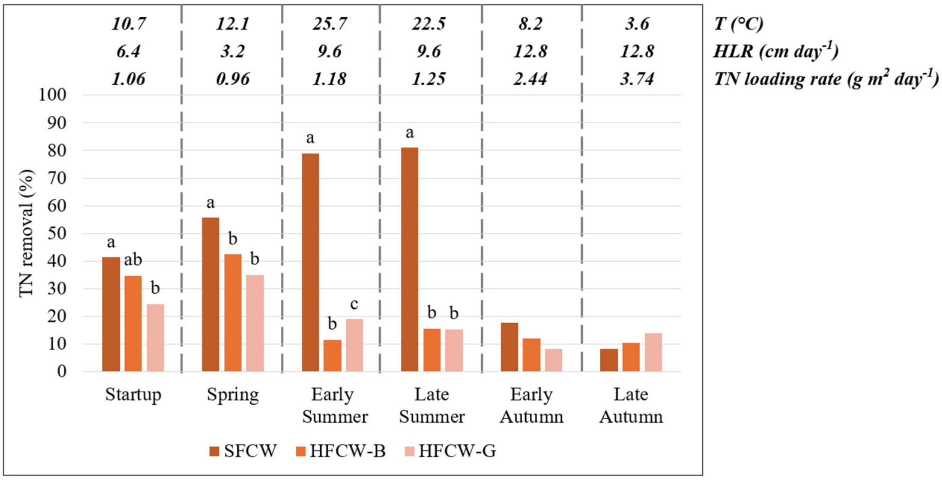

FIGURE 4 | Average TN removal of the three pilot systems per period. Different letters (a, b, c) indicate statistically significant differences among the systems within the same sampling period according to Tukey's HSD test ($p < 0.05$), whereas shared letters (e.g., ab) indicate no significant difference from groups labelled with either letter.

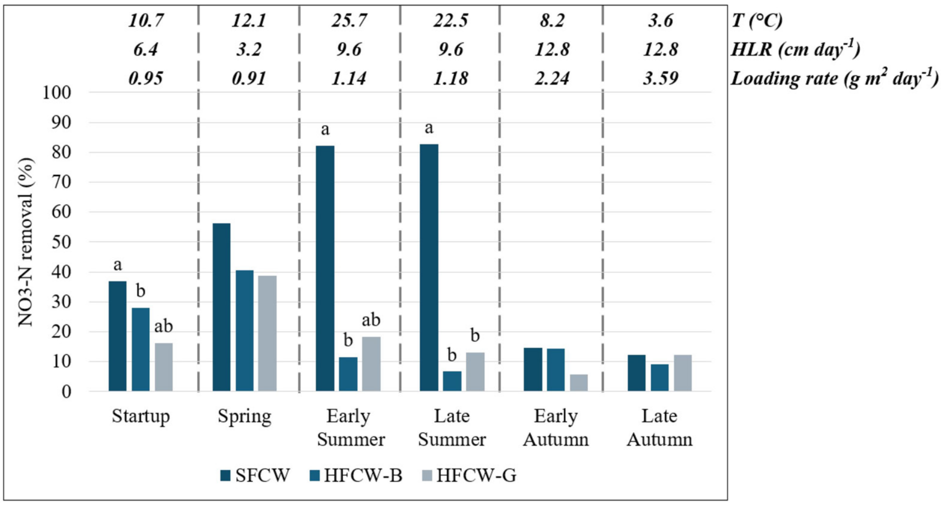

FIGURE 5 | Average NO_3-N removal of the three pilot systems per period. Different letters (a, b, c) indicate statistically significant differences among the systems within the same sampling period according to Tukey's HSD test ($p < 0.05$), whereas shared letters (e.g., ab) indicate no significant difference from groups labelled with either letter.

cause a decrease in compounds removal (Ranieri et al. 2013; Gao and Zhang 2022). In addition, the substrate within the HFCWs could have obstructed the distribution of TN (and NO_3-N) molecules and the exchange surfaces of the plants, resulting in overall lower performances in comparison to the SFCW, an aspect that is consistent with literature (Varma et al. 2021).

