



# Effects of Design and Operational Conditions on the Performance of Constructed Wetlands for Agricultural Pollution Control – Critical Review

Xi Nan · Stevo Lavrnić · Giuseppe Mancuso · Attilio Toscano

Received: 18 February 2023 / Accepted: 17 May 2023 / Published online: 29 June 2023  
© The Author(s) 2023

**Abstract** Constructed wetlands (CWs) can be considered as an efficient nature-based solution for the treatment of agricultural drainage water (ADW) and consequently for the mitigation of non-point source pollution. Aiming to provide suggestions for the construction and implementation of CWs, this paper proposes and discusses key parameters of CW design and operation. In order to verify the effect of these features, different case studies were reviewed, focusing on the performance of CWs that are treating agricultural drainage water. The findings showed that design and operational factors (e.g., the application of simple hydraulic structures and vegetation establishment) can improve pollutant removal efficiencies by increasing hydraulic retention time. Hydraulic efficiency of CWs can also be enhanced through certain shape characteristics (e.g., adoption of a high aspect

ratio and creation of a long and narrow CW shape). The careful consideration of these parameters before and during CW implementation can therefore help these systems to achieve their full potential. However, further study is recommended to assess the effects of some parameters (e.g., flow direction and the application of deep zones).

**Keywords** Nature-based solutions · Constructed wetland · Agricultural drainage water · Design and operational factors · Long-term operation · Agricultural pollution control

## 1 Introduction

ADW is one of the leading non-point sources of pollution (Braschi et al., 2022; Budd et al., 2011; Karpuzcu & Stringfellow, 2012). The transportation of agricultural pollutants causes water quality deterioration in the receiving water bodies (e.g., eutrophication) (Budd et al., 2011; Carstensen et al., 2019; Jayasiri et al., 2022; Johannesson et al., 2017) and increases human health risks (Díaz et al., 2012; Fida et al., 2022).

As a countermeasure, nature-based solutions are capable of managing non-point sources of pollution and attenuating the loads of agricultural contaminants through intercepting water flow (Díaz et al., 2010; Imfeld et al., 2013; Tournebize et al., 2015, 2017), especially suitable being CWs (Wu et al., 2023).

---

X. Nan (✉)  
Academy of Agricultural Planning and Engineering,  
Ministry of Agriculture and Rural Affairs, Beijing 100125,  
China  
e-mail: xi.nan@studio.unibo.it

X. Nan  
Key Laboratory of Farm Building in Structure  
and Intelligent Construction, Ministry of Agriculture  
and Rural Affairs, Beijing 100125, China

X. Nan · S. Lavrnić · G. Mancuso · A. Toscano  
Department of Agricultural and Food Sciences, Alma  
Mater Studiorum-University of Bologna, Viale Giuseppe  
Fanin 50, 40127 Bologna, Italy

Apart from the advantages of simple and low-cost operation (Beutel et al., 2013; Margalef-Marti et al., 2019), these ecological treatment systems can also provide diverse services, e.g., flood protection, wildlife habitat, groundwater recharge, aesthetic, and recreational values (Lavrnić et al., 2018; Lenhart et al., 2016; Maynard et al., 2011; McLaughlin & Cohen, 2013). Therefore, CW systems are an object of a growing interest and have been increasingly applied in agricultural landscapes worldwide (Abbassi et al., 2011; Dal Ferro et al., 2018; Djodjic et al., 2022; Préau et al., 2022; Takavakoglou et al., 2022).

To date, many studies have documented CWs as a promising method for reducing nutrients and pesticides concentration in agricultural runoff and drainage water (Brauer et al., 2015; Budd et al., 2011; Calvo-Cubero et al., 2014; Pavlidis et al., 2022). However, wide variability of pollutant removal efficiencies was observed in these systems (Crumpton et al., 2020; Díaz et al., 2012; O'Geen et al., 2010). For example, the reported removal efficiency of pesticides was in the range of 0–100% (O'Geen et al., 2010), while a similar ratio was found for total phosphorus (TP) (e.g., 3–80% measured in free water surface CWs (FWS CWs)) (Kill et al., 2018; Reinhardt et al., 2005). Most studies stated that N removal efficiency was generally within the range from 35 to 55% (Brauer et al., 2015), although some extreme values of nitrate removal efficiency (e.g., from negative values up to 98%) were also detected (O'Geen et al., O'Geen et al., 2010). The wide range of these pollutant retention efficiencies can be attributed to diverse factors, such as CW design and operation (e.g., location, hydraulic retention time (HRT) and vegetation characteristics), meteorological condition (e.g., climate, temperature), pollutant loading, seasonality, annual variations in water flow and dissolved oxygen concentration (Brauer et al., 2015; Kynkäänniemi et al., 2013). Furthermore, it is emphasized that some processes regarding nutrient removal may be temporary, such as nutrient uptake by plants and retention of Fe-bound P in the sediment (Margalef-Marti et al., 2019; Mendes et al., 2018a, b), which can also contribute to the variation of removal efficiencies. Finally, it was suggested that the treatment capacity of CWs was associated with their age (White, 2018).

Given the potential benefits of CWs for treating ADW and a lack of specific guidelines and related

research dedicated to design and operational factors (Ioannidou & Pearson, 2018; Soana et al., 2020), further information on how these parameters affect the performance of CWs is needed in order to maximize their removal efficiency. With this aim, an increasing number of research studies have been carried out worldwide on the implementation of CWs, even if these systems were often defined as not capable for efficient long-term operation period and so far relatively little is known about their long-term performance (Groh et al., 2015; Nilsson et al., 2020). Therefore, the long-term operational potential of these systems and its connection to design and operational factors are also something that needs further investigation.

In this context, the present work analyzed a series of field studies on the treatment of ADW, aiming to give better insight into the above-mentioned problems. The general objective of the present research was to identify the main design and operational parameters that can influence the performance of CWs and to analyze how these factors affect the treatment efficiency of the systems.

## 2 Materials and Methods

In order to have a better understanding on the relationship between design and operational factors and treatment performance of CWs treating ADW, a literature retrieval was conducted through the scientific database (e.g., Scopus, Web of Science). The field-scale studies were given priority since it was reported that only the implementation of field-scale CW treatment systems can precisely verify the actual systematic performance impacted by certain design parameters (White, 2018). This step—information collection—was used for the discussion in the following sections.

In Section 3, the authors summarized the primary considerations for the design and operation of CWs for the treatment of ADW. To this aim, different parameters (e.g., CW location, size, and hydraulic design; CW vegetation management) were analyzed. Meanwhile, through an overview of more than 40 case studies, the connection between the treatment performance of CWs and the main design and operation factors was discussed.

In Section 4, based on the field studies among the total introduced in Section 3, the authors analyzed the relationship between nitrogen removals and related factors (e.g., wetland-to-catchment ratio, average water depth) and discussed the treatment performance of some CWs after long-term monitoring.

It should be noted that most of the above studies considered on FWS CWs, as it is the most used type of CWs for treating water from the agricultural sector (Tournebize et al., 2017; Vymazal & Březinová, 2018). Their advantages were also commonly reported, such as the adaptability to flow rate variations and capacity to store and treat larger volumes of water (Land et al., 2016; Lavrnić et al., 2018).

### 3 Primary Design/Operational Factors and Their Effects

Below are listed the key points for CW design and operation that were frequently documented in the literature.

- Local climate condition and seasonal variation (Section 3.1)
- CW shape (Section 3.1)
- Wetland-to-catchment and length-to-width ratios of CWs (Section 3.1)
- Flow direction (Section 3.2)
- Hydraulic structures (Section 3.2)
- Configuration of inlet and outlet points (Section 3.2)
- Vegetation establishment (Section 3.3)
- Regular harvesting of CW vegetation (Section 3.3)

#### 3.1 Location and Size

Different characteristics, both related to the climatic conditions and the size and shape of the system, can have an effect on its performance, and this section will discuss these factors.

##### 3.1.1 Site and Local Condition

The location of a CW and local conditions were reported as the important prerequisites for design by Koskiaho and Puustinen (2019). Before determining a site where CWs should be built, detailed investigation

and assessment need to be done taking into account the features such as geology, topography, and soil layers, which were suggested as important for pollutant removal efficiencies (Lenhart et al., 2016; Tournebize et al., 2017). However, also site characteristics such as hydrology and climate conditions that influence it were proven to be of particular importance (Tanner & Kadlec, 2013).

Babbar-Sebens et al. (2013) developed a combined GIS-simulation-optimization-based approach for CW site identification, also able to optimize CW spatial coordinates to achieve the maximum peak flow reductions. Through this approach, they found 2953 potential CW sites in Eagle Creek Watershed, USA. These sites covered 1000 m<sup>2</sup>, equivalent to 1.5% of the watershed area, but they could capture ADW from a third of the entire watershed area. Similarly, in the Indiana watershed, USA, Kalcic et al. (2012) used a geospatial approach to determine the CW location. They found 18 potential sites for CWs, which could intercept 2.7% of ADW flows in the study area and remove 1.0% of all nitrates present in it, only accounting for 0.08% of the entire watershed area.

A study by Tournebize et al. (2013) evaluated the performance of two different CWs situated in-stream and off-stream. According to the two CW sites, the authors pointed out some interesting findings. For instance, the off-stream CW was considered as beneficial for biodiversity and had less land requirements for a similar pesticide removal compared to the in-stream CW. However, the latter one showed greater water storage potential for irrigation, especially during droughts.

Specially, the local conditions should be considered when siting CWs. For instance, a mesocosm study conducted in the Florida Everglades, USA, reported the importance of collecting local information before design and CW operation start (Mitsch et al., 2015). The authors pointed out that the studied wetlands should be operated for at least 2 years before the effective removal of TP can take place. It was due to the fact that TP concentrations in the effluents were constantly higher than the influent values during the first two monitoring years, affected by the reflux of labile phosphorus from the soils.

