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Research article

Performance of lagoon and constructed wetland systems for tertiary wastewater treatment and potential of reclaimed water in agricultural irrigation

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A R T I C L E I N F O

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A B S T R A C T

Climate change poses challenges to agricultural water resources, both in terms of quantity and quality. As an adaptation measure, the new European Regulation (EU) 2020/741 establishes different water quality classes for the use of reclaimed water in agricultural irrigation. Italy is also working on the definition of a new regulation on reclaimed water reuse for agricultural irrigation (in substitution of the current one) that will also include the specific requirements imposed by the European one. Nature-based Solutions (NBS) can be a cost-effective and environmentally friendly way to facilitate water reclamation and reuse. The present study reports the outcomes of a long-term monitoring campaign of two NBS (e.g., a constructed wetland (CW) and a lagoon system (LS)) comparing influent and effluent concentrations of different contaminants (e.g., *E. coli*, BOD_5, TSS, TN and TP) with the threshold values imposed by the new regulations. The results showed that in both the case studies, *E. coli* (about 100 CFU 100 mL^{-1}) and BOD_5 (lower than 25 mg L^{-1}) mean effluent concentration need to be further reduced in reclaimed water to be suitable for unlimited reuse. As a negative aspect, in both the monitored NBS, an increase in TSS mean concentration in the effluent was observed, up to 40 mg L^{-1} in the case of the LS, making reclaimed water unsuitable for agricultural reuse. The CW has proven to be more effective in nitrogen removal (the effluent mean concentration was 3.4 mg L^{-1}), whereas the LS was better at phosphorus removal (with an effluent mean concentration of 0.4 mg L^{-1}). Based on the results, recommendations were made to further improve the performance of both systems in order to have adequate water quality, even for class A. Furthermore, the capacity of reclaimed water to meet crop water and nutrient needs was analyzed, and total nitrogen removal rate coefficients were calculated for the design of future LSs.

1. Introduction

Climate change has altered the hydrological cycle, causing water scarcity and reducing water availability (Gosling and Arnell, 2016). This situation is having a negative impact on agriculture, a sector that requires high water volumes to ensure food production that must respond to the growing needs of the world population (Fukase and Martin, 2020). In this context, using reclaimed water in agricultural irrigation can represent a sustainable solution to adapt to climate change and mitigate its negative effects, such as water scarcity (Mancuso et al., 2020). Reclaimed water is considered a reliable and continuous water source, relatively independent from seasonal drought and weather variability, and it can be able to satisfy crop water demand partially, consequently reducing the risk of crop failure (Mancuso et al., 2022; Nan et al., 2020).

However, reclaimed water can pose a risk to human, animal and environmental health if not properly treated. Indeed, reclaimed water can contain conventional pollutants (e.g., organic matter, nutrients) (Henze and Comeau, 2008), emerging contaminants (e.g., microplastics, heavy metals, antibiotics) (Bolong et al., 2009) and pathogens (e.g.,

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bacteria, viruses) (Crockett, 2007). Not by chance, the European Union has recently introduced, through the new regulation (EU) 2020/741 (Regulation (EU) 2020/741, 2020), the minimum requirements for water reuse, consisting of a minimum of parameters that are essential to be monitored to ensure the provision of reclaimed water with a certain quality. For these parameters, the regulation provides threshold values in order to reduce the risk associated with the presence of different contaminants (Mancuso et al., 2021b). The presence of nutrients in reclaimed water can reduce the use of chemical fertilizers and avoid issues of secondary contamination, promoting the gradual transition from conventional irrigation to fertigation with reclaimed wastewater (Mainardis et al., 2022). So far, in Italy, the use of reclaimed water was regulated by the Ministerial Decree 185/03 (Ministerial Decree n. 185, 2003), which, together with the Legislative Decree 152/06 ("Legislative Decree 152/2006 - Testo Unico Ambientale, 3 aprile 2006, n. 152," 2006), established the technical standards for the reuse of reclaimed water in agricultural irrigation. Recently, a new decree (Decree of the President of the Republic, 2023) was proposed, aiming at standardizing the Italian regulation with the European one; it should effectively replace the old one. As in the older version, in the new decree standard limitations are also provided for nutrients in reclaimed water, since the sites that are particularly vulnerable to eutrophication are quite widespread within the Italian territory.

In this context, new technologies have emerged in wastewater treatment plants (WWTPs) in order to enhance the wastewater treatment efficiency before reclaimed water is discharged or reused, thus ensuring human and environmental health (Bairagi and Ali, 2020). However, even if these technologies are effective, they are often expensive (Ahmed et al., 2021). To overcome this issue, the implementation of efficient, low-cost and environmentally-friendly wastewater treatment methods is being proposed.

CWs and LSs are two kinds of NBS that mimic the processes occurring in natural wetlands and ponds, and they can be used to treat wastewater. NBS have been widely tested in the treatment of primary or secondary municipal wastewater effluents (Abou-Elela, 2017; Allen et al., 2022), allowing different physicochemical and biological processes, which involve the interaction of wetland vegetation, soil and microbial community (Mancuso et al., 2021a), and are therefore considered affordable and reliable green technologies. Compared to conventional treatment systems, NBS are low-cost (Waly et al., 2022), have fewer operational and maintenance requirements (Ayaz and Akca, 2000) and offer an environmental-friendly approach (Kataki et al., 2021; Singh et al., 2022). NBS are also used to promote additional ecosystem services, such as increasing biodiversity or providing recreational areas (Harrington and McInnes, 2009). However, NBS also have some drawbacks such as high land area requirements, the need for preliminary wastewater treatment and higher retention times if compared to conventional treatment facilities (Simelton et al., 2021).

In most cases, NBS are used for tertiary treatment, either as a polishing step after secondary treatment or to treat specific polluted streams, e.g., to reduce the nutrient load of agricultural runoff (Mancuso et al., 2021a). The NBS efficiency depends on i) wastewater composition (organic matter and nitrogen compounds content, etc.), ii) design and operational parameters (hydraulic retention time, water depth, etc.), and iii) environmental factors (temperature, presence of wildlife, etc.) (Kadlec and Wallace, 2009; Persson, 2000). The relationship between these factors and the NBS performance can be explained by mathematical modelling when sufficient data are available (Guo et al., 2020). In addition to understanding the functioning of NBS, models can also be used for their design or optimization to achieve an effluent quality according to the desired use, e.g., reclaimed water for agricultural irrigation. The most used simple hydraulic models are the tank-in-series (TIS) model and plug-flow with dispersion model coupled with first-order degradation kinetics to predict the average pollutant removal (Canet-Martí et al., 2022; Kadlec, 1994). To model the removal of different pollutants affected by temperature, Arrhenius equation can be applied to determine the temperature correction factor (θ) and the removal rate coefficient at 20 °C (k_{20}) (Kadlec and Wallace, 2009). Once θ and k_{20} are determined, they can be used for the design of NBS treating wastewater with similar characteristics.

The present study reports a comparison between two different NBS (e.g., CW and LS), that served as a tertiary treatment within two different WWTPs in Northern Italy. The systems were compared on the basis of the outcomes of a multi-year monitoring campaign aimed at evaluating their performance, assessing also their capability to produce reclaimed water that was suitable for agricultural irrigation according the current regulation on water reuse. These two full-scale systems can be considered as mature ones and therefore the study provides important indications about the removal efficiencies that can be reached, as well as on the future design and optimization strategies for NBS.

2. Materials and methods

2.1. Study area

The investigation was conducted at San Matteo della Decima (case study A) and Imola (case study B) treatment facilities, both of them located in the Emilia-Romagna region. The San Matteo della Decima WWTP (44°42′56.4″N, 11°13′13.3″E) was designed for 5500 population equivalents (PE), while the Imola WWTP (44°21′12.3″N, 11°44′15.1″E) has a higher treatment capacity since it was designed to treat 75,000 PE. Both WWTPs are managed by the HERA Group, which is the main utility within the Emilia-Romagna region that deals with waste, water and energy management and control.

2.1.1. Case study A: San Matteo della decima

The San Matteo della Decima WWTP is equipped with preliminary treatment (e.g., coarse and fine filtering grid, desander), biological treatment (e.g., oxidation/nitrification), and phosphate and nitrate removal unit. At the end of the treatment chain, a CW is implemented with the role of tertiary treatment (Fig. 1). The CW comprises a sedimentation basin (3), two free water surface (FWS) wetlands in series (4 and 5), and three horizontal flow (HF) wetlands in parallel (6–8). The CW is mainly used to ensure biological disinfection and solids reduction into the WWTP effluent. Moreover, this system is implemented to enhance nitrogen removal, particularly under critical conditions (e.g., high nitrogen loads and very low temperatures). All the treatment compartments can be considered altogether for the estimation of removal efficiency, with a total area of about 1.2 ha and an average water depth of 1.1 m.