During the Start-up period, SFCW removed 41% of TN (and 37% of NO_3-N), performing statistically better than HFCW-G, whereas the two HFCWs showed similar performances with TN (between 24% and 35%) and NO_3-N reductions (between 16% and 28%). The addition of biochar therefore did not significantly influence the performance of HFCW-B in the initial period. In Spring, the change of flow rate from 128 to 64 L d^{-1} and the increase in average inlet concentration (from 16.6 to 28.7 mg L^{-1} for TN and from 14.8 to 27.1 mg L^{-1} for NO_3-N) appear to have influenced the removal efficiency trend of HFCW-B compared to the Start-up period. The SFCW, however, showed statistically higher removal in comparison with the two HFCWs.

With the further increase in temperatures between Spring and Early Summer (from 12.1°C to 25.7°C) and flow rate (from 64 to 192 L d^{-1}), as well as with lower inlet average concentration (from 28.7 to 12.1 mg L^{-1} for TN and from 27.1 to 11.7 mg L^{-1} for NO_3-N), SFCW experienced a steep increase in TN and NO_3-N removal, presenting an Early Summer average of 79% and 82%, respectively and a strong difference with the HFCWs. Conversely, as said before, HFCW-B showed a notable decrease in performance, removing only around 11% of inlet TN and NO_3-N, lower than HFCW-G.

Although the plant uptake and microbial pathways can be considered as valid removals for all the systems, another reason for the superior TN and NO_3-N removal performance of the SFCW compared to the HFCWs may relate to the conditions associated with increased temperatures. During the warmer periods, in the case of HFCWs, with a higher substrate content, a higher energetical status of the active sites may have been achieved, consequentially lowering adsorption onto their surfaces, causing the observed decrease in TN and NO_3-N (Chabani et al. 2006; Liu et al. 2022). In contrast, for the SFCW, with a lower substrate content, the higher temperatures may have promoted the microbial reactions, causing higher removal of TN and NO_3-N (Brienza et al. 2023).

In addition to the significant performance differences between the SFCW and the HFCWs during the Summer periods, a statistically significant disparity was also observed between the two HFCW systems themselves. Notably, the performance of HFCW-B declined noticeably from Spring to Early Summer, resulting in poorer TN and NO_3-N removal compared to HFCW-G. The higher initial performance of the biochar-added HFCW-B in comparison to the gravel-only HFCW-G could be due to the adsorption potential of the material itself, which granted a higher removal during the first stages. However, throughout the experiment, its active sites may have quickly saturated; hence, the rapid decrease in performance observed during the later stages and thus the performances similar to HFCW-G (Yao et al. 2023).

Another perspective emerges from reports by various studies highlighting a relationship between biochar performance and temperatures around 15°C. For instance, He et al. (He et al. 2016) reported that biochar, when used as a carrier for biofilms, demonstrated promising results for TN removal at temperatures below 15°C. Additionally, several studies have indicated that under extremely high temperatures, biochar may lose its effectiveness in nutrient retention due to alterations in its physical and chemical properties (Dróżdż et al. 2023). These findings suggest that as temperatures rose from Spring (12.1°C) to Early Summer (25.7°C), passing the critical 15°C threshold, the functionality of biochar may have been adversely affected. When coupled with potential saturation issues, this may have caused the sharp decline in the performance of the HFCW-B during the Early Summer period. With slightly lower temperatures (22.5°C) and higher inlet TN and NO_3-N concentrations (13.1 mg L^{-1} and 12.3 mg L^{-1}, respectively), during the Late Summer period, the HFCWs showed no statistically significant difference in performance, with TN removal efficiency at 15% and NO_3-N removal efficiency ranging from 6% to 12%. In contrast, the SFCW continued to significantly outperform both HFCWs achieving average TN and NO_3-N removal efficiencies of 81% and 83%, respectively.