On the other hand, local climate factors can affect the treatment performance of CW systems. Steidl et al. (2019) monitored between October 2013 and May 2017 a FWS CW in north-eastern Germany

that served for mitigating nitrogen loads from ADW before discharging it into the river. The results showed that total nitrogen (TN) retention efficiency ranged from 0.2 to 8.9%, much lower than the values reported by other studies (Dal Ferro et al., 2018; Kadlec et al., 2010). It could be attributed to the influence of the inner-annual distribution of outflows from the local agricultural field. Specifically, the CW received a large proportion (>70%) of the annual nitrogen loads in winter, when temperatures lower than 10 °C limited plant uptake and microbial activities, resulting in low N reduction.

Similarly, in the study of Koskiaho and Puustinen (2019), a CW located in Finland was strongly affected by seasonal variations. Particularly, the performance of the CW from summer to autumn dominated water purification processes throughout the year, whilst winter efficiencies were extremely low. The author attributed this difference to the severe Finnish weather condition (e.g., mean winter temperature below 0 °C) and further proposed that if a CW of similar dimension were constructed in other regions in southern Europe, the effectiveness would likely be enhanced.

In China, Li et al. (2018) carried out 3-year seasonal (i.e., spring, summer, and autumn of each year) monitoring of a wetland restored from cropland. It was found that the pollutant concentrations from influent to effluent showed a reduction of 7.54–84.36% for TN and a reduction of up to 70.83% for TP. The lowest values of both pollutants were measured in the summer of 2016, which can probably be explained by the inadequate capacity of the wetland for purifying the excessive input loads that occurred in those months.

According to those findings, the effect of seasonal variability on water purification (e.g., seasonal rainfall) is thus advised to be taken into consideration when designing a CW. Additionally, in certain cases, it is also important to consider seasonal discharge regimes that are dependent on the farming activities in the area.

### 3.1.2 Constructed Wetland Size

Karpuzcu and Stringfellow (2012) found a way to calculate the wetland size needed for the target nitrate effluent concentration in the agricultural watershed. The key point of the approach was to determine the nitrate removal kinetic parameters, depending on the

results from field studies implemented in San Joaquin River Basin, USA, together with a laboratory microcosm study. This study demonstrated how important it was to estimate land area requirements for a target effluent concentration when designing a CW.

Similarly, Vallée et al. (2015) attributed the comparably lower nitrate removal of 5.4–10.9% they observed to the limited sizes of CWs. The CWs with larger surface area and consequently the larger ratio between the area covered by a CW and the catchment area (wetland-to-catchment ratio) can achieve greater treatment performance because of longer HRT (Moreno-Mateos et al., 2010; Steidl et al., 2019; Tournebize et al., 2017).

For example, the comparison of two Finish CWs done by Koskiaho and Puustinen (2019) showed that the one with a wetland-to-catchment ratio of 5% had much better annual removal efficiencies (74% for total suspended solids (TSS), 58% for TP and 54% for TN) than the one with a wetland-to-catchment ratio of 1.3% (7% for TSS, 12% for TP, and 9% for TN). Similar findings were obtained by Hoffmann et al. (2012). In their study, two restored riparian wetlands (Egeskov and Stor Å) in Denmark have been investigated for two successive seasons. The wetland-to-catchment ratio was reported to be 13.7% at Egeskov wetland higher than the ratio of 2.4% at Stor Å wetland. As expected, the load removal rates of TN were 43% and 75% at Egeskov higher than 32% and 26% at Stor Å in the first and second monitoring year, respectively. At the same time, similar nitrate values were also observed.

In the case study of Steidl et al. (2019) aforementioned, the wetland-to-catchment ratio was much smaller (0.4–0.5%), although it was the largest possible ratio according to the local conditions. Therefore, the TN removals were considerably lower compared to other studies, and they were less than 5% in most of the monitoring time.

However, the application of a reasonable wetland-to-catchment ratio for a satisfactory effect on water purification does not seem to be adequate (Moreno-Mateos et al., 2010). It is recommended to be considered together with other requirements, such as the shape, the aspect (length-to-width) ratio, and the depth of CWs.

Tournebize et al. (2017) and Kynkäänniemi et al. (2013) pointed out that the shape of CWs should be designed in order to be suitable for the landscape. For

instance, different shapes were applied such as the widely used rectangular (Ioannidou & Pearson, 2018; Song et al., 2019; Tournebize et al., 2017), trapezoidal (Dal Ferro et al., 2018), ellipsoidal (Maynard et al., 2014), but also irregular shape (Ioannidou & Pearson, 2019; Johannesson et al., 2017).

Among these, the long and narrow form was considered as a good option. Since the aspect ratio could affect the hydraulic performance of CW systems (Kadlec & Wallace, 2008; Su et al., 2009), for example, in Sweden, Kynkäänniemi et al. (2013) studied the P retention of a small FWS CW for ADW treatment. Although the CW occupied only 0.3% of the catchment area, its shape was long and narrow, which enhanced hydraulic efficiency. As a result, it showed a good treatment performance. During the study period, TP was reduced by 69 kg ha<sup>-1</sup> yr<sup>-1</sup>, equivalent to 36% of the incoming loads. In the study of Lavrić et al. (2020a), the FWS CW had a limited number of dead zones after long-term operation due to the presence of successive meanders within the system and a long and narrow watercourse indicated a higher value of aspect ratio.

An Irish study reported that the integrated CW could reduce the effluent P concentration to the desirable value of below 1 mg L<sup>-1</sup> only in the case of the aspect ratio lower than 2.2 (Scholz et al., 2007). Furthermore, the authors argued that an even greater treatment effect could be achieved with a ratio closer to 1. However, there are different opinions on this matter. For example, to reach laminar flow the recommended aspect ratio by Tao et al. (2014) ranged from 3 to 6, whilst Su et al. (2009) suggested the optimal ratio would be larger than 5, or at least 1.88 for maintaining uniform flow. Despite different aspect ratio values proposed, Kadlec and Wallace (2008) argued that there is no specific requirements as long as it is within a reasonable range (e.g., from 2 to 10).

Regarding water depth, the creation of deep cells possibly contributes to saving land resources needed for purification processes. However, there is still no clear conclusion on the benefits of the application of deep cells in FWS CWs (Kadlec, 2007). White (2018) investigated the effect of deep zones in two FWS CWs in the USA. CW1 was made up of both shallow cells (average water depth of 20.3 cm) and deep cells (average water depth of 76.2 cm), while CW2 had successive deep zones with an average water depth of 80 cm. It was found that in CW1, deep

zones contributed the most to the N removal in the systems. However, the overall removal efficiency in CW1 (65.1%) was not better than the one measured in CW2 (74.4%), which was attributed to the combined effect of several factors (e.g., water depth, HRT, inlet concentration, and plant species richness). In another study, Pugliese et al. (2020) performed tracer experiments on a 6-year-old Danish CW mainly composed by alternated deep and shallow zones. It was found that the shallow zones functioned as barriers when the flow from completely-mixed deep zones reached them at high velocity, which could increase the overall HRT in the system.

In summary, the length-to-width aspect ratio should be within the range of 2–10 (Kadlec & Wallace, 2008). It is encouraged to define both the wetland-to-catchment ratio and the shape of CW systems based on local conditions (Steidl et al., 2019; Tournebize et al., 2017). According to previous findings, the optimal range of wetland-to-catchment ratios recommended was within 1–8% to reach 50% N removal (Garnier et al., 2014; Ligi et al., 2015; Tanner & Kadlec, 2013), or within 1.5–4% suggested by Moreno-Mateos et al. (2010) for ideal nitrate removal. Particularly, the lowest wetland-to-catchment ratio was suggested to be 1% for CWs with an average water depth of 0.8 m (Tournebize et al., 2017).

## 3.2 Hydraulic Design

### 3.2.1 Hydraulic Characteristics

The hydraulics of CW systems can not only determine the distribution of contaminants, but can also affect removal efficiencies (Pugliese et al., 2020). First of all, it can affect HRT, one of the key parameters that determines the overall performance of CW treatment systems (Pugliese et al., 2020). It is reported that longer HRTs can always result in a greater treatment performance of CWs.

For example, Allred et al. (2014) operated a wetland reservoir sub-irrigation system (WRSIS) in Ohio, USA, with the goal of agricultural water recycling. It was built in 2003 and consisted of a CW and reservoir connected with subsurface pipes, achieving water purification, storage, and farmland irrigation via the transportation of water from the reservoir to plant roots. In 2009, four field tests on nitrogen removal have been performed under different

conditions (i.e., inflow volume, HRT, nitrogen input load), occurring in different time periods from May to November. The most effective was test 3 with reductions of 44% for  $\text{NO}_3^-$ -N, 87.5% for  $\text{NH}_4^+$ -N, and 44.9% for TN. The authors attributed such a result to a longer HRT (5.3 days) of test 3 if compared to test 1 (1.8 days) and test 2 (1.7 days). Nevertheless, even though Test 4 had a HRT of 11.1 days, the highest one, the removal of  $\text{NO}_3^-$ -N and TN was still as low as 15.6% and 16.1%, respectively. Unlike the other tests carried out during warm months (May and June), test 4 occurred between October and November, thus the cold temperatures had a negative influence on the activities of denitrifying bacteria. In fact, the maximum TN removal obtained by the authors was not so satisfactory, but consistent with the common range of 40–55% reported by Vymazal (2007). Furthermore, the application of hydraulic structures (i.e., an adjustable height weir, a peninsula) should be highlighted. The outlet weir regulated water discharge and the peninsula functioned as a baffle, both advantageous to optimization of the treatment efficiency. In terms of the wetland-to-catchment ratio, it was reported to be 2%, which is within the range recommended (Section 3.1).

