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REVIEW PAPER

Application of constructed wetlands for treating agricultural runoff and agro-industrial wastewater: a review

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Abstract With the unique advantages of cost-effectiveness and low energy consumption, constructed wetlands (CWs) are commonly used for treatment of secondary municipal wastewaters. Over the last decades, CWs have gained increased popularity for treating agricultural runoff and agro-industrial wastewater. This review highlights the practice, application, and research on wetland technology, placing them in the overall context of the need for reliable and sustainable solutions to managing agricultural runoff and agro-industrial wastewater. A critical assessment of the performance and effectiveness of wetland systems for removing various contaminants of importance to agriculture is presented. The design parameters and operational conditions affecting the efficiency of contaminant removal in CWs receiving agricultural runoff and agro-industrial wastewater are also discussed. The role of proper pretreatment, artificial aeration, effluent recirculation, in-series design, and microbial dynamics on the enhancement of treatment is provided. Challenges and perspectives for future research on agricultural treatment wetlands are also addressed.

Keywords Constructed wetlands · Agricultural wastewaters · Nutrient · Removal efficiencies · Design optimization

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Introduction

Runoff from agricultural irrigation and agro-industrial production, such as dairy, potato processing, plant nursery, sugar processing and aquaculture wastewater, has been considered as the primary cause of excess nutrients in fresh water sources and can be a major contributor to eutrophication of surface water (Healy et al., 2007; Kominami & Lovell, 2012). These types of wastewaters usually contain higher levels of nutrients (such as nitrogen and phosphorous) and organic matter (e.g., 5-days biochemical oxygen demand (BOD_5)) than municipal effluent (Dunne et al., 2005a; Healy & O'Flynn, 2011). Dunne et al. (2005a) determined the quality and quantity of wastewater generated at a 4,800 m^2 dairy farm, and indicated that yearly mass loads were 47 kg year^{-1} of soluble reactive phosphorus (SRP) and 5,484 kg year^{-1} of BOD_5. The generated runoff from agricultural irrigation and agro-industrial production can lead to diffuse and non-point source pollution, which is neither uniform nor predictable as it is associated with agricultural practices and weather conditions (Borin & Tocchetto, 2007). Both diffuse and non-point source pollution may result in the degradation of water quality, contamination of groundwater, siltation, and direct toxicity to organisms, which consequently affect biodiversity, fisheries, recreation and public health (Healy et al., 2007; Carty et al., 2008).

Constructed wetlands (CWs) are ecologically engineered systems that use natural processes involving wetland vegetation, soils and their associated microbial assemblages to improve water quality (Kadlec & Knight, 1996). According to water flow regime, CWs may be classified into three groups: free water surface flow (FWS) CWs, subsurface flow (SSF) CWs, and hybrid systems. SSF CWs usually include two basic flow types: horizontal subsurface flow (HSSF) and vertical subsurface flow (VSSF) CWs (Fig. 1). Interest in the use of CWs for remediation of runoff from agricultural irrigation and agro-industrial production has become increasingly popular over the last decades, due to their low capital and operational cost, low energy consumption, and environmental friendliness (Huett et al., 2005; Scholz, 2007; Carty et al., 2008). Agricultural treatment wetlands have been used to treat dairy wastewater (Rousseau et al., 2004; Dunne et al., 2005a; VanderZaag et al., 2008; Zhang et al., 2016), potato farm wastewater (Bosak et al., 2016), swine wastewater (Reddy et al., 2001; Lee et al., 2004; Kantawanichkul & Somprasert, 2005; Tunçsiper et al., 2015), plant nursery runoff (Huett et al., 2005), barnyard manure (Hill et al., 2000), winery wastewater (De la Varga et al., 2013; Serrano et al., 2011; Rozema et al., 2016), and aquaculture runoff (Behrends et al., 2000; Li et al., 2007; Snow et al., 2010; Zhong et al., 2011; Boxman et al., 2015). The application of agricultural treatment wetlands has been reported in many countries worldwide, such as the United States of America (Newman et al., 1999; Reddy et al., 2001; Karpuzcu & Stringfellow, 2012; Travis et al., 2012; Tunçsiper et al., 2015), Ireland (Dunne et al., 2005a; Zhang et al., 2009; Forbes et al., 2011), Canada (Smith et al., 2006; Maltais-Landry et al., 2007; Gottschall et al., 2007; VanderZaag et al., 2008; Bosak et al., 2016), Germany (Sindilariu et al., 2008), Italy (Mantovi et al., 2003; Borin & Tocchetto, 2007; Comino et al., 2011; Gorra et al., 2014), the United Kingdom (Mustafa et al., 2009), Turkey (Yalcuk & Ugurlu, 2009), Australia (Huett et al., 2005; Headley et al., 2001), New Zealand (Tanner et al., 1995), Sweden (Thorèn et al., 2004), Belgium (Rousseau et al., 2004; Boets et al., 2011), Spain (Serrano et al., 2011), Kenya (Bojceyska and Tonderski, 2007), China (He et al., 2006; Li et al., 2007; Yang et al., 2008; Lu et al. 2009a, b; Gao & Hu, 2012), Thailand (Kantawanichkul & Somprasert, 2005), Taiwan (Lin et al., 2002; Lee et al., 2004), Korea (Maniquiz et al., 2012), Vietnam (Konnerup et al., 2011), and Japan (Sharma et al., 2013; Zhang et al., 2016).