From Late Summer to Early Autumn, the inlet concentration was increased to 15.3 mg TN L^{-1} and 14.0 mg NO_3-N L^{-1}, as well as the flow rate to 320 L d^{-1}, whereas the average temperature experienced a sharp drop to 8.2°C. As a result, SFCW exhibited a notable decline in TN and NO_3-N removal efficiency and all the CW performances ranged from 8% to 18% for TN removal and from 6% to 15% for NO_3-N removal, with no statistically significant differences. This decline may be attributed to a combination of factors, including reduced microbial activity, decreased plant uptake and slower biochemical reaction rates (Chen et al. 2022). Such temperature-induced limitation may have lowered the performance of the SFCW to levels comparable to those of the HFCWs. With the even lower temperatures of Late Autumn (3.6°C) and higher inlet TN and NO_3-N concentration (23.4 mg L^{-1} and 22.4 mg L^{-1}, respectively), the observed performances were similar to Early Autumn, with slight variations. During this period, no statistically significant differences were observed among the systems and the CWs exhibited TN removal efficiencies ranging from 8% to 14% and NO_3-N removal between 9% and 12%.

3.2.2 | Phosphorus Removal in Pilot-Scale Systems

As shown in Figure 6 and Figure 7, the HFCWs proved more effective in removing TP and PO_4-P compared to the SFCW, with HFCW-B and HFCW-G each achieving an overall average TP removal of 43%, compared to 33% for the SFCW. Consistently, phosphate (PO_4-P) removal efficiencies were noted at 44% for HFCW-B, 41% for HFCW-G and 33% for the SFCW.

It was observed that all the systems presented a progressive reduction in terms of TP and PO_4-P removal efficiency over time, in particular with colder temperatures, the trend consistent with the fact that the TP (and PO_4-P) removal is a phenomenon characterised by saturation of both the plants and the substrate. In

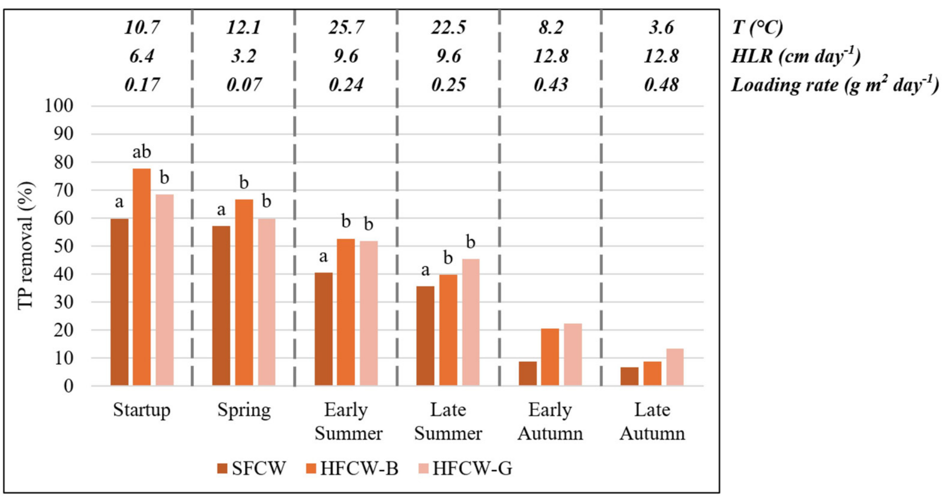

FIGURE 6 | Average TP removal of the three pilot systems per period. Different letters (a, b, c) indicate statistically significant differences among the systems within the same sampling period according to Tukey's HSD test ($p < 0.05$), whereas shared letters (e.g., ab) indicate no significant difference from groups labelled with either letter.