In contrast to HRT, the effect of flow characteristics on CWs was not always easy to generalize (e.g., flow direction, hydrological regime). Regarding subsurface flow CWs (SSF CWs), Hoffmann et al. (2019) applied three types of flow direction (i.e., horizontal flow, up-flow, down-flow) in a total of six 100 m<sup>3</sup> woodchip-based CWs. After a 2-year monitoring period, they observed a difference in the range of 12–15% for N removal efficiency between the most (horizontal CW) and least (up-flow CW) performing systems, under similar HLRs and water temperatures. Furthermore, the impact of flow design on N removal was in accordance with the hydraulic efficiencies (the highest in horizontal CWs and the lowest in vertical up-flow CWs). It was thus demonstrated that flow directions could affect the treatment capacity of CW systems due to the variation of hydraulic efficiency. On the other hand, it was not the case in Bruun et al. (2016), which investigated the same systems. The authors reported that the vertical down-flow SSF CW achieved the highest removal rate of 3.64 g N m<sup>-2</sup> d<sup>-1</sup> under the low flow rate applied (0.49 L s<sup>-1</sup>), while the horizontal CW had the highest removal rate of 10.5 g N m<sup>-2</sup> d<sup>-1</sup> among all the systems (horizontal, vertical

up-flow, and vertical down-flow CWs) under the high flow rate of 1.83 L s<sup>-1</sup>. Such differences in findings could be attributed to the different conditions applied in the two studies. For instance, the study of Bruun et al. (2016) was performed with two fixed flow rates (0.49 and 1.83 L s<sup>-1</sup>), whilst Hoffmann et al. (2019) were conducted under natural conditions and daily fluctuation of HLRs, and a higher HLR causes a decrease in HRT and therefore a reduction in the pollutant removal (Mancuso et al., 2021).

Similar findings were reported by Tanner and Kadlec (2013) who applied a simple first-order dynamic model, in order to explore the impact of influent hydrological regimes on  $\text{NO}_3^-$ -N removal efficiency in FWS CWs. The findings showed that  $\text{NO}_3^-$ -N removal in the systems with fluctuating flows was 5–21% lower compared to the ones under steady flows. It was thus concluded that a wetland could have greater removal performance under less variable flow regimes.

Another case study for treating drainage water from a hydroponic farm in Jordan, by Abbassi et al. (2011), presented that the vertical CW had overall better effectiveness on pollutant removals (e.g. BOD<sub>5</sub>, COD, nutrients) than the horizontal one, under a series of HRTs applied. It can be explained by the fact that distribution of water flow within the vertical system led to a greater contact with the substrates and plants, therefore promoting treatment effects.

Moreover, in order to achieve an optimum performance, flow direction was recommended to be considered together with HLR, which was regarded as another critical parameter for the CW design by Vymazal et al. (2020).

In summary, the different findings reported that the effects of flow direction on CWs should be designed together with specific experimental conditions, since the treatment performance of CWs on different pollutant depends on the combination of these factors.

### 3.2.2 Hydraulic Configuration

Owing to the fact that the presence of simple hydraulic structures (e.g., gated spillways, baffle curtains) can generally lead to an increase of HRT, the application of such structures within a FWS wetland system was often highlighted.

For example, Lebrun et al. (2019) studied a 5-year-old French CW and sampled both suspended

sediments and free inorganic metallic contaminations and weak organic complexes in water. In this study, the CW configuration was divided into several sub-basins by bunds, thereby lengthening HRT. During the three-month observation, the positive relationship between removal efficiency and HRT was demonstrated. The results revealed that the metals present in trapped sediments were reduced by 11–23% from the inlet to the outlet (i.e., As, Cd, Cr, Co, Cu, Ni, Pb, Sb, Se, and Zn). On the other hand, among all the metals measured in the water column, five of them (i.e., Cd, Cr, Co, Mn, and Ni) indicated significant abatements ranging from 13 to 51%.

Groh et al. (2015) reported two CWs investigated in Illinois, USA. Their study focused on the nitrogen removal and greenhouse gas (GHG) emissions. The results showed that the wetlands provided a similar treatment capacity for nitrate after 18 years of operation, together with relatively low GHG emissions. Specifically, the wetlands achieved a nitrate removal of 56% during the monitoring period of 2012–2013, mainly attributed to denitrification. Additional removal of 6.1%, caused by the seepage losses that occurred in riparian buffer strip, further increased the total nitrate removal to 62%. To some extent, it demonstrated the impact of hydraulic structures on CW treatment efficiency. The authors also found that most GHG was emitted from the wetland originated from its the wetland originated from its terrestrial zones, probably influenced by some of the specific factors discussed in Section 3.1, such as CW size and climate conditions.

Another study by Carrer et al. (2011) studied 7-year (2003–2009) variations of a CW system located in Northern Italy before and after the construction of a gated spillway and a storage basin. The results showed that the greater purification efficacy of the system was achieved during the period of 2006–2009, after the construction of these hydraulic structures, rather than during the 2003–2005, period before their construction. Specifically, in comparison with the first part of the monitoring campaign, annual nutrient discharge from the system after the introduction of new hydraulic structures was reduced by 71%, 57%, 55%, and 15% for N-NO<sub>x</sub>, P-PO<sub>4</sub>, TN, and TP, respectively. Those improvements can be certainly attributed to the presence of the gated spillway and larger water storage, both resulting in a higher HRT in the CW

system and therefore increased removal efficiencies. Similarly, Tournebize et al. (2017), reported that small dikes (i.e., vegetated embankments) applied in the investigated CW lengthened the flow path and decreased the flow rates, leading to an increase of HRT in systems. These findings were also confirmed by Ioannidou and Pearson (2018), who analyzed the performance of 6 full-scale systems with different design parameters in the UK (including 4 FWS CWs and 2 lagoons). It was concluded that the implementation of obstacles (e.g., baffle curtains) can effectively decrease short-circuiting levels and optimize hydraulic performance and treatment efficiency of systems.

On the other hand, it is recommended that the proper location and configuration of both inlets and outlets should be taken into consideration. For example, Tournebize et al. (2017) reported that positioning inlets and outlets at the edges of the flow pathway could limit the occurrence of hydraulic dead zones. In the study of Ioannidou and Pearson (2018), the bunded outlet (closed elevated exit pipe) was recommended. It can be explained by the fact that such CWs had higher water depth, better pollutant spread, and higher HRT.

In addition, it should be also highlighted that the hydraulic efficiency index ( $\lambda$ ) is an important reference index when modifying the aforementioned hydraulic design (e.g., the configuration of inlet and outlet points, implementation of obstacles) (Su et al., 2009). It is usually considered to be good hydraulic efficiency when  $\lambda$  exceeds 0.75, though the satisfactory range may be within  $0.5 < \lambda \leq 0.75$  (Persson et al., 1999). Hydraulic efficiency not only reflects the flow distribution of influents, but also indicates the amount of mixing or recirculation within CW systems (Persson et al., 1999). Its calculation can result in an advanced understanding of the processes occurring within FWS CWs and can shed better light on the removal rates.

Accordingly, it should be noted that the CW configurations can enhance the overall treatment capacity of systems, for example, through the proper positioning of inlet and outlet points and the implementation of different hydraulic structures (e.g., gated spillways, baffle curtains) (Carrer et al., 2011; Ioannidou & Pearson, 2018; Tournebize et al., 2017).

### 3.3 Vegetation Management

The treatment processes of CW systems are dominated by the interaction of water with vegetation cover (Ioannidou & Pearson, 2019). The presence of vegetation within a CW system facilitates multiple treatment processes and enhances HRT by decreasing flow rates (Allred et al., 2014; Lee et al., 2017). Moreover, plant uptake and biomass accumulation could contribute to the removal of nutrients and pesticides (Nan et al., 2020; Picard et al., 2005; Vymazal & Březinová, 2015). They enhance oxygen availability of CW systems through root oxygen release while the decaying plant residues could be a source of carbon, satisfying treatment process needs to some extent (e.g., microbial decomposition, nitrification) (Álvarez-Rogel et al., 2020; Barbera et al., 2009; Lavrić et al., 2020b; Vymazal, 2017). In addition, vegetation can promote water column-sediment interactions (e.g., the soluble transport of nitrogen species due to macrophyte water uptake) (Martin et al., 2003) and provide surface area for biofilm growth and microbial attachment (Barco & Borin, 2020; Brix, 1997; Kumwimba et al., 2017).

A study, taking place in France, reported the fate of the herbicide glyphosate and its main degradation product aminomethylphosphonic acid (AMPA) in a stormwater wetland during three consecutive seasons of applying this product (Imfeld et al., 2013). The authors aimed to quantify the variation of the total glyphosate loadings (derived from both glyphosate and AMPA). The data reflected a gradual growth of removal over three monitoring years, i.e., 75% in 2009, 90% in 2010, and 99% in 2011. In this study, it was explained by the fact that (i) increasing plant cover (from less than 1% to about 100%) in the period 2009–2011 led to more sorption of glyphosate and AMPA, (ii) there was a better adaptation of microorganisms over time. Similarly, Koskiaho and Puustinen (2019) reported the stable and even better treatment performance of a CW (already serving for 15 years) compared to the capacity during the early years of establishment. Similar removals of TSS (68% and 74%) and TP (62% and 58%), more effective removals of  $\text{NO}_3^-$ -N (35% and 69%) and dissolved reactive P (DRP) (27% and 76%) were measured during 1999–2000 and 2007–2014, respectively. The authors attributed it to vegetation growth, increased biological activity, and unexhausted soil adsorption capacity.