The secondary treated wastewater is discharged from the San Matteo della Decima WWTP (1) into an open canal (2). Within the study area (Emilia-Romagna region), open canals are common waterways to convey drainage water into the main receiving water bodies or to supply irrigation water to agricultural cultivated areas; hence, the importance of ensuring appropriate water quality. Before entering the CW, wastewater is collected from the open canal (10) by means of automatic water control gates. The wastewater enters the sedimentation basin (3) to reduce solids content and avoid the clogging of the CW treatment compartments.

The sedimentation basin (3) is connected to the FWS wetland 1 (4), which fed FWS wetland 2 (5). Both FWS wetlands have their edges planted with *Phragmites australis*. Their maximum water depth is 1.6 m and has a total capacity of about 11.4 × 10^3 m^3, being the largest compartments of the system. The sinuous path and the hydraulic disconnection between the two FWS wetlands enhance hydraulic efficiency. Wastewater is then conveyed to the three HF wetlands in parallel (6–8). These treatment compartments are filled with gravel and planted with *Phragmites australis*. The depth of the filter material is 0.6 m, and the wastewater is constantly kept 0.1 m below its surface. All three HF wetlands are connected to the same outlet (11), from which the effluent is discharged into the irrigation canal (9). The reclaimed water is later

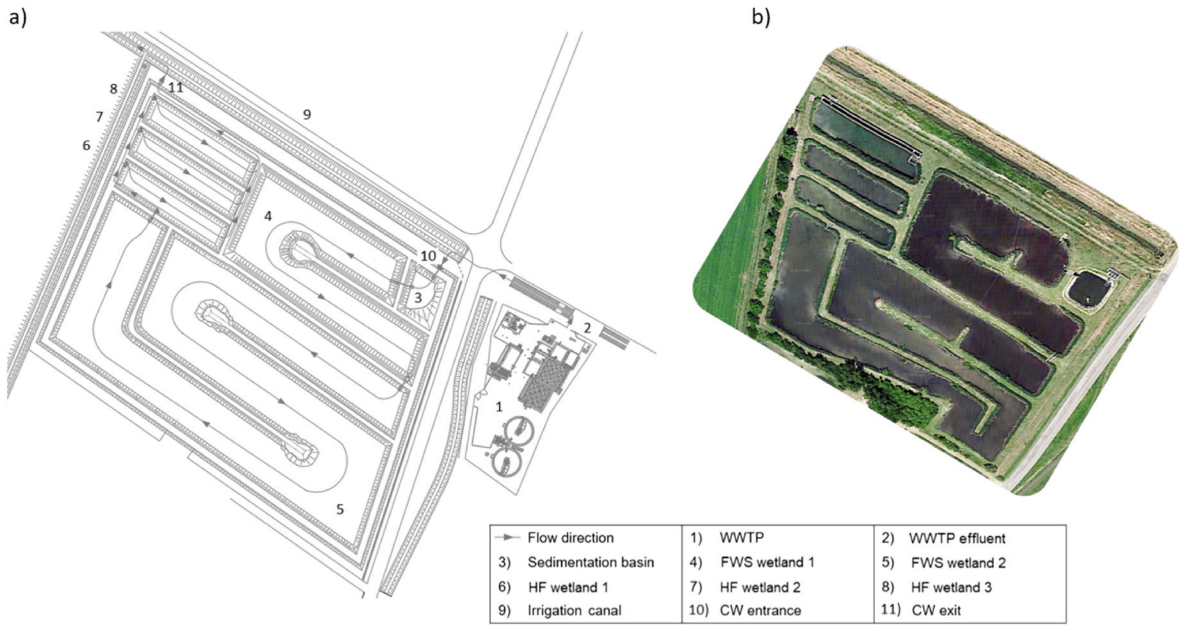

Fig. 1. Case study A: a) schematic representation of the San Matteo della Decima wastewater treatment plant (WWTP) followed by CW; b) picture of the CW.

used for agricultural irrigation within the study area.

Table 1 summarizes the process parameters and operational conditions measured within the CW. In each compartment, the measured flow rate was lower during summer than in winter. HRT was strictly related to flow rate, so it also had higher values during summer compared to winter (data not shown). Wastewater temperature varied between 3.8 °C (winter) and 23.8 °C (summer).

2.1.2. Case study B: Imola

The Imola WWTP has intensive preliminary and secondary treatment units, followed by an LS as tertiary treatment (Fiorentino et al., 2016; Fiorentino and Mancini, 2019). The WWTP is characterized by the presence of two identical treatment lines ("Line 1" and "Line 2"). Primary sedimentation is absent in the WWTP. After the pre-treatment unit (screening), wastewater is treated at the secondary unit, based on pre-denitrification/nitrification within the activated sludge process. The sludge line consists of a thickener, anaerobic digestion and a mechanical dewatering unit with a centrifuge. The treated wastewater (sum of Line 1 and Line 2 effluents) is conveyed to the LS. The WWTP is also equipped with an emergency chemical disinfection unit (i.e., sodium hypochlorite).

Fig. 2 shows a schematic representation of the Imola WWTP, followed by the LS. The main purpose of the LS was to increase retention time to stabilize and polish secondary treated wastewater, especially for pathogens removal.

The Imola LS consists of five basins, as illustrated in Fig. 2 (3–7). For this study, all five basins were considered as a whole, with a total area of about 7.4 ha and an average water depth of 4.2 m. The WWTP effluent (2) is distributed equally to the basins (3) and (4), from which it flows into the basins (5) and (6), and later to the basin (7) before being released into the Santerno river (8). The first two basins (3) and (4) are characterized by a regular shape and reduced dimensions, while the basins (5) and (6) have a higher volume and a more elongated shape. Basin (7) is the final treatment step of the LS, with the possibility of being used also as a reservoir in case of flood waves of the Santerno river. In all the basins, a conspicuous presence of duckweed (*Lemna minor*) was detected during the year, temporarily occupying the whole surface area in summer.

Table 1
Process parameters and operational conditions for the CW (case study A).

CW treatment compartment	Volume	Flow rate	HLR	HRT
	(m^3) x 10^3	(m^3 d^{-1}) x 10^3	(m^3 m^{-2} d^{-1})	(d)
Sedimentation basin (3)	0.5	1.5–1.9	2.5–3.2	0.2–0.3
FWS wetland 1 (4)	5.2	1.5–1.9	0.5–0.6	2.7–3.4
FWS wetland 2 (5)	6.2	1.5–1.9	0.1–0.2	3.3–4.1
HF wetland 1-2-3 (6–8)	0.9	0.5–0.6	0.3–0.4	1.5–1.8

Table 2 summarizes the process parameters and the operational conditions of the LS. The basins (5) and (6) cover the largest area of the LS, with volumes of 78.3 × 10^3 m^3 and 82.1 × 10^3 m^3, respectively. Similar to case study A, the measured flow rate in each compartment was lower in summer compared to winter (data not shown). Wastewater temperature varied between 3.2 °C (winter) and 25.8 °C (summer).

2.2. Analytical methodology

The monitoring period was 7 years (from 2016 to 2022) and 6 years (from 2017 to 2022) for case studies A and B, respectively. In both, wastewater samples were collected at the inlet of the WWTPs, in order to monitor the influent, and at the entrance and exit of the two different NBS, in order to evaluate their performance.

Wastewater samples were collected by the HERA Group approximately every two weeks and at least once a month. They were stored in a cooler at 4 °C, transported to the HERA Group laboratory and analyzed within 3 h. For case studies A and B, a total of 859 (influent 493 + effluent 366) and 4358 (influent 727 + effluent 3631) wastewater samples were collected, respectively. The Biochemical Oxygen Demand (BOD_5), total suspended solids (TSS), total nitrogen (TN), ammonium (NH_4^+-N), nitrate (NO_3^--N), nitrite (NO_2^--N) and total phosphorous (TP) concentrations were determined in accordance with the standard methods (APHA, 2005). The *E. coli* content was determined following the APAT and IRSA-CNR guidelines ("APAT and IRSA CNR - Manuali e Linee Guida 29/2003. Metodi analitici per le acque. Volume Terzo. Sezione 6000- Metodi microbiologici - Parte generale. Sezione 7000-Metodi per la determinazione di indicatori di inquinamento e patogeni.," 2003). It is worth noting that the limit of detection to determine BOD_5 concentration was 5 or 10 mg L^{-1}, while the limit of detection to determine NH_4^+-N was 1.0 mg L^{-1}, representing a limitation for data analysis.