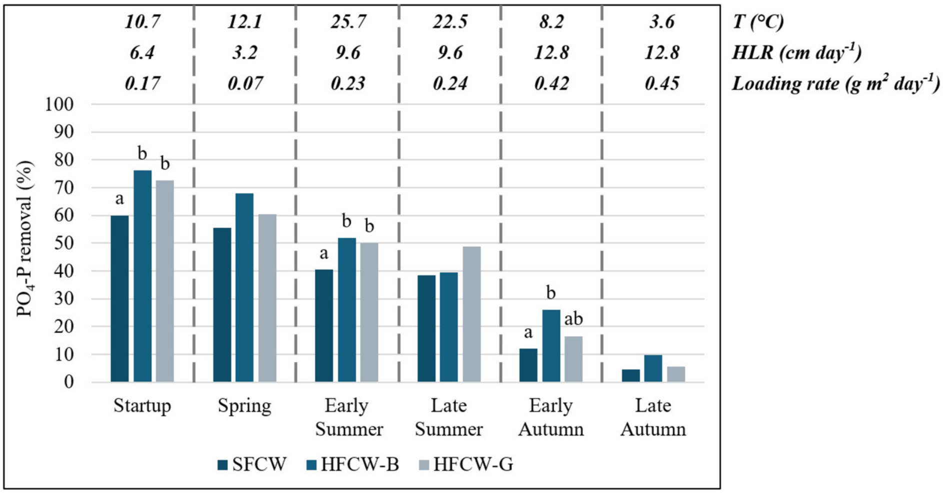

FIGURE 7 | Average PO_4-P removal of the three pilot systems per period. Different letters (a, b, c) indicate statistically significant differences among the systems within the same sampling period according to Tukey's HSD test ($p < 0.05$), whereas shared letters (e.g., ab) indicate no significant difference from groups labelled with either letter.

fact, the better TP (and PO_4-P) removal capacity observed for HFCW-B and HFCW-G compared to the SFCW, suggest a higher reliance on the substrate than on the biotic components in such systems (Vymazal 2007; Gao and Zhang 2022). Additionally, factors such as longer contact time with the substrate and anaerobic-anoxic conditions can also contribute to improved phosphorus removal. In HFCWs, water flows through a porous medium (e.g., gravel), which provides ample surface area for the adsorption and precipitation of phosphorus onto substrate particles (Gao and Zhang 2022). Ronkanen et al. (Ronkanen et al. 2016) also emphasise that TP removal often relies on chemical binding (e.g., with iron or aluminium), a process that can be enhanced in the low-oxygen environments typical of HFCWs. Based on these insights, the better performance of HFCWs in TP and PO_4-P removal appears also reasonable. Furthermore, the initially higher performance of HFCW-B (until Early Summer) in TP and PO_4-P removal and its steeper decrease in comparison to HFCW-G matches with the biochar saturation hypothesis.

During the Start-up period, SFCW showed a significantly lower performance regarding TP in comparison to HFCW-B, with 60% TP removed. Additionally, a difference was also observed between the two HFCWs (68% for HFCW-G and 78% for HFCW-B), although not on a statistical level. A similar observation was made by (Gupta et al. 2016), who emphasised the role of biochar in enhancing the removal of P-related pollutants in CWs. Indeed, the TP removal performance of HFCW-B during the early stages, specifically the Start-up, was noticeably better than that of the other systems. Furthermore, during the Start-up period, similar to the TP behaviour, the SFCW demonstrated significantly lower PO_4-P removal (60%) compared to HFCWs, whereas the HFCW systems showed statistically similar removal efficiencies, ranging from 72% to 76%. This may suggest that whereas biochar can be effective in TP removal, its efficiency in removing PO_4-P during the early phases of treatment may be relatively lower. This aspect may be attributed to factors such as biochar type, modification methods and environmental conditions (Zhang et al. 2022, 2024).