Due to plant uptake capacity, biomass harvesting can effectively remove pollutants from CWs (Vymazal, 2007). Furthermore, harvested biomass might also be recycled for different uses (Mancuso et al., 2021). Regular harvesting could not only avoid the release of pollutants accumulated by plant uptake back into CW treatment systems, but could alter the original plant cover and facilitate the new plant growth (Díaz et al., 2012; Giannini et al., 2018; Kumwimba et al., 2018; Vymazal, 2007). It is reported by Hoffmann et al. (2012) that P accumulation in the aboveground biomass reached  $10.3 \text{ kg P ha}^{-1} \text{ yr}^{-1}$  and  $16.5 \text{ kg P ha}^{-1} \text{ yr}^{-1}$  for the two studied wetlands, respectively. The results implied the good potential for plant uptake of phosphorus. In order to retain the great capacity; therefore, annual harvesting was recommended by the authors. To some extent, small-scale harvest can be also seen as an economical way to select the species most suitable for water purification, while at the same time reducing ecosystem disturbances compared with the full-scale harvesting (Carty et al., 2008).

On the other hand, the time needed for vegetation establishment and development should be taken into account when operating a new CW. Lenhart et al. (2016) reported that the establishment of vegetation was slow during the first monitoring year due to the effect of surface water flooding and was completed by the second and third year of monitoring. Consequently, the initial plant uptake was negatively affected by the delay in plant development. This can be explained by the fact that a delay in plant development can influence the plant root system, and the morphological and physiological properties of roots are known to affect initial plant uptake and transport (Wang et al., 2006). A study conducted by Steidl et al. (2019) attributed the good potential for nitrogen removal to the quick vegetation development in the CW they investigated. This was in agreement with the findings of Nilsson et al. (2020), which pointed out the importance of emergent and developed vegetation existence in young CWs, since CWs fully covered by vegetation were observed to be more efficient in terms of N removal if compared to unplanted systems. Moreover, a subsurface accumulation of plant litter and roots during the vegetation maturation can positively contribute to additional surfaces for microbial biofilm colonization and carbon subsidies for aquatic microbes, which are also responsible of N



removal. Moreover, the microbial population control was suggested by different researchers, which proposed the NAS method to enrich the nitrifiers population and therefore to increase the N removal (Sepehri & Sarrafzadeh, 2018).

At the same time, research on vegetation species within CW systems, and the effect of different species, was widely carried out with the aim of improving treatment efficiencies. In Italy, Pappalardo et al. (2017) evaluated the adaptability of seven local macrophytes within a floating treatment wetland system. The authors summarized that high plant growth rate, high tissue nutrient content, and sustainability were primary characteristics for the ideal species that could increase the pollutant uptake (e.g., nutrients) or create better conditions for their removal. In this study, *Carex* spp. was considered as the most suitable species of all, due to its highest biomass production (433.13 g m<sup>-2</sup>) and total nitrogen uptake (4.84 g m<sup>-2</sup>). In Denmark, Olesen et al. (2018) carried out a mesocosm experiment to explore the effects of submerged species. It was reported that in comparison with unplanted mesocosms, the ones planted with submerged macrophytes exhibited fivefold and threefold enhancement of NH<sub>4</sub><sup>+</sup> and PO<sub>4</sub><sup>3-</sup> uptake rates, respectively. In their study, it was also indicated that multi-species communities did not perform more efficiently in nutrient uptake than single-species communities. In Southern China, Yang et al. (2016) reported the effectiveness of aquatic vegetables—water spinach and water bamboo planted in wetlands. It was reported that TN and TP uptake by the two plants reached 274 kg ha<sup>-1</sup> and 24 kg ha<sup>-1</sup> during the entire growth period, respectively. In Northern Italy, an 18-year-old FWS CW studied by Lavrnić et al. (2020b) found that plant uptake of pollutants TN, TP, and total organic carbon (TOC) was in accordance with coverage of the dominant species *Phragmites australis* in the CW, although *Typha latifolia* and *Carex* spp. covered a considerable portion of the same CW system.

However, the possible adverse effects related to vegetation were observed. For example, Maynard et al. (2014) indicated that the carbon retention potential of CW systems could be negatively affected by algal growth. In their study, actual C removal of the investigated CW (located in California, USA) was weakened due to the production of algal C, especially during years when the system had a lower plant cover.

Similarly, the findings by Ridge et al. (2019) showed that the exotic invasive species were more likely to be infested by plant pathogen *Phytophthora* species. Such studies implied the importance of vegetation management in CWs, whether for native plants or for invasive ones.

In summary, it can be concluded that the application of vegetation-related management techniques (e.g., vegetation establishment and routine harvesting) are crucial for long-term and effective nutrient removal (Hoffmann et al., 2012; Margalef-Marti et al., 2019; Nilsson et al., 2020; O'Geen et al., 2010). Plants were recommended to be developed before a start-up period for the system, avoiding the possible delay of their establishment process due to uncontrolled water level during operation period, which would further affect the potential of pollutant removal (Izadmehr & Rockne, 2018; Lenhart et al., 2016; Steidl et al., 2019). Vegetation species, density, and harvest regimes, which are closely associated with removal efficiencies, are the important factors to be taken into account (Lenhart et al., 2016; Steidl et al., 2019; Wu et al., 2013). In addition, it is not advisable to plant seedlings in CW systems, since they generally need longer time to develop and are relatively more sensitive to pollutants and change of water level (Carty et al., 2008). Of course, all these are affected by the climate type at the CW site since it, among other parameters, determines species that can be applied and the time needed for their development.

#### 4 The Performance of Constructed Wetlands in Field Studies

Based on the research on full-scale CWs, mesocosm experiments, and the application of the model, Sect. 3 provides detailed information on the design and operation of CW systems. In Sect. 4, focusing on field studies (Table 1), the association between the actual performance of CWs and related factors was further discussed.

##### 4.1 The Relationship Between Nitrogen Removals and Related Factors

Although not all the field studies provided the necessary data, we used those available to assess the

**Table 1** General information on the constructed wetland systems extracted from 22 field studies

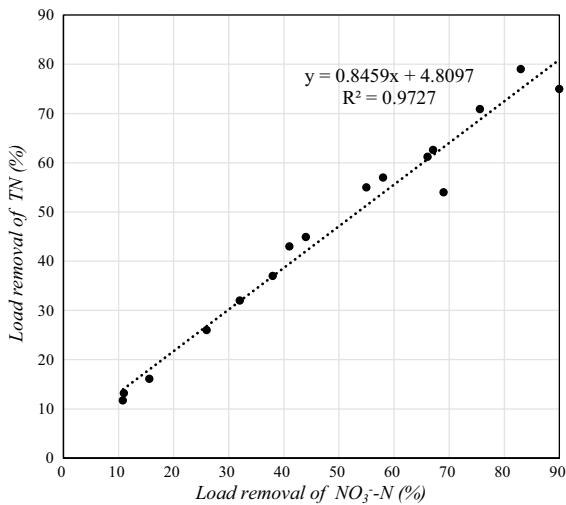
Location	Construction year	Monitoring period	Surface area	Wetland/catchment area ratio	Reference
USA	2003	May to November 2009	0.34 ha	2.00%	Allred et al. (2014)
USA	1994	2012–2013	0.60 ha	4.00%	Groh et al. (2015)
USA	1994	2012–2013	0.30 ha	3.33%	Groh et al. (2015)
Denmark	2001	2007–2009	0.62 ha	13.70%	Hoffmann et al. (2012)
Denmark	1990	1996–1998	0.59 ha	2.40%	Hoffmann et al. (2012)
France	2002	2009–2011	0.03 ha	0.07%	Imfeld et al. (2013)
USA	2005	April to October 2008	20.65 ha	-	Karpuzcu and Stringfellow (2012)
USA	In the 1960s	April to August 2008	8.09 ha	-	Karpuzcu and Stringfellow (2012)
USA	-	April to October 2008	27.00 ha	-	Karpuzcu and Stringfellow (2012)
Finland	1998	2007–2014	0.60 ha	5.00%	Koskiahho and Puustinen (2019)
Sweden	2009	2010–2011	0.08 ha	0.30%	Kynkäänniemi et al. (2013)
Italy	2000	2017, 2018–2019	0.40 ha	3.00%	Lavrnić et al. (2020a), Lavrnić et al. (2020b)
France	2010	March to May 2015	0.53 ha	0.15%	Lebrun et al. (2019)
China	2005	2014–2016	-	-	Li et al. (2018)
Italy	1996	2007–2012	0.32 ha	5.82%	Maucieri et al. (2014)
USA	1993	2004–2005	7.30 ha	0.32%	Maynard et al. (2011)
USA	1993	April to September 2007	4.50 ha	0.20%	Maynard et al. (2014)
Italy	2014	2014–2015	3.30 ha	-	Pappalardo et al. (2017)
Denmark	2010	March to April 2016	0.30 ha	0.79%	Pugliese et al. (2020)
USA	-	2011–2012	3.12 ha, 1.62 ha	-	Ridge et al. (2019)
Italy	1996	2007–2013	0.32 ha	7.11%	Tolomio et al. (2019)
France	2001	2008–2010	0.42 ha	1.20%	Tournebize et al. (2013)
France	2007	2008–2010	0.13 ha	0.50%	Tournebize et al. (2013)
France	-	2012–2014	0.01 ha	-	Vallée et al. (2015)
USA	1997	2009–2013	3.80 ha	7.82%	White (2018)
China	-	2012	0.02 ha	-	Yang et al. (2016)

relationship between nitrogen removals (TN and  $\text{NO}_3^-$ -N) and main design or operational parameters. Wetland-to-catchment ratio and average water depth were the two factors for which enough data was available and for which the strongest relationship was obtained. In general, the treatment performance of full-scale CW systems reflected connections with these factors to some extent.