Fig. 2. a) Case study B: schematic representation of the Imola wastewater treatment plant (WWTP) followed by the LS; b) picture of the LS.

Table 2
Process parameters and operational conditions for the LS (case study B).

LS section	Volume	Flow rate	HLR	HRT
	$(m^3) \times 10^3$	$(m^3\ d^{-1}) \times 10^3$	$(m^3\ m^{-2}\ d^{-1})$	(d)
Basin (3)	49.5	16.5–24.7	1.5–2.3	2.0–3.0
Basin (4)	48.9	16.3–24.4	1.5–2.3	2.0–3.0
Basin (5)	78.3	13.0–19.6	0.7–1.0	4.0–6.0
Basin (6)	82.1	13.7–20.5	0.7–1.0	4.0–6.0
Basin (7)	52.3	13.0–26.1	1.1–2.2	2.0–4.0

2.3. Evaluation of NBS performance and potential of reclaimed water reuse in agricultural irrigation

This research focused on the main parameters considered by the regulation (EU) 2020/741 and the new Italian DPR, namely *E. coli*, BOD_5 and TSS. In addition to the mentioned parameters, TN and TP were also monitored since nutrients are essential elements for plant growth, and they are present in the reclaimed water that is used as an irrigation source. Nutrients can also be the cause of eutrophication, which led the Italian regulation to limit their content in reclaimed water when it is used in agricultural irrigation (Decree of the President of the Republic, 2023).

As shown in Table 3, the regulation (EU) 2020/741 identifies four reclaimed water quality classes (A, B, C, D), depending on the intended agricultural use (crop category) and the adopted irrigation method (Regulation (EU) 2020/741, 2020), e.g., water quality class A refers to food crops whose edible parts are in direct contact with reclaimed water and are consumed raw, and all irrigation methods are allowed.

NBS effluent concentrations were used to compare the observed values with European and Italian regulations to determine if reclaimed water was suitable for agricultural irrigation. The removal performance

Table 3
European Union and Italian regulation on reclaimed water reuse in agricultural irrigation.

Regulation	EUROPEAN UNION				ITALY				
Parameter									
	(EU) 2020/741				M.D. 185/03	DPR 2023			
	Water quality classes				***				
	A	B	C	D		A	B	C	D
E. coli (CFU 100 mL^{-1})	≤10	≤100	≤1,000	≤10,000	≤10	≤10	≤100	≤1,000	≤10,000
BOD_5 (mg L^{-1})	≤10	≤25			≤20	≤10	≤25		
TSS (mg L^{-1})	≤10	≤35 (more than 10,000 PE) or ≤ 60 (2,000–10,000 PE)			≤10	≤35 (more than 10,000 PE) or ≤ 60 (2,000–10,000 PE)			
TN (mg L^{-1})	–	–	–	–	≤15	≤15			
TP (mg L^{-1})	–	–	–	–	≤2	≤2			

of the systems was evaluated using removal efficiency, which was calculated using the average influent and effluent concentrations of those parameters with enough data, i.e. *E. coli* (case study A), TSS, TN and TP. Due to the limit of detection at 5 and 10 mg L^{-1} for the analysis of BOD_5, it was not considered appropriate to calculate the removal efficiency, as it would not show the real performance of the system. The mass load removal per unit of surface area (g m^{-2} $year^{-1}$) was calculated for those parameters that showed a positive value for the removal efficiency. Furthermore, for some parameters (e.g., TN), the effluent concentration in both the NBS was influenced by seasonality.

2.4. Estimation of first-order removal rate coefficients for total nitrogen

The design equation chosen to estimate output concentrations in the LS was based on the hydraulic model Tanks In Series (TIS), an adaptation of a completely stirred tank reactor in series (Kadlec and Wallace, 2009). TIS has been widely used to model and design different types of CWs and has proven to model their performance even under unsteady-state conditions (Canet-Martí et al., 2022). As suggested in recent studies (Von Sperling et al., 2022, 2023), in the present work, the hydraulic and biokinetic models were decoupled, and the volumetric and areal removal rate coefficients (k_V and k_A) were calculated using Eq. (1) and Eq. (2).

$$k_V = \frac{N\left(\left(\frac{(C_i - C^*)}{(C_e - C^*)}\right)^{(1/N)} - 1\right)}{\tau} \quad (1)$$

$$k_A = k_V \times h \times e_V \quad (2)$$

where k_V and k_A represented the volumetric and areal removal rate coefficients expressed in d^{-1} and m d^{-1}; C_i and C_e were the influent and effluent concentrations (mg L^{-1}); C^* was the background concentration (mg L^{-1}), a minimum concentration in the system from internal sources that cannot be removed, e.g. the degradation of microbial communities, animals and plants (Kadlec and Wallace, 2009); N was the number of tanks in series (NTIS) (dimensionless); τ was the theoretical HRT (d) ($\tau = V \times e_v/Q$); V was the total volume (m^3); e_V was the effective volume ratio (dimensionless); Q was the flow rate (m^3 d^{-1}); and h was the water depth (m).

The temperature of each period was calculated as the average air temperature from days prior (equal to HRT for each date) to the effluent sampling. Average k values were calculated for temperature ranges of 0–5, 5–10, 10–15, 15–20, 20–25 and 25–30 °C. The resulting k values were fitted to Arrhenius equation ($k_T = k_{20} \cdot \theta^{(T-20)}$), where k_T was the average k value at different temperatures to find the areal and volumetric removal rate coefficient at 20 °C ($k_{A,20}$; $k_{V,20}$) and the temperature correction factor (θ).

Both systems use treatment compartments where the effluent water temperature can be balanced with air temperature, creating distinct treatment regions, especially noticeable in wetlands with long retention times (days) (Kadlec and Wallace, 2009). In this study, we assumed treatment temperature equals the daily mean air temperature, as it typically falls between daily maximum and minimum water temperatures. This approach yields a useable k-value for designing wetlands without precise water temperature data. In extremely cold climates with potentially freezing top wetland layers, calculating the annual water temperature cycle is recommended (Kadlec, 1994).

To calculate k_{20} and θ for TN removal, data from case study B were used because it had the largest dataset, with 55 data between 2017 and 2022. The data selected corresponded to those for which both inlet and outlet concentrations and flow rates were available. Then, the calculated k_{20} and θ were tested in case study A, using a dataset of 17 data.

The model parameters were calculated only for TN because it had the most complete dataset. For the other parameters, the same methodology could not be applied because there were not enough input and/or output data or they were not accurate, i.e., there were no data below the limit of detection. This is a representative case where it is not recommended to apply the model because its parameters could not be used for the design or optimization of other systems.

In order to evaluate how well the model parameters fitted the observation data, the TN effluent concentrations were predicted for both case studies using $k_{A,20}$, $k_{V,20}$ and θ, and were compared with the measured TN effluent concentrations to test the performance of the NTIS model. The root mean squared error (RMSE) and R^2 were chosen as the goodness-of-fit measure to compare predicted data with real data points (Ahnert et al., 2007). The background concentration was considered constant in order to minimize the propagation of uncertainty, as this parameter is difficult to predict for every event (Canet-Martí et al., 2022). C^*, N and e_v were adjusted to minimize RMSE.

3. Results and discussion

3.1. Evaluation of quality parameters

3.1.1. Biochemical oxygen demand

The regulations on wastewater reuse consider the organic matter content in reclaimed water through the BOD_5 parameter. The provision of the correct organic matter content to soil where reclaimed water is applied is of fundamental importance since soil organic matter can influence the availability of water and nutrients for crops, enhance carbon sequestration, provide a measure of resilience against diseases and plagues, and ensure crop production (Cavicchioli et al., 2019; Erisman et al., 2016). The effect of the soil organic matter content on the nutrient balance is tricky since it causes nutrient-soil bonding. When organic matter decomposes, nutrients become available; thus, the use of fertilizers can be reduced. However, when nutrients are released while the crop does not need them, they are lost and can lead to environmental pollution (Li et al., 2022).