No significant differences in performances were observed during Spring, when the CWs presented comparable TP and PO_4-P removal. During the Early Summer period, the SFCW performed worse than the HFCWs, removing 41% compared to 51%–52% of TP. This may imply that substrate plays a more dominant role than biotic components in the removal of P-related pollutants (Vymazal 2007; Gao and Zhang 2022). In Late Summer, the CWs experienced a decline in performance, although their performance difference was not statistically significant. However, similar to previous stages, the HFCWs, with removal efficiencies between 40% and 45%, performed slightly better than the SFCW, which removed around 36% of inlet TP. In this period, also, there was no statistically significant difference in the PO_4-P removal performance of the CWs, with their efficiencies ranging between 39% and 49%.

Interactions with plants and microorganisms, however, could have occurred and may have influenced the systems. From the Late Summer period onwards, the CWs showed statistically similar TP removal performance and a steady decrease along time, especially with the lower temperature of the last phases, supporting the hypothesis of active sites saturation (Vymazal 2007). This declining trend was particularly evident in Early Autumn, when the TP removal efficiency of the SFCW dropped to 9%, whereas the HFCWs maintained removal efficiencies between 20% and 22%. Similarly, the SFCW achieved only 12% PO_4-P removal, compared to 26% by HFCW-B, a statistically significant difference. In contrast, HFCW-G, with a removal efficiency of 17%, remained statistically comparable to the other two systems. This may indicate that, although SFCWs were identified as less effective in phosphorus removal compared to HFCWs, the presence of biochar may have mitigated the decline in PO_4-P removal efficiency during the transition from warmer to colder seasons. Incorporating biochar into HFCW-B may have therefore effectively buffered against the seasonal decline in phosphorus removal efficiency by providing a stable adsorption mechanism less affected by temperature variations (Gao and Zhang 2022). Additionally, the drop of performance observed with higher flow rates (during

Autumn periods) could be explained by findings of former research, where lower removal rates were connected to higher hydraulic loading rates (Kill et al. 2022). In Late Autumn, the performances of all systems were similar, with TP and PO_4-P removal efficiencies ranging from 7% to 14% and between 12% and 17%, respectively.

3.3 | Biochar Contribution to Nutrient Removal

The effect of biochar addition on nutrient removal was assessed by comparing the performance of the biochar-amended (HFCW-B) and gravel-only (HFCW-G) systems. For TN, the presence of biochar resulted in a higher removal during the Start-up period, with an additional 22.4 g removed compared to the HFCW-G. However, this difference decreased over time, consistent with the biochar saturation hypothesis and in several subsequent periods the performances of the two systems were similar or the HFCW-B showed slightly lower removal. A notable reduction in the performance of HFCW-B compared to HFCW-G was observed during the Early Summer period, which may be related to the combined effect of higher temperatures and possible saturation of adsorption sites, reducing the contribution of sorption processes and favouring biologically driven removal pathways (Chabani et al. 2006; Liu et al. 2022). Overall, the cumulative contribution of biochar to TN removal remained limited, suggesting that nitrogen removal in the studied systems was predominantly governed by microbial processes, in agreement with previous findings for CWs (Vymazal 2013; Chen et al. 2022).

For TP, the addition of biochar did not result in a clear improvement in removal performance. Although slightly higher removal was observed in the initial phase, the differences between HFCW-B and HFCW-G were generally small and in several periods the gravel-only system performed similarly or better than the biochar-amended system. This behaviour suggests that the contribution of biochar to phosphorus removal was limited and not sustained over time. The initial improvement followed by a decrease in performance may indicate progressive saturation of available sorption sites, a process commonly reported for phosphorus removal in porous media and CWs (Vymazal 2007). A temporary increase in the relative performance of HFCW-B was observed during the Early Autumn period, which may be associated with lower temperatures influencing sorption processes or microbial activity on the biochar surface (Zhang et al. 2015; Guo et al. 2023). However, this effect was not maintained in subsequent periods.

The trends observed for NO_3^--N and PO_4^{3-}-P were consistent with those obtained for TN and TP, respectively, supporting the interpretation that the addition of biochar did not provide a sustained improvement in nutrient removal. Overall, these results indicate that, under the tested conditions and at the applied dosage (10% v/v), biochar acted as a reactive component mainly during the initial stages of operation, whereas its long-term contribution to nutrient removal remained limited.