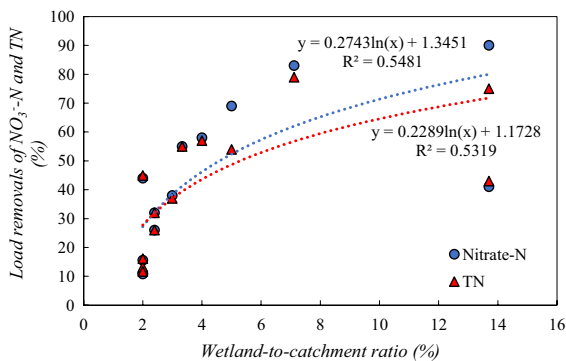
Figure 1 presents the relationship between load removals of  $\text{NO}_3^-$ -N and TN. Similar values of parameters  $\text{NO}_3^-$ -N and TN were shown, which can be explained by the fact that  $\text{NO}_3^-$ -N is usually considered as the primary N form in ADW (Hoffmann et al., 2012; Lavrnić et al., 2020b).

Figure 2 presents the relationship between load removal and the wetland-to-catchment ratio. Overall

similar trend was found for parameters  $\text{NO}_3^-$ -N and TN. However, the goodness of fit ( $R^2$ ) of both parameters was not high. The reasons might be that (i) the data quantity was not sufficient to precisely describe the relationship between load removal and wetland-to-catchment ratio, or (ii) the wetland-to-catchment ratio was not the only factor that dominated the removals, as already suggested by Moreno-Mateos et al. (2010). Based on these results, the recommended wetland-to-catchment ratio should be at least 5% to achieve  $\text{NO}_3^-$ -N and TN removals higher than 50%. This finding was not consistent with Vymazal (2017), who suggested 1% was sufficient to reach 40% TN removal and who also reported that the ratio larger than 1% did not significantly increase TN removal.

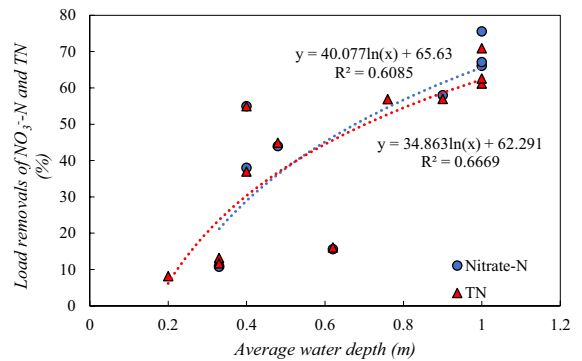


**Fig. 1** Relationship between load removals of  $\text{NO}_3^-$ -N and TN



**Fig. 2** Relationship between load removals of  $\text{NO}_3^-$ -N and TN and wetland-to-catchment ratio

Figure 3 shows that the load removals of  $\text{NO}_3^-$ -N and TN may be positively affected by average water depth. The increased water depth could yield a relatively higher removal, which was in agreement with the findings of White (2018). It should be noted that the goodness of fit reached 0.67 and 0.61 for TN and  $\text{NO}_3^-$ -N, respectively. However, they were both lower than 0.70, which indicated that the mathematical models were not good enough. This finding was also supported by Kadlec (2007), who reported that the relationship remained uncertain and there were not enough data to make a clear conclusion on the advantage of FWS CWs with different water depth.



**Fig. 3** Relationship between load removals of  $\text{NO}_3^-$ -N and TN and average water depth

#### 4.2 Variation in Treatment Performance Over the Long-Term Operation

Among the total field studies, few reported the variation in treatment performance of the same CW over the long term. Although the findings are limited, they can still provide a reference for the design and operation of CWs in the future.

For instance, In Tolomio et al. (2019), the authors confirmed the reliable treatment performance regarding nutrient removal of an Italian FWS CW system, which already had a 10-year-long operation lifespan. The yearly TN removal remained relatively stable and was on average 79% during the monitoring period (2007–2013), slightly lower than 90% provided in the previous study of Borin and Tocchetto (2007), measured in earlier period (1998–2002) of the same CW system establishment. On the other hand, the mass load removal efficiency substantially varied from 3 to 93% and from 25 to 94%, for  $\text{PO}_4$ -P and TP, respectively. The fluctuation was considered as rational and expected, especially for short-term investigations (Mitsch et al., 2012). Moreover, Koskiaho and Puustinen (2019) reported that the CW they had tested showed better treatment performance in later experimental years, attributed to vegetation growth, increased biological activity and unexhausted soil adsorption capacity. Furthermore, García et al. (2020) applied a mechanistic phosphorus model in combination with wetland mesocosm data calibration to explore the behavior of wetlands under different design scenarios. They indicated that the design characterized by a bottom inlet together with an increase in HRT was more likely to result in a higher TP

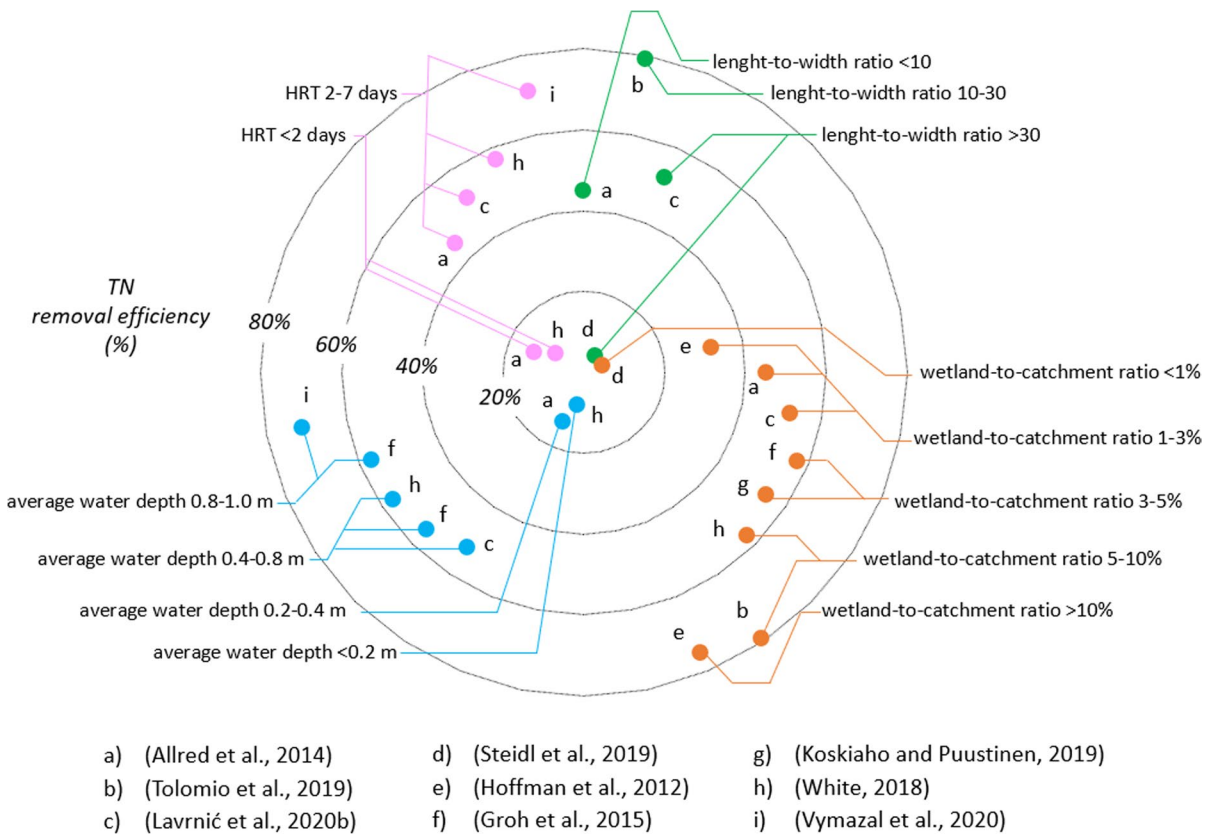
removal efficiency and therefore achieve the required effluent target concentration.

In addition, the potential of CW systems to be a CO<sub>2</sub> sink was highlighted by Maucieri et al. (2014). In this study, the authors reported abundant organic carbon (OC) storage in CW soil during the monitoring period. In particular, OC sequestration of 32.6 Mg ha<sup>-1</sup> for the 0–20 cm soil layer (measured in 2007–2012) and 78.1 Mg ha<sup>-1</sup> for the 20–50 cm layer (measured in 2009–2012) was observed. It was also found that the soil OC concentration in the top 20 cm layer had a marginal increase during the research period 2007–2012, varying from 12.3 g·kg<sup>-1</sup> to 13.1 g·kg<sup>-1</sup>. However, it increased considerably compared to 7.3 g·kg<sup>-1</sup>, the value of OC concentration measured in the construction year of the CW (1996). Similarly, Lavrnić et al. (2018) reported organic matter content in their studied CW soil increased more than 2.5 times (from 19.6 g·kg<sup>-1</sup> to 49.8 g·kg<sup>-1</sup>) during 14 years. The findings of C storage capacity were

also supported by Maynard et al. (2011), who studied the spatial and temporal variation on carbon sources of a 13-year-old CW.