For case study A, during the monitoring period, the BOD_5 content in CW effluent was constant over time (mean value of 10 ± 1 mg L^{-1}), with slight differences from the measured content in the CW influent (mean value of 9 ± 3 mg L^{-1}) (Fig. 3). As observed in Fig. 3 and as explained in section 2.2., the limit of detection of the analytical methods was 5 or 10 mg L^{-1} for BOD_5. Therefore, it was not possible to see to what extent the systems were capable of removing BOD_5. It is very likely that the average was lower than the value of 10 mg L^{-1}. However, reclaimed water was certainly suitable for water reuse in agricultural irrigation as water quality classes from B to D, since the mean value for BOD_5 was lower than 25 mg L^{-1}.

For case study B, data on the BOD_5 concentration in the WWTP effluent (i.e., the LS influent) were not available. During the monitoring period, BOD_5 concentration in the LS effluent had a mean of 11 ± 2 mg L^{-1} (Fig. 3). As for case study A, the limit of detection was 10 mg L^{-1}; thus, the observed mean concentration was not lower than that. For case study B reclaimed water was also suitable for water reuse in agricultural irrigation as water quality classes from B to D.

3.1.2. Total suspended solids

In NBS, the main mechanisms involved in the TSS removal from wastewater are sedimentation and, in CWs also filtration, due to the presence of the substrate that can filter the wastewater. In CWs, biofilm formation over the filter media also supports TSS removal, as the biofilm adsorbs colloidal and particulates that can be metabolized and converted into soluble compounds (Zhou et al., 2020).

With the view of water reuse in agricultural irrigation, a higher TSS content can affect the correct functioning of the irrigation system (particularly drip irrigation systems) and compromise water quality. Usually, TSS levels less than 50–100 mg L^{-1} are considered safe for drip irrigation (Regulation (EU) 2020/741, 2020). On the contrary, a higher TSS content can lead to the clogging of the irrigation system (e.g., blockage of sprinklers or other emitters, interference with water flow in irrigation pipe networks, etc.), lowering irrigation performances and

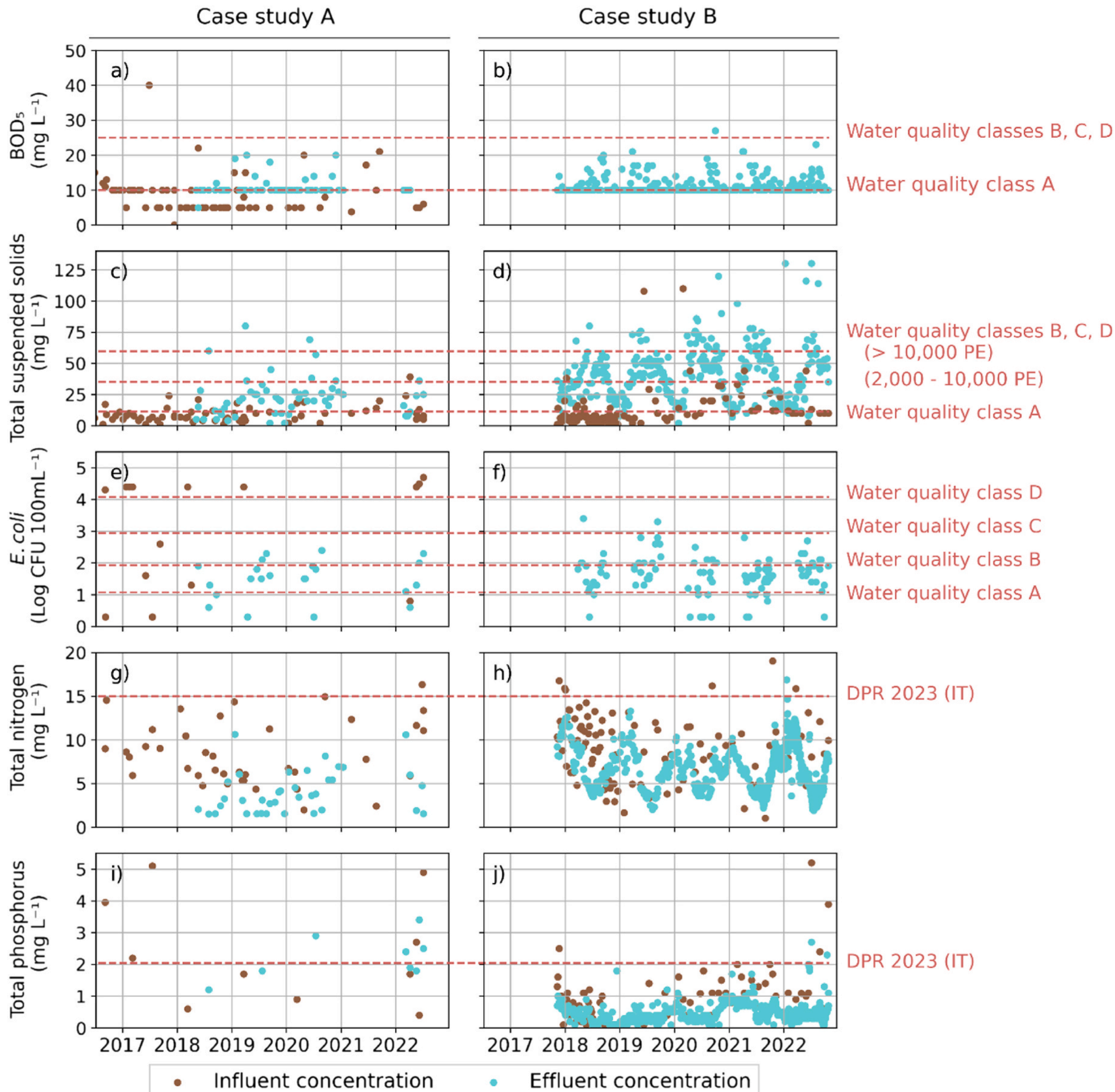

Fig. 3. Influent (where data were available) and effluent concentrations of BOD_5 (a, b), TSS (c, d), *E. coli* (e, f), TN (g, h) and TP (i, j) (mg L^{-1}) in case study A and case study B.

causing water stress of the not-properly irrigated crops. Furthermore, excessive amounts of TSS can cause soil plugging in crop fields, hindering the correct infiltration of irrigation water into the soil. TSS in the irrigation water can also have negative effects on crop growth. Indeed, TSS lead to higher concentrations of dissolved salts in the soil, which increases water osmotic potential, meaning that plants need more energy to take up water from the soil. As a result, plant respiration is increased, and plant growth and yield can progressively decline.

For case study A, during the monitoring period, the TSS content in CW influent was low, around 10 mg L^{-1}, indicating that the treatment units of the WWTP could reduce the TSS concentration to a minimum value. Surprisingly, an increase in TSS concentration in the CW effluent (22 ± 5 mg L^{-1} as mean value) was observed (Fig. 3), although a sedimentation basin at the beginning of the CW was used. It is noteworthy that TSS production can occur in CWs, especially in FWS wetlands, due to sediment resuspension, the death of microbes, fragmentation, detritus from plants, and the formation of chemical precipitates (Vymazal et al., 1998). The TSS increase can also be explained by the growth of algae, which can involve the deterioration of the water quality (Šereš et al., 2021). Finer filter media in HF wetlands could improve TSS removal as long as it is not too fine to clog (Zidan et al., 2015). Thus, such an approach should be considered if the goal is to have CW effluent of water quality class A. However, the CW effluent was suitable for water reuse in agricultural irrigation as water quality classes from B to D.

The same trend was observed in the LS of case study B. In fact, an increase from the mean influent to effluent TSS concentration from 15 ± 4 mg L^{-1} to 40 ± 21 mg L^{-1} (Fig. 3) was observed. It was noticed that TSS concentration in the effluent was higher in summer than in winter, following a sawtooth trend throughout all the monitored periods. This was probably due to algae and vegetation (e.g., *Lemna minor*, *Phragmites australis*, etc.) life cycle. The LS effluent was not suitable for water reuse in agricultural irrigation since TSS mean concentration was higher than the threshold value of 35 mg L^{-1} imposed by the reuse regulation.

To reduce the TSS effluent concentration, it is advisable to decrease water depth and introduce vegetated areas to increase hydraulic efficiency, resulting also in the decrease of flow velocity and TSS detainment within the matrix formed by vegetation. This can also reduce the surface area available for a biofilm layer to form, potentially increasing pathogen removal. Moreover, a deeper sedimentation zone at the beginning of the system can also enhance TSS removal.