3.4 | Verification of the Farm Wetland Intensification Hypothesis

Rather interesting conclusions can be drawn when the performance of the farm wetland is compared to those of the pilot systems. As previously discussed, the farm wetland consistently demonstrated strong nutrient retention efficiencies over the monitoring period. When not considering years without outlet and therefore 100% retention, ranges of 40%–87% for TN and of 46%–90% for TP were observed, in line with the values expected for full-scale SFCWs, which benefit from extended hydraulic retention times and large buffer volumes (Vymazal 2007; Moreno-Mateos et al. 2010). However, when areal retention rates are considered, there appears to be potential for optimisation. The farm wetland's long-term TN retention ranged from 0.01 to 0.43 $g\,m^{-2}\,d^{-1}$, with a maximum of 0.76 $g\,m^{-2}\,d^{-1}$ during storm events (Table 4). TP retention was much lower and it amounted to less than 0.01 $g\,m^{-2}\,d^{-1}$, spiking only to 0.03 $g\,m^{-2}\,d^{-1}$, highlighting the traditional limitations of SFCW systems for phosphorus treatment when not coupled with special removal measures, such as the use of a phosphorus capturing substrate (Kadlec and Wallace 2009).

In contrast, the pilot-scale systems demonstrated higher removal rates under intensified conditions. The SFCW pilot achieved TN areal removals of 0.38–0.97 $g\,m^{-2}\,d^{-1}$, overlapping with and even exceeding the rates of the farm wetland. Similarly, TP removal in the pilot system was in the range 0.04–0.11 $g\,m^{-2}\,d^{-1}$, much higher than those achieved in the farm wetland (Table 4). The HFCW systems also showed comparable TP removal rates (0.03–0.12 $g\,m^{-2}\,d^{-1}$ for HFCW-B and 0.04–0.13 $g\,m^{-2}\,d^{-1}$ for HFCW-G), while TN removal remained within the range of the farm wetland. These results suggest that similar or higher areal removal rates can be achieved under higher loading conditions, supporting the hypothesis that intensification can compensate for reduced surface area (Ding et al. 2025).

TABLE 4 | Comparison between the farm wetland and pilot systems performance.

	Farm wetland			Pilot systems		
Parameter	2018–2023 (range)	Event 2018	Event 2023	SFCW	HFCW-B	HFCW-G
TN mass retention/removal[a] (%)	40–100	37	76	8.4–81.0	10.4–42.4	8.2–35.1
TN areal retention/removal[a] ($g\,m^{-2}\,day^{-1}$)	0.01–0.43	0.34	0.76	0.38–0.97	0.14–0.40	0.16–0.41
TP mass retention/removal[a] (%)	46–100	—	88	6.8–59.9	8.9–77.6	13.5–68.4
TP areal retention/removal[a] ($g\,m^{-2}\,day^{-1}$)	0.00–0.01	—	0.03	0.04–0.11	0.03–0.12	0.04–0.13

[a]Retention refers to the farm wetland, whereas removal characterises pilot systems.

As discussed in Section 3.3, no consistent improvement in nutrient removal was observed due to biochar addition in the HFCW across the different operational periods. Although a slight enhancement was observed during the initial stages, the performance of the biochar-amended system generally converged towards that of the gravel-based system over time, indicating limited long-term benefits.

From a design perspective, these findings suggest that system intensification can be achieved through a combination of strategies. Increasing areal loading rates, while maintaining comparable removal efficiencies, indicates that the footprint of farm wetlands could potentially be reduced. Furthermore, the implementation of staged or modular configurations, where treatment units are activated progressively under increasing flow conditions, may improve hydraulic control and enhance system performance during peak events. In this context, the use of reactive media such as biochar may be more suitable in targeted polishing units rather than as a bulk substrate, given its limited long-term effectiveness and susceptibility to saturation.