### 4.3 Optimization of Constructed Wetlands for Agricultural Pollution Control

Major agricultural contributors to water pollution, and thus the main target for water pollution control are nutrients (e.g., nitrogen and phosphorus). On the basis of the main design and operational parameters that have been identified above, this section includes combinations of the design and operational recommendations that should be taken into account to enhance CWs treatment efficiency. These considerations have been provided only for TN, due to the limited available data on TP (Kynkäänniemi et al., 2013; Scholz et al., 2007) and its low concentrations in ADW (Lavrnić et al., 2020b) (Fig. 4). For the considered studies, the highest



**Fig. 4** Main design and operational factors affecting removal of TN in CWs

TN removal rates have been observed for the longest HRTs (Vymazal et al., 2020; White, 2018). The treatment performance of CWs can largely depend on the selected HRT because it represents the contact period between agricultural drainage water and the wetland system. As recommendation, the longer HRT, the more removal of pollutants occurs. As it was already discussed in the previous Section 3.1.2, the literature analysis that has been performed on the influence of the aspect ratio on pollutants removal confirmed that there is no clear correlation between this factor and the treatment performance of CWs. This was confirmed by the fact that the highest TN removal efficiency was reported for the lowest aspect ratio (Fig. 4) (Tolomio et al., 2019); on the contrary, other researchers have suggested CWs with long and narrow form (Kynkäänniemi et al., 2013), since this configuration enhanced hydraulic efficiency, and, thus, removal efficiency. For the aspect ratio, it is thus not possible to provide precise recommendations: however, it is preferable to consider the specific characteristics of each wetland system (e.g., available space, flow hydraulic type, etc.). Average water depth is also likely to influence nutrient removal performance in CWs. Among the investigated studies, the highest TN removal efficiency has been reported for the wetland systems with the highest average water depth (e.g., 0.8–1.0 m) (Fig. 4) (Groh et al., 2015; Vymazal et al., 2020). Average water depth should be thus selected in order to positively influence both flow hydraulic (e.g., HRT, etc.) and pollutants (e.g., distribution in water, etc.) characteristics in order to enhance treatment performances in CWs. Lastly, an increase in the wetland-to-catchment ratio can increase the TN removal efficiency (Fig. 4) (Hoffmann et al., 2012). Hence, it is recommended, whenever it is possible, to increase the CW surface/volume, since it can influence the flow hydraulic characteristics as well as the interception and path of pollutants in ADW.

In order to optimize the performance of CWs, the above-mentioned design and operational factors need to be considered together with the other aspects that have been already discussed earlier, such as local climate conditions and seasonal variation (Section 3.3.1), flow hydraulic characteristics/configuration (Section 3.2.1 and 3.2.2) and vegetation management (Section 3.2).

## 5 Conclusions

The goal of this study was to explore design and operational recommendations for CWs treating ADW and to link them to the removal efficiencies of already established CW systems. Through the literature review and especially the overview of more than 40 case studies, it can be concluded that a longer HRT, being one of the most important parameters affecting the treatment, can promote pollutant removal efficiencies of CW systems. In addition, a large wetland-to-catchment ratio can result in a high HRT, but it can also be increased by (1) the application of simple hydraulic structures within CW systems (e.g., baffle curtains, small dikes), (2) the alternated distribution of deep and shallow zones, (3) the application of bunded outlets, and (4) the presence of vegetation. These changes were shown to be beneficial for treatment efficiencies in different systems. Moreover, in order to reduce the occurrence of hydraulic dead zones and promote hydraulic efficiency of CWs, it is advised to (1) position inlet and outlet points at the edges of the flow pathway, (2) adopt a high aspect ratio, and (3) create a long and narrow CW shape. Vegetation establishment and regular harvesting can help CW systems to achieve greater pollutant removal potential. Besides, the effect of local climate condition and seasonal variation should be taken into consideration prior to CW design. As shown in the scientific literature considered, these systems may have great potential for pollutant treatment in ADW, even a long time after their establishment.

**Author Contribution** Original draft preparation: Xi Nan. Conceptualization: Xi Nan, Stevo Lavrić, Attilio Toscano. Review and editing: Xi Nan, Stevo Lavrić, Giuseppe Mancuso. Supervision: Attilio Toscano.

**Funding** This work was supported by the WATERAGRI Project (water retention and nutrient recycling in soils and streams for improved agricultural production) that has received funding from the European Union's Horizon 2020 research and innovation program under grant agreement No 858375—<https://wateragri.eu/>. The first author Xi Nan is grateful for the support from the program of the China Scholarship Council (No. 201803250059).

**Data Availability** Not applicable.

**Declarations**

**Ethics Approval and Consent to Participate** Not applicable.

**Consent for Publication** Not applicable.

**Competing Interests** The authors declare no competing interests.

**Open Access** This article is licensed under a Creative Commons Attribution 4.0 International License, which permits use, sharing, adaptation, distribution and reproduction in any medium or format, as long as you give appropriate credit to the original author(s) and the source, provide a link to the Creative Commons licence, and indicate if changes were made. The images or other third party material in this article are included in the article's Creative Commons licence, unless indicated otherwise in a credit line to the material. If material is not included in the article's Creative Commons licence and your intended use is not permitted by statutory regulation or exceeds the permitted use, you will need to obtain permission directly from the copyright holder. To view a copy of this licence, visit <http://creativecommons.org/licenses/by/4.0/>.

## References

- Abbassi, B., Al-Zboon, K., Radaideh, J., & Wahbeh, A. (2011). Using constructed wetlands to improve drainage water quality from hydroponics farms. *Irrigation and Drainage*, *60*, 370–380.
- Allred, B. J., Gamble, D. L., Levison, P. W., Scarbrough, R. L., Brown, L. C., & Fausey, N. R. (2014). Field test results for nitrogen removal by the constructed wetland component of an agricultural water recycling system. *Applied Engineering in Agriculture*, *30*, 163–177.
- Álvarez-Rogel, J., Barberá, G. G., Maxwell, B., Guerrero-Brontons, M., Díaz-García, C., Martínez-Sánchez, J. J., ... & Gómez, R. (2020). The case of Mar Menor eutrophication: State of the art and description of tested Nature-Based Solutions. *Ecological Engineering*, *158*, 106086.
- Babbar-Sebens, M., Barr, R. C., Tedesco, L. P., & Anderson, M. (2013). Spatial identification and optimization of upland wetlands in agricultural watersheds. *Ecological Engineering*, *52*, 130–142.
- Barbera, A. C., Cirelli, G. L., Cavallaro, V., Di Silvestro, I., Pacifici, P., Castiglione, V., Toscano, A., & Milani, M. (2009). Growth and biomass production of different plant species in two different constructed wetland systems in Sicily. *Desalination*, *246*, 129–136.
- Barco, A., & Borin, M. (2020). Treatment performances of floating wetlands: A decade of studies in North Italy. *Ecological Engineering*, *158*, 106016.
- Beutel, M. W., Whritenour, V., & Brouillard, E. (2013). Fecal coliform removal in a lightly loaded surface-flow constructed treatment wetland polishing agricultural runoff. *Water Science and Technology*, *68*, 909–915.
- Borin, M., & Tocchetto, D. (2007). Five year water and nitrogen balance for a constructed surface flow wetland treating agricultural drainage waters. *Science of the Total Environment*, *380*, 38–47.
- Braschi, I., Blasioli, S., Lavrić, S., Buscaroli, E., Di Prodi, K., Solimando, D., & Toscano, A. (2022). Removal and fate of pesticides in a farm constructed wetland for agricultural drainage water treatment under Mediterranean conditions (Italy). *Environmental Science and Pollution Research*, *29*, 7283–7299.
- Brauer, N., Maynard, J. J., Dahlgren, R. A., & O'Geen, A. T. (2015). Fate of nitrate in seepage from a restored wetland receiving agricultural tailwater. *Ecological Engineering*, *81*, 207–217.
- Brix, H. (1997). Do macrophytes play a role in constructed treatment wetlands? *Water Science and Technology*, *35*, 11–17.
- Bruun, J., Pugliese, L., Hoffmann, C. C., & Kjaergaard, C. (2016). Solute transport and nitrate removal in full-scale subsurface flow constructed wetlands of various designs treating agricultural drainage water. *Ecological Engineering*, *97*, 88–97.
- Budd, R., O'Geen, A., Goh, K. S., Bondarenko, S., & Gan, J. (2011). Removal mechanisms and fate of insecticides in constructed wetlands. *Chemosphere*, *83*, 1581–1587.
- Calvo-Cubero, J., Ibáñez, C., Rovira, A., Sharpe, P. J., & Reyes, E. (2014). Changes in nutrient concentration and carbon accumulation in a mediterranean restored marsh (Ebro Delta, Spain). *Ecological Engineering*, *71*, 278–289.
- Carrer, G. M., Bonato, M., Smania, D., Barausse, A., Comis, C., & Palmeri, L. (2011). Beneficial effects on water management of simple hydraulic structures in wetland systems: The Vallevecchia case study, Italy. *Water Science and Technology*, *64*, 220–227.
- Carstensen, M. V., Larsen, S. E., Kjærgaard, C., & Hoffmann, C. C. (2019). Reducing adverse side effects by seasonally lowering nitrate removal in subsurface flow constructed wetlands. *J Environ Manag*, *240*, 190–197.
- Carty, A., Scholz, M., Heal, K., Keohane, J., Dunne, E., Gouriveau, F., & Mustafa, A. (2008). Constructed farm wetlands (CFW) design manual for Scotland and Northern Ireland (Scottish Environment Protection Agency (SEPA)).
- Crumpton, W. G., Stenback, G. A., Fisher, S. W., Stenback, J. Z., & Green, D. I. S. (2020). Water quality performance of wetlands receiving nonpoint-source nitrogen loads: Nitrate and total nitrogen removal efficiency and controlling factors. *Journal of Environmental Quality*, *49*, 735–744.
- Dal Ferro, N., Ibrahim, H. M. S., & Borin, M. (2018). Newly-established free water-surface constructed wetland to treat agricultural waters in the low-lying Venetian plain: Performance on nitrogen and phosphorus removal. *Science of the Total Environment*, *639*, 852–859.
- Díaz, F. J., O'Geen, A. T., & Dahlgren, R. A. (2010). Efficacy of constructed wetlands for removal of bacterial contamination from agricultural return flows. *Agricultural Water Management*, *97*, 1813–1821.
- Díaz, F. J., O'Geen, A. T., & Dahlgren, R. A. (2012). Agricultural pollutant removal by constructed wetlands: Implications for water management and design. *Agricultural Water Management*, *104*, 171–183.
- Djodjic, F., Geranmayeh, P., Collentine, D., Markensten, H., & Futter, M. (2022). Cost effectiveness of nutrient retention in constructed wetlands at a landscape level. *Journal of Environmental Management*, *324*, 116325.
- Fida, M., Li, P., Wang, Y., Alam, S. K., & Nsabimana, A. (2022). Water contamination and human health risks in Pakistan: a review. *Exposure and Health*, *1*–21.