3.1.3. Escherichia coli

Wastewater can contain human, animal and plant pathogens that can cause viral, bacterial, or parasitic infections (Godfree and Farrell, 2005). There are several routes whereby pathogens can affect human health, including direct contact, contamination of food crops, zoonoses, and vectors (Godfree and Farrell, 2005). The type and content of pathogens in municipal wastewater can differ depending on e.g., the level of endemic disease in the community and the presence of discharges from hospitals or commercial activities (Carraro et al., 2016). Most pathogens in wastewater can survive in the environment long enough to be conveyed to humans through contact with wastewater or consumption of contaminated food irrigated with wastewater (Ungureanu et al., 2020). In FWS wetlands and LSs, mechanisms of inactivation and pathogens removal are a combination of physical (filtration and sedimentation), biological (predation and natural die-off) and chemical (oxidation and disinfection by exposure to sunlight) processes.

The measured *E. coli* concentrations of influent and effluent of both case studies are shown in Fig. 3, and they were expressed as Log CFU 100 mL^{-1}. In case study A, the mean concentration of *E. coli* in the WWTP effluent (indicated as CW influent in Fig. 3) was 13,900 $\pm$ 3000 CFU 100 mL^{-1}. This value was higher than the water quality class D limit (10,000 CFU 100 mL^{-1}) imposed by the European and Italian regulations. High *E. coli* concentrations in the WWTP effluent were the main reason for the CW implementation, and the design values have considered seasonal variations and treatment efficiencies. As shown in Fig. 3, in the CW effluent, *E. coli* concentration was always lower than 10,000 CFU 100 mL^{-1}, with a mean value of 62 $\pm$ 22 CFU 100 mL^{-1}, meaning that reclaimed water was suitable for agricultural irrigation as water quality classes from B to D. The removal efficiency was of 58%, when influent and effluent data were available. A disinfection unit is required after the CW to further reduce *E. coli* concentration and meet quality water quality class A. It is important to mention that disinfection might lead to increased costs due to the unit management (Chhetri et al., 2014), and, therefore, it should be evaluated based on the long-term goals and effluent uses.

For case study B, data on the *E. coli* concentration in the WWTP effluent (i.e., LS influent) were not available since (Fig. 3), according to the current Italian regulation [26], it is mandatory to measure the *E. coli* concentration only where reclaimed water is discharged (in this case after the LS), and, for the specific case of the Emilia-Romagna region, only during the period of the year from April to September. The *E. coli* effluent concentration of the LS was always lower than 10,000 CFU 100 mL^{-1}. However, the mean value was 137 $\pm$ 27 CFU 100 mL^{-1}, which was higher than 100 CFU 100 mL^{-1} (minimum required for water quality class B), indicating that the effluent of the LS was suitable only for water quality classes C and D. As for case study A, a disinfection unit would also be required to ensure a water quality for class A and B.

In both case studies, the high *E. coli* concentrations might also be due to the high TSS concentration that was discussed in the previous section 3.1.2. Indeed, TSS increases the surface area where some components can adhere, e.g. pathogens, or it can shed them from solar radiation. Therefore, enhancing TSS removal is also recommended to reduce pathogens.

3.1.4. Total nitrogen

When wastewater is treated in WWTP, its discharge is also regulated in terms of nitrogen. The reason behind this is that reclaimed water with a high TN content can lead to a significant load of this nutrient to surface water (Smith and Siciliano, 2005) and groundwater (Mas-Pla and Menció, 2019), involving a decrease in water quality (Scanlon et al., 2007), promoting eutrophication (Huang et al., 2017), and altering biodiversity (Sutton et al., 2014). However, in WWTPs, the traditional biological methods that are commonly used to remove TN require aeration, which accounts for nearly half of the total energy consumption of WWTPs (Keene et al., 2017). In NBS, the mechanisms for TN conversion and removal can be attributed to various physical, chemical and biological processes, including ammonia volatilization, adsorption, desorption, plant uptake, ammonification, nitrification, denitrification, nitrogen fixation, etc. (García et al., 2010). Nitrification and denitrification are the main biological processes that are involved in TN removal in wetlands (Dong and Sun, 2007). During nitrification, ammonia is converted to nitrate through the nitrite intermediate under aerobic conditions (Biswal et al., 2021), and the process is facilitated by ammonia oxidizing bacteria (e.g., *Nitrosomonas*) and nitrite oxidizing bacteria (e.g., *Nitrosospira* and *Nitrospira*) (Wang et al., 2016). During denitrification, nitrate is reduced to nitrogen gas when dissolved oxygen is limited, degradable carbon is available, and denitrifying bacteria switch from aerobic to anaerobic respiration (Verduzo Garibay et al., 2021).

As mentioned above, TN and its organic/inorganic forms (e.g., NH_4^+-N, NO_3^--N and NO_2^--N) do not have threshold values imposed by the regulation (EU) 2020/741. On the contrary, the Italian one regulates TN content considering a limit of 15.0 mg L^{-1}, as reported in Table 3. In addition to comparing the observed values for TN, other forms of N (NH_4^+-N, NO_3^--N and NO_2^--N) were also monitored to understand better the TN removal mechanisms (e.g., nitrification/denitrification process). Moreover, it was also evaluated whether the nitrogen load of the reclaimed water was satisfying crop nutritional needs.

The mean removal efficiencies of TN were 66% and 49% for case studies A and B, respectively. Similarly, the mass load removals per unit of surface area were 347 and 439 g N m^{-2} $year^{-1}$, with similar values to those in the literature (Kadlec, 2012; Rizzo et al., 2023). The measured TN influent and effluent concentrations for both case studies are shown in Fig. 3. For case study A, the CW was designed to treat a WWTP effluent with a maximal TN concentration of 31.6 mg L^{-1} during winter and 16.8 mg L^{-1} during summer. However, during the monitoring period, the mean TN concentration in CW influent was 10.1 $\pm$ 3.4 mg L^{-1}, much lower than the design values, and TN effluent concentration was 3.4 $\pm$ 2.4 mg L^{-1}. Therefore, reclaimed water was suitable for reuse since the mean effluent concentration was lower than 15.0 mg L^{-1} imposed by the Italian regulation. Mean TN influent and effluent concentrations in the LS for case study B were 12.4 $\pm$ 4.1 mg L^{-1} and 6.3 $\pm$ 3.0 mg L^{-1}. For case study B, the effluent concentration was lower than 15.0 mg L^{-1}, thus, reclaimed water was suitable for agricultural irrigation.

The different TN removal efficiencies in case studies A and B might be due to the different geometry and water depth of NBSs. In case study A, the CW had an elongated shape, which improved hydraulic efficiency, whereas in case study B, the oval shape of the basins in the LS was probably the reason for dead zones and shortcuts (Nan et al., 2023). Likewise, water depth plays an important role in TN removal. The shallower the system, the more oxygen in the water column, which enhances complete nitrification. In fact, it is recommended to have a water depth of less than 0.5 m in order to ensure oxygen exchange (Langergraber et al., 2019). In both case studies, TN and NO_3^--N removal occurred simultaneously, indicating that nitrification-denitrification was the main removal mechanism, and both systems had anoxic conditions in their water column, probably in the deeper water levels. Moreover, the observed seasonal sawtooth trend in TN and NO_3^--N concentrations demonstrate that they were temperature-dependent. A seasonal increase in temperature resulted in a higher removal efficiency (Fig. 4). These results were consistent with those of previous studies showing that nitrifying bacteria were sensitive to temperature (Dong et al., 2011; Langergraber et al., 2019; Wang et al., 2021).