Although intensification may allow for a reduction in system footprint and maintain high areal removal rates, it may also reduce the capacity of the system to retain water. The full-scale wetland demonstrated a significant buffering function, with high water retention rates during several periods, whereas the pilot systems operated under continuous flow conditions with limited storage capacity. This suggests that intensification strategies may involve a trade-off between treatment efficiency and hydrological buffering, which should be carefully considered in system design. Moreover, systems operating under higher loading rates and reduced volumes may exhibit increased sensitivity to temperature fluctuations and hydraulic variability. Therefore, although the pilot systems demonstrated promising performance under controlled conditions, careful consideration of operational stability and resilience is required when translating these results to full-scale applications.

It should be noted however that pilot-scale systems often report higher removal efficiencies than their full-scale counterparts, partly due to reduced hydraulic complexity, lower risk of preferential flow and optimised loading during controlled experiments. In addition, the pilot trials in this study were of relatively short duration and therefore might not capture long-term behaviour well and possible performance reductions that might happen over time (Riquelme et al. 2022).

4 | Conclusions

The current research focused on farm wetlands that are systems often used to treat agricultural runoff. A mature system in Italy, constructed more than 20 years ago, still had good performance for water and nutrient management. On the other hand, one of the major problems of these systems is associated with their surface requirements. Therefore, different types of CWs were tested on a pilot scale to explore possible intensification of these systems and lower area needed to achieve high performance.

The present study confirmed the effectiveness of CWs as NbSs for the treatment of agricultural runoff, both at full scale and pilot scale. The long-term monitoring of the farm wetland demonstrated stable performance in terms of water and nutrient retention, highlighting its suitability for managing diffuse pollution and mitigating runoff volumes.

The pilot-scale experiments provided additional insights into system optimisation and intensification potential. The results showed that similar or higher areal removal rates could be achieved under increased hydraulic and nutrient loading conditions, suggesting that the footprint of farm wetlands could be reduced through appropriate design strategies. However, system performance was strongly influenced by seasonal variations, particularly temperature, which affected nitrogen removal processes more markedly.

The comparison between different configurations, when the same HLR was applied, indicated that SFCW systems were more effective for nitrogen removal, whereas HFCW systems showed better performance in phosphorus removal. The addition of biochar did not result in a consistent improvement in nutrient removal over time. Although a positive effect was observed during the initial stages of operation, this effect was not sustained, suggesting a limited long-term contribution of biochar under the tested conditions.

Overall, the results highlight that the intensification of CWs is feasible, but requires careful balancing between increased loading rates and system resilience. Design strategies such as hydraulic optimisation and modular configurations may support performance under intensified conditions, whereas the use of reactive media should be evaluated considering their long-term effectiveness. These findings contribute to the development of more efficient and scalable NbSs for agricultural water management.

Author Contributions

Stevo Lavrnić: writing – original draft, writing – review and editing, validation, conceptualization; **Francesco Chioggia:** methodology, formal analysis, writing – original draft, writing – review and editing; **Salvatore Luca Gentile:** data curation, investigation, writing – review and editing; **Giuseppe Mancuso:** methodology, investigation, writing – review and editing; **Peyman Arjomandi Akram:** visualisation, formal analysis, writing – review and editing; **Francesco Cavazza:** validation, resources, writing – review and editing; **Attilio Toscano:** writing – review and editing, resources, conceptualization, supervision.

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During the preparation of this work, the authors used ChatGPT to improve the language and readability of the manuscript. After using this tool, the authors carefully reviewed and edited the content as needed and take full responsibility for the content of the published article.

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Ethics Statement

The authors confirm that this manuscript is original, has not been published before and is not currently being considered for publication elsewhere.

Consent

The authors have nothing to report.

Conflicts of Interest

The authors declare no conflicts of interest.

Data Availability Statement

The data that support the findings of this study are available on request from the corresponding author. The data are not publicly available due to privacy or ethical restrictions.

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