- García, J., Solimeno, A., Zhang, L., Marois, D., & Mitsch, W. J. (2020). Constructed wetlands to solve agricultural drainage pollution in South Florida: Development of an advanced simulation tool for design optimization. *Journal of cleaner production*, 258, 120868.
- Garnier, J., Billen, G., Vilain, G., Benoit, M., Passy, P., Tallec, G., Tournebize, J., Anglade, J., Billy, C., Mercier, B., et al. (2014). Curative vs. preventive management of nitrogen transfers in rural areas: Lessons from the case of the Orgeval watershed (Seine River basin, France). *J Environ Manag*, 144, 125–134.
- Giannini, V., Bertacchi, A., Bonari, E., & Silvestri, N. (2018). Rewetting in Mediterranean reclaimed peaty soils and its potential for phyto-treatment use. *J Environ Manag*, 208, 92–101.
- Groh, T. A., Gentry, L. E., & David, M. B. (2015). Nitrogen removal and greenhouse gas emissions from constructed wetlands receiving tile drainage water. *Journal of Environmental Quality*, 44, 1001–1010.
- Hoffmann, C. C., Heiberg, L., Audet, J., Schønfeldt, B., Fuglsang, A., Kronvang, B., Ovesen, N. B., Kjaergaard, C., Hansen, H. C. B., & Jensen, H. S. (2012). Low phosphorus release but high nitrogen removal in two restored riparian wetlands inundated with agricultural drainage water. *Ecological Engineering*, 46, 75–87.
- Hoffmann, C. C., Larsen, S. E., & Kjaergaard, C. (2019). Nitrogen removal in woodchip-based biofilters of variable designs treating agricultural drainage discharges. *Journal of Environmental Quality*, 48, 1881–1889.
- Imfeld, G., Lefrancq, M., Maillard, E., & Payraudeau, S. (2013). Transport and attenuation of dissolved glyphosate and AMPA in a stormwater wetland. *Chemosphere*, 90, 1333–1339.
- Ioannidou, V. G., & Pearson, J. M. (2018). Hydraulic and design parameters in full-scale constructed wetlands and treatment units: Six case studies. *Environmental Processes*, 5, 5–22.
- Ioannidou, V. G., & Pearson, J. M. (2019). The effects of flow rate variation and vegetation ageing on the longitudinal mixing and residence time distribution (RTD) in a full-scale constructed wetland. *Ecological Engineering*, 138, 248–263.
- Izadmehr, M., and Rockne, K. (2018). “pocket Wetlands” for nutrient removal in tile-drained agriculture. Paper presented at: World Environmental and Water Resources Congress 2018: Watershed Management, Irrigation and Drainage, and Water Resources Planning and Management - Selected Papers from the World Environmental and Water Resources Congress 2018.
- Jayasiri, M. M. J. G. C. N., Yadav, S., Dayawansa, N. D. K., Propper, C. R., Kumar, V., & Singleton, G. R. (2022). Spatio-temporal analysis of water quality for pesticides and other agricultural pollutants in Deduru Oya river basin of Sri Lanka. *Journal of Cleaner Production*, 330, 129897.
- Johannesson, K. M., Tonderski, K. S., Ehde, P. M., & Weisner, S. E. B. (2017). Temporal phosphorus dynamics affecting retention estimates in agricultural constructed wetlands. *Ecological Engineering*, 103, 436–445.
- Kadlec, R. H. (2007). The effects of deep zones on wetland nitrogen processing. *Water Science and Technology*, 56, 101–108.
- Kadlec, R. H., Roy, S. B., Munson, R. K., Charlton, S., & Brownlie, W. (2010). Water quality performance of treatment wetlands in the Imperial Valley, California. *Ecological Engineering*, 36, 1093–1107.
- Kadlec, R. H., & Wallace, S. (2008). *Treatment wetlands*, second edn (CRC press).
- Kalcic, M., Chaubey, I., Frankenberger, J., & Kladivko, E. (2012). A geospatial approach to targeting constructed wetlands for nitrate removal in agricultural watersheds. *Applied Engineering in Agriculture*, 28, 347–357.
- Karpuzcu, M. E., & Stringfellow, W. T. (2012). Kinetics of nitrate removal in wetlands receiving agricultural drainage. *Ecological Engineering*, 42, 295–303.
- Kill, K., Pärn, J., Lust, R., Mander, Ü., & Kasak, K. (2018). Treatment efficiency of diffuse agricultural pollution in a constructed wetland impacted by groundwater seepage. *Water*, 10.
- Koskiaho, J., & Puustinen, M. (2019). Suspended solids and nutrient retention in two constructed wetlands as determined from continuous data recorded with sensor. *Ecological Engineering*, 137, 65–75.
- Kumwimba, M. N., Zhu, B., Muyembe, D. K., & Dzakpasu, M. (2017). Growth characteristics and nutrient removal capability of eco-ditch plants in mesocosm sediment receiving primary domestic wastewater. *Environmental Science and Pollution Research*, 24, 23926–23938.
- Kumwimba, M. N., Meng, F. G., Iseyemi, O., Moore, M. T., Bo, Z., Tao, W., Liang, T. J., & Ilunga, L. (2018). Removal of non-point source pollutants from domestic sewage and agricultural runoff by vegetated drainage ditches (VDDs): Design, mechanism, management strategies, and future directions. *Science of the Total Environment*, 639, 742–759.
- Kynkäänniemi, P., Ulén, B., Torstensson, G., & Tonderski, K. S. (2013). Phosphorus retention in a newly constructed wetland receiving agricultural tile drainage water. *Journal of Environmental Quality*, 42, 596–605.
- Land, M., Granéli, W., Grimvall, A., Hoffmann, C. C., Mitsch, W. J., Tonderski, K. S., & Verhoeven, J. T. A. (2016). How effective are created or restored freshwater wetlands for nitrogen and phosphorus removal? A systematic review. *Environmental Evidence*, 5.
- Lavrnić, S., Braschi, I., Anconelli, S., Blasioli, S., Solimando, D., Mannini, P., & Toscano, A. (2018). Long-term monitoring of a surface flow constructed wetland treating agricultural drainagewater in Northern Italy. *Water (Switzerland)*, 10.
- Lavrnić, S., Alagna, V., Iovino, M., Anconelli, S., Solimando, D., & Toscano, A. (2020a). Hydrological and hydraulic behaviour of a surface flow constructed wetland treating agricultural drainage water in northern Italy. *Science of the Total Environment*, 702.
- Lavrnić, S., Nan, X., Blasioli, S., Braschi, I., Anconelli, S., & Toscano, A. (2020b). Performance of a full scale constructed wetland as ecological practice for agricultural drainage water treatment in Northern Italy. *Ecological Engineering*, 154.
- Lebrun, J. D., Ayrault, S., Drouet, A., Bordier, L., Fechner, L. C., Uher, E., Chaumont, C., & Tournebize, J. (2019). Ecodynamics and bioavailability of metal contaminants in a constructed wetland within an agricultural drained catchment. *Ecological Engineering*, 136, 108–117.