On the other hand, the presence of NO_2^--N in NBS effluents indicated that denitrification was incomplete. This happens when there is insufficient organic matter in the anoxic zone, conditions change from anoxic to aerobic, or HRT is too low to complete denitrification (Narkis et al., 1979). Although HRT in both NBS was assumed to be high enough for nitrification-denitrification to occur, especially in case study B, where the average HRT was approximately 18 days, the results indicate that it might not be completely true. Assuming that conditions remained

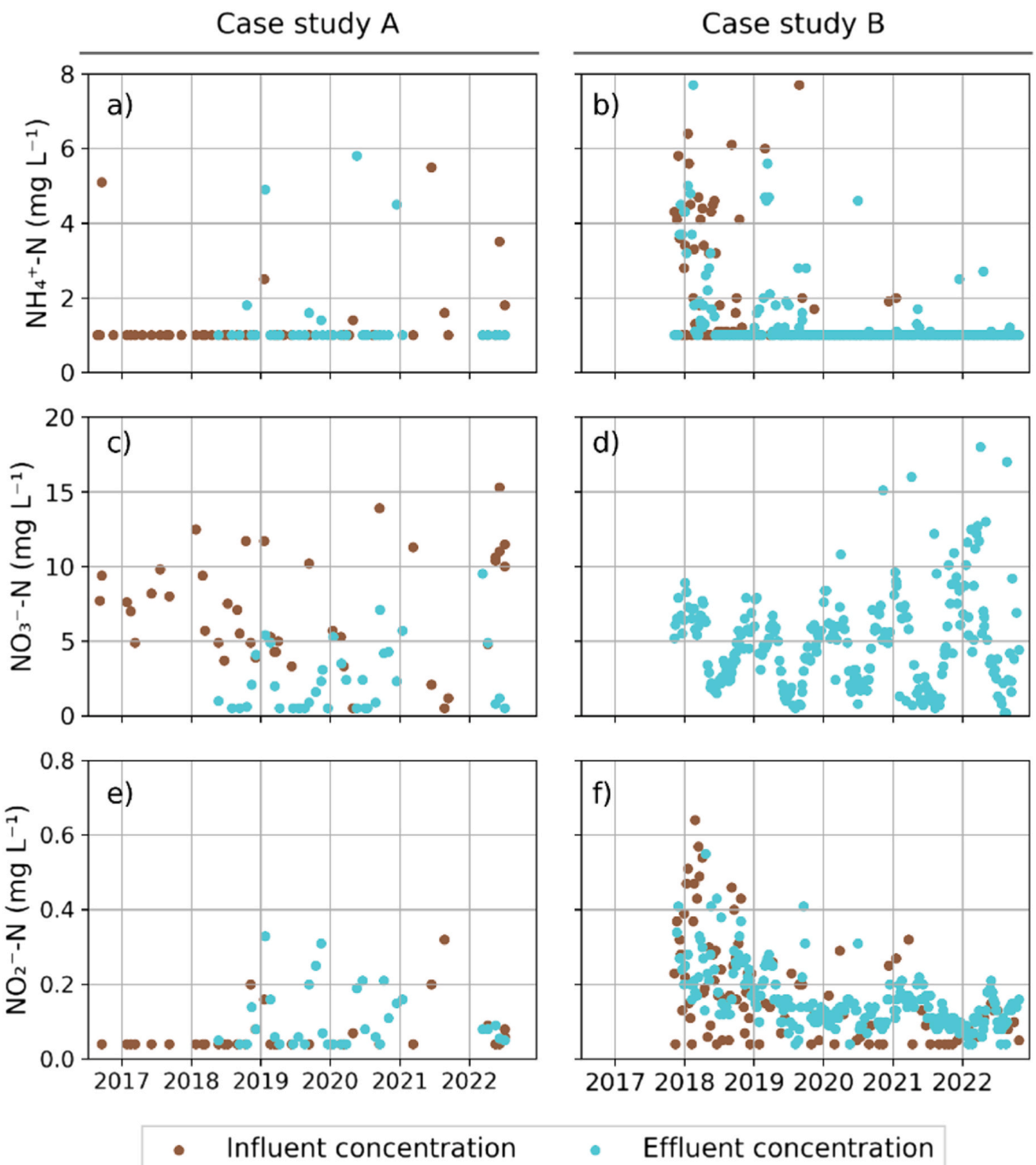

Fig. 4. Influent (where data were available) and effluent concentrations of NH_4^+-N (a, b), NO_3^--N (c, d) and NO_2^--N (e, f) (mg L^{-1}) in case study A and case study B.

anoxic throughout the wetland, the reason for the incomplete denitrification could be the lack of carbon supply (Kadlec, 2012). In order to increase carbon availability for complete denitrification, the addition of an organic-based substrate is recommended (Wang and Chu, 2016).

3.1.5. Total phosphorous

As for TN, when wastewater is treated in WWTPs, its discharge is also regulated in terms of TP since this nutrient can also cause eutrophication and other water quality issues in the ecosystem (Preisner et al., 2020). As already said, concerning the reuse of reclaimed water in agricultural irrigation, the new regulation (EU) 2020/741 has not established any limitation for the TP content, while the Italian regulation has imposed the limit of 2.0 mg L^{-1} as reported in Table 3.

Fig. 3 shows the TP effluent and influent concentrations in NBS of case studies A and B. For case study A, TP effluent and influent mean concentrations were very similar 2.0 ± 1.2 mg L^{-1} and 2.2 ± 0.7 mg L^{-1}, respectively, although both of them were quite variable over time. Therefore, WWTP effluent was suitable for agricultural reuse, but that was not the case with CW effluent, with a concentration higher than 2.0 mg L^{-1}. To solve the underlying problem, it would perhaps be appropriate to implement additional treatment techniques in the upstream WWTP for the enhanced TP precipitation. The LS of case study B was more effective in reducing the TP, with mean influent and effluent concentrations of 0.8 ± 0.2 and 0.4 ± 0.3 mg L^{-1}, and it was thus suitable for agricultural reuse.

The total mass load removal per unit of surface area for case study B was 31 g P m^{-2} $year^{-1}$. The mean removal efficiency in case study B was 43%, while in case study A there was no removal efficiency, indicating that the CW was not capable of removing TP from reclaimed water. The high variability of TP removal efficiency showed the complexity of removing this compound from wastewater, which depends on the interaction of different processes. The main TP removal mechanisms are adsorption-precipitation based reactions, which depend on the media's redox conditions and sorption capacity, and plant uptake if biomass is harvested (García et al., 2010; Vymazal, 2007).

TP can be adsorbed into the suspended solids and the filter media, or it can form inorganic precipitates and settle in the bottom of the system. This makes it dependent on the availability of solids and sorption surface and the hydraulic retention time required for these reactions to occur (Dunne et al., 2012). In the case of the LSs or FWS wetlands, if there are occasional inputs with higher flow rates, the particles may be lifted, and phosphorus accumulated in the past is flushed out of the system (Lavrnić et al., 2020). For this reason it is recommended to make a deeper zone in the first meters of the system, followed by a shallow and vegetated bed, to reduce the solids carryover at the outlet (Vymazal and Dvořáková Březinová, 2018). The addition of plant growth-promoting rhizobacteria (PGPR) have showed also beneficial results in terms of TP removal (Ji et al., 2021). In HF wetlands, TP that can be adsorbed in the substrate surface tends to decrease over time, until the substrate is saturated. For example, some researchers have suggested the use of sand instead of gravel as CW substrate, in order to increase the TP retention capacity due to the considerable differences in surface area of these two substrates (Brix et al., 2001; Dong et al., 2020). The use of reactive media with higher sorption capacities would also improve TP removal, e.g., light-expanded clay aggregate (Mlih et al., 2020). However, both options have a limited capacity and it would only delay the media saturation. Neither in case study A nor case study B, it was possible to state that adsorption was the predominant removal mechanism, because this tendency was not observed. However, the fact that both NBS were inefficient in retaining TP may be beneficial. This aspect is also important from the point of view of agricultural production. Being a limited resource, if treated water contains phosphate, the amount of fertilizers can be reduced, costs can be lowered, as well as the dependence on the availability of P-rock reserves (Schoumans et al., 2015).

Unlike TN removal, seasonality in the performance of both NBS for TP removal was not observed. This is because microbial uptake is considered only as temporary storage of TP with a very short turnover rate. Indeed, TP that is assimilated by microbiota is soon released back into the wastewater after the decay of the organisms (Vymazal, 2007), resulting in the increase of TP content in NBS effluents. In addition, the concentration of TP in rainwater could, in certain cases, exceed that in the influent, which increases the concentration of this element in NBS and makes it more difficult to observe trends in removal efficiency (Lavrnić et al., 2020).

3.1.6. *Summary of the treatment performances and final considerations*

Table 4 summarizes the mean concentrations observed for the parameters relevant to agricultural reuse, namely *E. coli*, BOD_5, TSS, TN and TP, in both case studies, which have been discussed in the previous sections. The results show a satisfactory overall performance of the two NBS as tertiary municipal wastewater treatment. However, as recommended, some modifications are necessary to improve the removal efficiency of both NBS and comply with the new regulations on reclaimed water reuse.

In case study A, BOD_5, TSS and *E. coli* exceed the threshold for water quality class A, but the effluent is suitable for water quality classes B to D. Moreover, comparing the monitored data with the Italian regulation, TN content was lower than the limit imposed, but TP content slightly exceeded the allowed maximum concentration.

In case study B, similarly, data on BOD_5 and *E. coli* showed that effluent was suitable for water quality classes from B to D. However, it is noteworthy that reclaimed water had a TSS content higher than the threshold value imposed by the regulation, making the reclaimed water unsuitable for agricultural reuse. As per the Italian regulation, the effluent was suitable in terms of TN and TP content.

In order to achieve higher water quality, modifications have to be made to increase the removal of contaminants that were above the limits provided by the regulation. A disinfection unit would be necessary to ensure that *E. coli* is below 10 CFU 100 mL^{-1} (water quality class A) in both case studies. Regarding BOD_5, the mean effluent concentration was probably below 10 mg L^{-1}, as explained above. However, the regulation sets the limit as the maximum weekly detected, not as an average. As for TSS content, the reclaimed water regulations impose that the effluent concentration must be lower than 10 mg L^{-1} for water quality class A, while for water quality classes from B to D, higher values up to 35 and 70 mg L^{-1} are permitted. Decreasing the water depth and adding vegetated islands would improve hydraulic efficiency and likely increase removal efficiency (Nan et al., 2023). Although reclaimed water reuse is a sustainable way of recovering nutrients (both TN and TP) for agricultural irrigation, nutrient pollution and eutrophication of water resources should be avoided by reducing their content in the reclaimed water. On the other hand, when reclaimed water is reused in agriculture, it is advisable not to target maximum nutrient removal to have reclaimed water based on the nutritional needs of individual crops.

Table 4
CW and LS contaminants concentration. Case study A: mean values for the years of operation 2016–2022, total number of wastewater samples 859 (influent 493 + effluent 366); Case study B: mean values for the years of operation 2017–2022, total number of wastewater samples 4358 (influent 727 + effluent 3631).

Parameter	Case study A			Case study B		
	WWTP influent	CW influent	CW effluent	WWTP influent	LS influent	LS effluent
E. coli (CFU 100 mL^{-1})	–	13,900 ± 3000	62 ± 22	–	–	137 ± 27
BOD_5 (mg L^{-1})	176 ± 45[a]	9 ± 3[a]	10 ± 1[a]	939 ± 295[a]	–	11 ± 2[a]
TSS (mg L^{-1})	179 ± 40	10 ± 3	22 ± 5	1225 ± 495	15 ± 4	40 ± 21
TN (mg L^{-1})	44.9 ± 13.7	10.1 ± 3.4	3.4 ± 2.4	72.4 ± 22.4	12.4 ± 4.1	6.3 ± 3.0
TP (mg L^{-1})	5.8 ± 2.1	2.0 ± 1.2	2.2 ± 0.7	16.0 ± 9.7	0.8 ± 0.2	0.4 ± 0.3

[a] Value considering that the limit of detection for BOD_5 was 10 mg L^{-1}.

3.2. *Removal rate coefficients for TN removal*

The removal rates at 20 °C (k_{20}) for k_V and k_A were 15 m $year^{-1}$ and 60 m $year^{-1}$ for case study B. These values were similar to those reported in the literature for TN removal in FWS wetlands (Kadlec and Wallace, 2009). Since the water depth of the LS was higher than 1.0 m (i.e., 4.2 m), k_A was also higher than k_V. Fig. 5 shows the average calculated k values and the predicted values calculated with the Arrhenius equation to find k_{20} and θ. The temperature correction factor θ was 1.075, indicating that TN removal increased with temperature as expected.

For case study B, the goodness-of-fit measure (i.e., RMSE and R^2) showed that the model could give a good estimate of TN effluent concentrations using $k_{A,20}$ and $k_{V,20}$ when N = 8, C* = 2 mg L^{-1} and e_v = 0.96 (RMSE = 1.09 mg L^{-1}; R^2 = 0.81). However, e_v seems too high, considering that the LS does not have an elongated shape or obstacles to improve its hydraulic performance (Persson and Wittgren, 2003). Although some researchers recommend using zero as background concentration to estimate more conservative k values for design (Von Sperling et al., 2022), the authors preferred to use 2 mg L^{-1} because it showed the best fit for both NBS systems.

k_{20} and θ, previously calculated using the dataset from case study B, were tested in case study A, with a dataset of 17 dates with influent and effluent concentrations and flow rates. The lowest RMSE value (i.e., 1.17 mg L^{-1}) was achieved using the same value of $k_{A,20}$ (i.e., 60 m $year^{-1}$), C* = 1.5 mg L^{-1}, N = 12, and e_v = 0.86. $k_{V,20}$ was 67 $year^{-1}$, considering the water depth. Overall R^2 in case study A was 0.86,

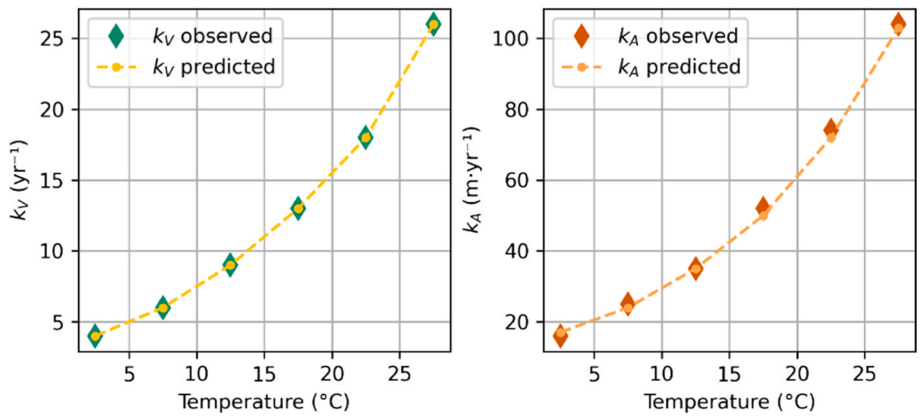

Fig. 5. Average values of volumetric and areal removal rate coefficients (k_V and k_A) at different temperatures and simulated values using the Arrhenius equation for Case study B.

showing a good performance of the calculated K_{20}-values. The results showed that the use of the areal removal rate coefficient ($k_{A,20}$) is more appropriate than the volumetric one, and it can be used to design systems with similar wastewater characteristics.

3.3. Potential of reclaimed water reuse in agricultural irrigation

For case study A, the NBS effluent was discharged into the irrigation channel, and after mixing with surface water, it was later used to irrigate crops. The surrounding area is mainly cultivated with the Protected Geographical Indication (PGI) muskmelon (*Cucumis Melo*), belonging to the Cucurbitaceae family, producing considerable benefits for the area due to its high quality. In 2022, the cultivated area with muskmelon near the San Matteo della Decima WWTP was around 18 ha ("AGREA," 2022). This area was irrigated, from March to July, with water abstracted from the irrigation channel (the same where the reclaimed water was discharged).

As reported in Table 5, during the irrigation season (e.g., from March to July) and the entire monitoring campaign (from 2017 to 2022), the average reclaimed water (RW) volume was 1.8×10^3 m^3 year^{-1} (min 1.2×10^3 m^3 year^{-1}, max 3.6×10^5 m^3 year^{-1}). These volumes were used to evaluate the potential of reclaimed water to satisfy crop water and nutrient needs. Due to the small treatment capacity of the San Matteo della Decima WWTP, the average reclaimed water (RW = 1.8×10^3 m^3 year^{-1}) volume was lower if compared to the muskmelon irrigation water requirement (IWR = 4.8×10^4 m^3 year^{-1}), representing

Table 5
Potential of reclaimed water to satisfy muskmelon water and nutrient needs: a) calculation of muskmelon water and nutrient needs; b) evaluation of potential volume and nutrient content of reclaimed water; c) capability of reclaimed water to satisfy muskmelon water and nutrient needs.