- Lee, S., Maniquiz-Redillas, M. C., Choi, J., & Kim, L. H. (2017). Causes and control measures for algae occurrence in a constructed wetland treating stream runoff from agricultural land use. *Desalination Water Treat*, *63*, 404–411.
- Lenhart, C., Gordon, B., Gamble, J., Current, D., Ross, N., Herring, L., Nieber, J., & Peterson, H. (2016). *Design and hydrologic performance of a tile drainage treatment wetland in Minnesota* (p. 8). Water (Switzerland).
- Li, N., Tian, X., Li, Y., Fu, H. C., Jia, X. Y., Jin, G. Z., & Jiang, M. (2018). Seasonal and spatial variability of water quality and nutrient removal efficiency of restored wetland: A case study in Fujin National Wetland Park, China. *Chinese Geographical Science*, *28*, 1027–1037.
- Ligi, T., Truu, M., Oopkaup, K., Nölvak, H., Mander, Ü., Mitsch, W. J., & Truu, J. (2015). The genetic potential of N<sub>2</sub> emission via denitrification and ANAMMOX from the soils and sediments of a created riverine treatment wetland complex. *Ecological Engineering*, *80*, 181–190.
- Mancuso, G., Bencresciuto, G. F., Lavrnić, S., & Toscano, A. (2021). Diffuse water pollution from agriculture: A review of nature-based solutions for nitrogen removal and recovery. *Water*, *13*, 1893.
- Margalef-Martí, R., Carrey, R., Merchán, D., Soler, A., Causapé, J., & Otero, N. (2019). Feasibility of using rural waste products to increase the denitrification efficiency in a surface flow constructed wetland. *Journal of Hydrology*, *578*, 124035.
- Martin, J., Hofherr, E., & Quigley, M. F. (2003). Effects of *Typha latifolia* transpiration and harvesting on nitrate concentrations in surface water of wetland microcosms. *Wetlands*, *23*, 835–844.
- Maucieri, C., Salvato, M., Tamiazzo, J., & Borin, M. (2014). Biomass production and soil organic carbon accumulation in a free water surface constructed wetland treating agricultural wastewater in North Eastern Italy. *Ecological Engineering*, *70*, 422–428.
- Maynard, J. J., Dahlgren, R. A., & O'Geen, A. T. (2011). Soil carbon cycling and sequestration in a seasonally saturated wetland receiving agricultural runoff. *Biogeochemistry*, *8*, 3391–3406.
- Maynard, J. J., Dahlgren, R. A., & O'Geen, A. T. (2014). Autochthonous and allochthonous carbon cycling in a eutrophic flow-through wetland. *Wetlands*, *34*, 285–296.
- McLaughlin, D. L., & Cohen, M. J. (2013). Realizing ecosystem services: Wetland hydrologic function along a gradient of ecosystem condition. *Ecological Applications*, *23*, 1619–1631.
- Mendes, L. R. D., Tonderski, K., Iversen, B. V., & Kjaergaard, C. (2018). Phosphorus retention in surface-flow constructed wetlands targeting agricultural drainage water. *Ecological Engineering*, *120*, 94–103.
- Mendes, L. R. D., Tonderski, K., & Kjaergaard, C. (2018). Phosphorus accumulation and stability in sediments of surface-flow constructed wetlands. *Geoderma*, *331*, 109–120.
- Mitsch, W. J., Zhang, L., Stefanik, K. C., Nahlik, A. M., Anderson, C. J., Bernal, B., Hernandez, M., & Song, K. (2012). Creating wetlands: Primary succession, water quality changes, and self-design over 15 years. *BioScience*, *62*, 237–250.
- Mitsch, W. J., Zhang, L., Marois, D., & Song, K. (2015). Protecting the Florida Everglades wetlands with wetlands: Can stormwater phosphorus be reduced to oligotrophic conditions? *Ecological Engineering*, *80*, 8–19.
- Moreno-Mateos, D., Pedrocchi, C., & Comín, F. A. (2010). Effects of wetland construction on water quality in a semi-arid catchment degraded by intensive agricultural use. *Ecological Engineering*, *36*, 631–639.
- Nan, X., Lavrnić, S., & Toscano, A. (2020). Potential of constructed wetland treatment systems for agricultural wastewater reuse under the EU framework. *Journal of Environmental Management*, *275*, 111219.
- Nilsson, J. E., Liess, A., Ehde, P. M., & Weisner, S. E. (2020). Mature wetland ecosystems remove nitrogen equally well regardless of initial planting. *Science of The Total Environment*, *716*, 137002.
- O'Geen, A. T., Budd, R., Gan, J., Maynard, J. J., Parikh, S. J., & Dahlgren, R. A. (2010). Chapter one - mitigating non-point source pollution in agriculture with constructed and restored wetlands. In Adv Agron, D.L. Sparks, ed. (Academic Press), pp. 1–76.
- Olesen, A., Jensen, S. M., Alnoe, A. B., Baatrup-Pedersen, A., Lauridsen, T. L., Sorrell, B. K., & Riis, T. (2018). Nutrient kinetics in submerged plant beds: A mesocosm study simulating constructed drainage wetlands. *Ecological Engineering*, *122*, 263–270.
- Pappalardo, S. E., Ibrahim, H. M. S., Cerinato, S., & Borin, M. (2017). Assessing the water-purification service in an integrated agricultural wetland within the Venetian Lagoon drainage system. *Marine & Freshwater Research*, *68*, 2205–2215.
- Pavlidis, G., Zotou, I., Karasali, H., Marousopoulou, A., Bariamis, G., Tsihrintzis, V. A., & Nalbantis, I. (2022). Performance of pilot-scale constructed floating wetlands in the removal of nutrients and pesticides. *Water Resources Management*, 1–18.
- Persson, J., Somes, N. L. G., & Wong, T. H. F. (1999). Hydraulics efficiency of constructed wetlands and ponds. *Water Science and Technology*, *40*, 291–300.
- Picard, C. R., Fraser, L. H., & Steer, D. (2005). The interacting effects of temperature and plant community type on nutrient removal in wetland microcosms. *Bioresour Technology*, *96*, 1039–1047.
- Préau, C., Tournebize, J., Lenormand, M., Alleaume, S., Bousada, V. G., & Luque, S. (2022). Habitat connectivity in agricultural landscapes improving multi-functionality of constructed wetlands as nature-based solutions. *Ecological Engineering*, *182*, 106725.
- Pugliese, L., Kusk, M., Iversen, B. V., & Kjaergaard, C. (2020). Internal hydraulics and wind effect in a surface flow constructed wetland receiving agricultural drainage water. *Ecological Engineering*, *144*, 105661.
- Reinhardt, M., Gächter, R., Wehrli, B., & Müller, B. (2005). Phosphorus retention in small constructed wetlands treating agricultural drainage water. *Journal of Environmental Quality*, *34*, 1251–1259.
- Ridge, G. A., Bell, N. L., Gitto, A. J., Jeffers, S. N., & White, S. A. (2019). *Phytophthora* species associated with plants in constructed wetlands and vegetated channels at a commercial plant nursery over time. *HortTechnology*, *29*, 736–744.



- Scholz, M., Harrington, R., Carroll, P., & Mustafa, A. (2007). The integrated constructed wetlands (ICW) concept. *Wetlands*, 27, 337.
- Sepehri, A., & Sarrafzadeh, M. H. (2018). Effect of nitrifiers community on fouling mitigation and nitrification efficiency in a membrane bioreactor. *Chemical Engineering and Processing*, 128, 10–18.
- Soana, E., Gavioli, A., Vincenzi, F., Fano, E. A., & Castaldelli, G. (2020). Nitrate availability affects denitrification in *Phragmites australis* sediments. *Journal of Environmental Quality*, 49, 194–209.
- Song, X., Ehde, P. M., & Weisner, S. E. (2019). Effects of water depth and phosphorus availability on nitrogen removal in agricultural wetlands. *Water*, 11(12), 2626.
- Steidl, J., Kalettka, T., & Bauwe, A. (2019). Nitrogen retention efficiency of a surface-flow constructed wetland receiving tile drainage water: A case study from north-eastern Germany. *Agriculture Ecosystems & Environment*, 283.
- Su, T.-M., Yang, S.-C., Shih, S.-S., & Lee, H.-Y. (2009). Optimal design for hydraulic efficiency performance of free-water-surface constructed wetlands. *Ecological Engineering*, 35, 1200–1207.
- Takavakoglou, V., Pana, E., & Skalkos, D. (2022). Constructed wetlands as nature-based solutions in the post-COVID agri-food supply chain: Challenges and opportunities. *Sustainability-Basel*, 14(6), 3145.
- Tanner, C. C., & Kadlec, R. H. (2013). Influence of hydrological regime on wetland attenuation of diffuse agricultural nitrate losses. *Ecological Engineering*, 56, 79–88.
- Tao, W., Bays, J., Meyer, D., Smardon, R., & Levy, Z. F. (2014). Constructed wetlands for treatment of combined sewer overflow in the US: A review of design challenges and application status. *Water*, 2014, 3362–3385.
- Tolomio, M., Dal Ferro, N., & Borin, M. (2019). Multi-Year N and P removal of a 10-year-old surface flow constructed wetland treating agricultural drainage waters. *Agronomy*, 9(4), 170.
- Tournebize, J., Passeur, E., Chaumont, C., Fesneau, C., Guenne, A., & Vincent, B. (2013). Pesticide de-contamination of surface waters as a wetland ecosystem service in agricultural landscapes. *Ecological Engineering*, 56, 51–59.
- Tournebize, J., Chaumont, C., Fesneau, C., Guenne, A., Vincent, B., Garnier, J., & Mander, Ü. (2015). Long-term nitrate removal in a buffering pond-reservoir system receiving water from an agricultural drained catchment. *Ecological Engineering*, 80, 32–45.
- Tournebize, J., Chaumont, C., & Mander, Ü. (2017). Implications for constructed wetlands to mitigate nitrate and pesticide pollution in agricultural drained watersheds. *Ecological Engineering*, 103, 415–425.
- Vallée, R., Dousset, S., Schott, F. X., Pallez, C., Ortar, A., Cherrier, R., Munoz, J. F., & Benoît, M. (2015). Do constructed wetlands in grass strips reduce water contamination from drained fields? *Environmental Pollution*, 207, 365–373.
- Vymazal, J. (2007). Removal of nutrients in various types of constructed wetlands. *Science of the Total Environment*, 380, 48–65.
- Vymazal, J. (2017). The use of constructed wetlands for nitrogen removal from agricultural drainage: A review. *Scientia Agriculturae Bohemica*, 48, 82–91.
- Vymazal, J., & Březinová, T. (2015). The use of constructed wetlands for removal of pesticides from agricultural runoff and drainage: A review. *Environment International*, 75, 11–20.
- Vymazal, J., & Březinová, T. D. (2018). Treatment of a small stream impacted by agricultural drainage in a semi-constructed wetland. *Science of the Total Environment*, 643, 52–62.
- Vymazal, J., Sochacki, A., Fučík, P., Šereš, M., Kaplická, M., Hnátková, T., & Chen, Z. (2020). Constructed wetlands with subsurface flow for nitrogen removal from tile drainage. *Ecological Engineering*, 155.
- Wang, H., Inukai, Y., & Yamauchi, A. (2006). Root development and nutrient uptake. *Critical Reviews in Plant Sciences*, 25(3), 279–301.
- White, S. A. (2018). Design and season influence nitrogen dynamics in two surface flow constructed wetlands treating nursery irrigation runoff. *Water*, 10.
- Wu, M., Tang, X. Q., Li, Q. Y., Yang, W. J., Jin, F., Tang, M. Z., & Scholz, M. (2013). Review of ecological engineering solutions for rural non-point source water pollution control in Hubei Province, China. *Water, Air, & Soil Pollution*, 224.
- Wu, H., Wang, R., Yan, P., Wu, S., Chen, Z., Zhao, Y., ... & Zhang, J. (2023). Constructed wetlands for pollution control. *Nature Reviews Earth & Environment*, 1–17.
- Yang, S., Xu, J., Zhang, J., Wang, Y., & Peng, S. (2016). Reduction of non-point source pollution from paddy fields through controlled drainage in an aquatic vegetable wetland–ecological ditch system. *Irrigation and Drainage*, 65, 734–740.

**Publisher's Note** Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.