a) Muskmelon water and nutrient needs						
		IWR m^3 year^{-1}	TN kg ha^{-1} year^{-1}	TP kg ha^{-1} year^{-1}	TN (18 ha) kg year^{-1}	TP (18 ha) kg year^{-1}
Case study A		4.8×10^4	120	39	2160	702
		IWR m^3 year^{-1}	TN kg ha^{-1} year^{-1}	TP kg ha^{-1} year^{-1}	TN (5205 ha) kg year^{-1}	TP (5205 ha) kg year^{-1}
Case study B		6.2×10^5	120	39	624,600	201,434
b) Volume and nutrient content in reclaimed water						
		RW m^3 year^{-1}	TN[b] mg L^{-1}	TP[b] mg L^{-1}	TN[c] kg year^{-1}	TP[c] kg year^{-1}
Case study A	mean[a]	1.8×10^3	3.4 ± 2.4	2.2 ± 0.7	6.1	4.0
	min[a]	1.2×10^3			4.1	2.6
	max[a]	3.6×10^5			1224.0 (163.2)	792.0 (105.6)
Case study B	mean[a]	1.1×10^7	6.3 ± 3.0	0.4 ± 0.3	69,300.0	4400.0
	min[a]	8.6×10^6			54,180.0	3440.0
	max[a]	1.4×10^7			88,200.0	5600.0
c) Capability of reclaimed water to satisfy muskmelon water and nutrient needs						
		RW/IWR (%)		N/N (18 ha) (%)		P/P (18 ha) (%)
Case study A	mean[a]	3.8		0.3		0.6
	min[a]	2.5		0.2		0.4
	max[a]	100.0		7.6		15.0
		TWW/IWR		N/N (5205 ha)		P/P (5205 ha)
Case study B	mean[a]	78.6		11.1		2.2
	min[a]	61.4		8.7		1.7
	max[a]	100.0		14.1		2.8

NOTES: [1] Nutrient content (in kg year^{-1}) in reclaimed water, considering that the maximum reclaimed water volume was equal to IWR.
[2] All the values that were reported above refer to the irrigation season (from March to July and for the entire experimental campaign).
[3] Nutrient content (in kg year^{-1}) in the reclaimed water volume that was strictly necessary to meet the muskmelon water needs.
[a] Mean, minimum and maximum reclaimed water volume (in m^3 year^{-1}) produced by the two NBS.
[b] Nutrient mean concentration (in mg L^{-1}) in NBS effluents.
[c] Nutrient content in reclaimed water in NBS effluents.

only 3.8% of the IWR volume. Therefore, in the case of direct reuse of reclaimed water, it is not enough to cover the complete muskmelon water needs, but it can limit the impact of water stress. On the other hand, considering the maximum reclaimed water volume, muskmelon water needs can be fully satisfied (RW = 3.6×10^5 m^3 $year^{-1}$ > IWR = 4.8×10^4 m^3 $year^{-1}$).

As for the nutrient needs, the Emilia-Romagna region recommends standard doses for TN and TP of 120 and 39 kg ha^{-1} $year^{-1}$, respectively, aiming at ensuring a muskmelon production of about 32–48 tonnes ha^{-1} $year^{-1}$ ("Regione Emilia-Romagna. Agricoltura, caccia e pesca," 2022). For case study A, TN and TP concentrations in the CW effluent were 3.4 ± 2.4 and 2.2 ± 0.7 mg L^{-1}, respectively; these values represented the average during the irrigation season and the entire monitoring campaign. In this period, the average volume of reclaimed water (RW = 1.8×10^3 m^3 $year^{-1}$) contained 6.1 and 4.0 kg $year^{-1}$ of TN and TP, respectively. For the same period, the total nutrients provided to muskmelon due to the use of fertilizers were 2160 and 702 kg $year^{-1}$ for TN and TP, respectively. Therefore, the direct use of reclaimed water would not have a high impact on the muskmelon nutritional needs (only 0.3% and 0.6% for TN and TP, respectively). However, if the maximum reclaimed water volume was considered, these percentages can rise up to 7.6% and 15.0%, respectively.

In case study B, the NBS effluent was discharged into the Santerno river; therefore, reclaimed water was not used for irrigation purposes. However, since Imola WWTP is bigger as compared to San Matteo della Decima WWTP, the produced reclaimed water (RW) volume (mean 1.1 $\times 10^7$ m^3 $year^{-1}$, min 8.6×10^6 m^3 $year^{-1}$, max 1.4×10^7 m^3 $year^{-1}$) was also higher during the irrigation season and the entire monitoring campaign. Hence, to evaluate the potential of reclaimed water to satisfy both crop water and nutrient needs, it was assumed that all the produced reclaimed water was used for the irrigation of the same crop typology (muskmelon) as in case study A, regarding the standard doses that were reported above (TN and TP of 120 and 39 kg ha^{-1} $year^{-1}$, respectively). It was estimated that the maximum produced reclaimed water (RW = 1.4×10^7 m^3 $year^{-1}$) was enough to satisfy the water needs of a 5205-ha muskmelon cultivation. This cultivated area was much higher if compared to the cultivated area of case study A, also implying an increase in muskmelon irrigation water requirement (IWR = 6.2×10^5 m^3 $year^{-1}$).

For case study B, TN and TP concentrations in the LS effluent were 6.3 ± 3.0 and 0.4 ± 0.3 mg L^{-1}, respectively; as for case study A, these values represented the average during the irrigation season and the entire monitoring campaign. For the same period, considering the 5205-ha cultivated area, the total nutrients provided to muskmelon due to the use of fertilizers would have been 624,600 and 201,434 kg $year^{-1}$ for TN and TP, respectively. Therefore, the direct use of reclaimed water for case study B would have a comparable impact on the muskmelon nutritional needs as for case study A, with a 14.1% (for TN) and 2.8% (for TP) nutrient intake considering the maximum produced reclaimed water flow rate (RW = 1.4×10^7 m^3 $year^{-1}$).

These findings are in line with the recently published literature that states that NBS (if properly designed and implemented) can provide effluents with adequate nutrient content, capable of satisfying nutrient crop needs (Chavan and Mutnuri, 2021), especially when long-term reclaimed water irrigation is adopted (Bedbabis and Ferrara, 2018).

The results presented in this section represent a preliminary assessment that can be used as a starting point for further studies and long-term monitoring campaigns aimed at investigating different operational conditions and process parameters in NBS, useful for the removal of targeted contaminants and the maintenance of the proper nutrient content in reclaimed water when the latest is used in agricultural irrigation.

4. Conclusions

The use of reclaimed water in agricultural irrigation can lead to a reduction in freshwater demand, as well as a reduction in added fertilizers. Nature-based solutions (e.g., CW and LS) are environmental-friendly and cost-effective technologies for wastewater treatment and pollution control, but their capacity to produce effluents that are suitable for water reuse in agricultural irrigation still needs to be assessed, especially considering their long-term performance.

The present study investigated a hybrid CW and a LS, which treated effluents from the secondary treatment from two municipal wastewater WWTPs in Northern Italy. The effluents of the two NBS were continuously monitored during the multi-year experimental campaign (6–7 years) in order to determine compliance with the current regulations on minimum requirements for reclaimed water reuse. The outcomes of this research showed that the effluents of the NBS would need further treatment to reduce the concentration of monitored contaminants and to achieve the water quality required for water quality class A, (e.g., implementation of filtration and disinfection treatment units). Some system adaptations have been proposed to improve removal efficiency, which could significantly improve system performance to sufficiently good water quality for unrestricted reuse. Some of these measures are the implementation of vegetation islands in both systems, the addition of HF wetlands after the LS and the addition of a disinfection system after both systems to ensure a low concentration of *E. coli*. When designing new FWS wetlands or LSs, it is advisable to reduce the water depth to less than 1 m.

The research also showed that the use of reclaimed water would allow to satisfy water needs of the crops cultivated in the area, as well as a certain reduction of chemical fertilizers application, since nutrients (mainly nitrogen and phosphorous) that are in reclaimed water can help to meet these needs.

However, further research is needed to improve the removal efficiency of NBS and to facilitate their design. In this way, policymakers and local communities may become aware of the importance of these NBS for the provision of reclaimed water and other essential environmental services.

In addition, the monitoring activity allowed kinetic coefficients calculation for TN removal. Specifically, the temperature correction factor ($\theta = 1.075$) and areal removal rate coefficient ($k_{A,20} = 60$ m $year^{-1}$) were calculated from data in case study B and successfully tested in case study A. These coefficients can be used for the design of new NBS systems with similar characteristics in the tanks-in-series model and plug-flow model with dispersion since the data originate from a long-term monitoring campaign.

Author contributions

Conceptualization, G.M, S.L.; methodology, G.M, A.C.M. and S.L.; investigation, G.M.; data curation, G.M., A.C.M., A.Z.; writing—original draft preparation, G.M.; writing—review and editing, G.M., A.C.M., S.L., A.Z., F.A., G.L. and A.T.; supervision, A.T. All authors have read and agreed to the published version of the manuscript.

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Informed Consent Statement

Not applicable.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

The data that has been used is confidential.

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