However, treatment wetlands present a wide range of environmental features, which may not only significantly affect their performance, but also make it difficult to extend the results from one scenario to other different situations (Kadlec & Knight, 1996). The removal of contaminants in treatment wetlands involves complex physical, chemical, and biological processes (Knight et al., 2000; Zhang et al., 2014). The removal efficiency in agricultural wetland systems depends on a number of variables, including organic loading (Yang et al., 2008; Sindilariu et al., 2008; Dunne et al., 2005b), soil substrate (Yates & Prasher, 2009; Braskerud, 2002; Zhu et al., 2012), hydraulic retention time (HRT) (Sindilariu et al., 2008;Yang et al., 2008; Reyes and Vidal, 2015), seasonal variation (Tunçsiper et al., 2015, Reyes and Vidal, 2015; Gorra et al., 2014), pH value (Tao et al., 2012; He et al., 2012), and the presence and type of vegetation (Huett et al., 2005; Gottschall et al., 2007; Comin et al., 1997). Vymazal (2009) reported average treatment efficiencies in HSSF CWs for treating agricultural wastewaters of 77, 51, and 54% for total suspended solids (TSS), total nitrogen (TN), and total phosphorus (TP), respectively. Healy and Flynn (2011) revealed average removal efficiencies in CWs treating dairy wastewater in Ireland of 88% for ammonium (NH_4^{+-}–N) and 80% for phosphate (PO_4^{3-}), respectively. Forbes et al. (2011) evaluated the performance of a CW for treating dairy farm wastewater and reported high reduction efficiencies of 99, 95, and 93% for BOD_5, phosphorus (P), and nitrogen (N), respectively.

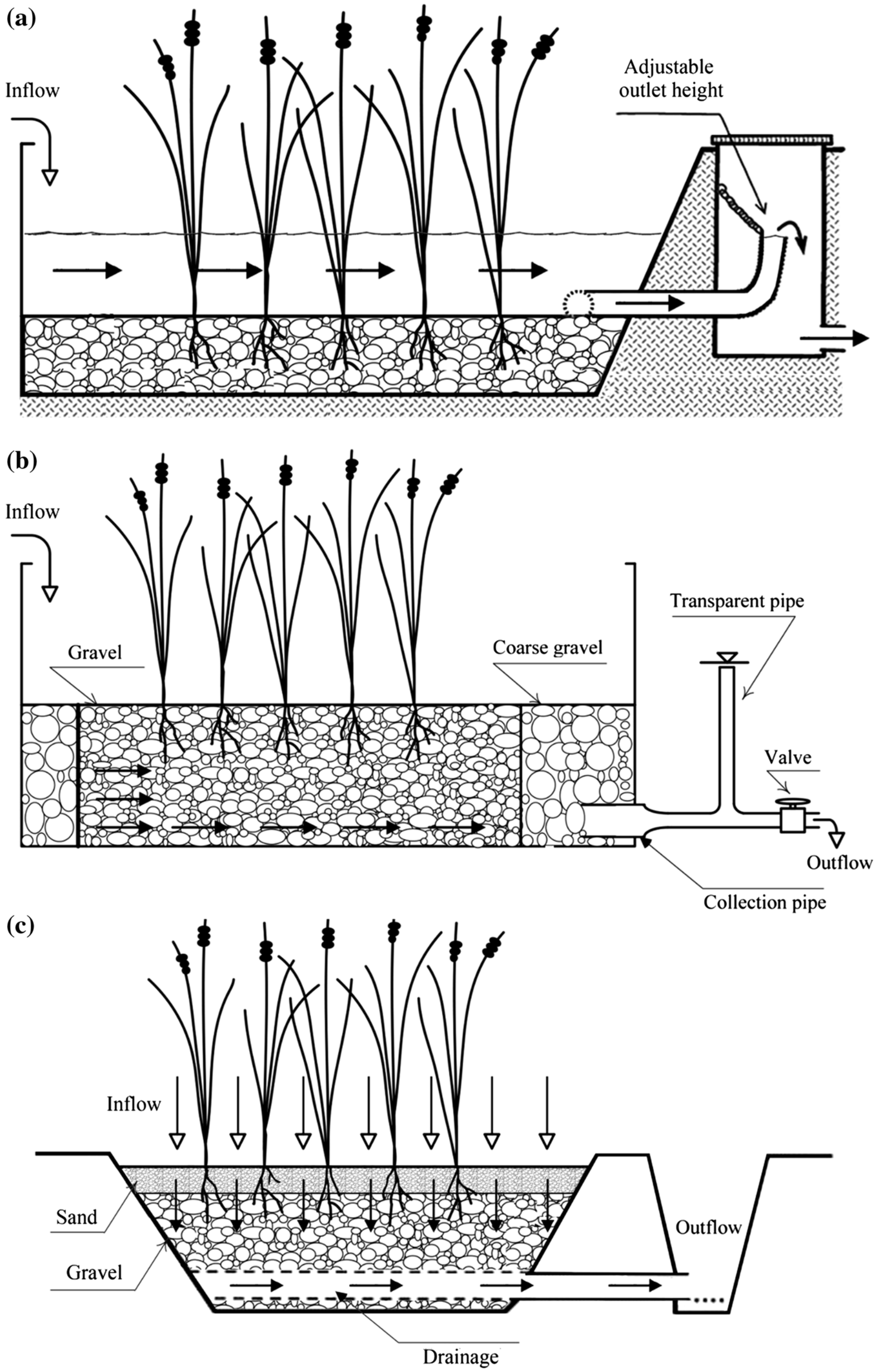

Fig. 1 Schematic layout of different types of constructed wetlands (CWs): **a** free water surface flow CWs (Vymazal, 2007); **b** horizontal subsurface flow CWs (Zhang et al., 2011); and **c** vertical subsurface flow CWs (Vymazal, 2007)

In designing CWs as management tools for treating runoff from agricultural irrigation and agro-industrial production, it is important to understand not only the contaminant removal mechanisms which occur naturally in a treatment wetland, but also how these mechanisms are affected by wetland structural components, the local environment, and the operational conditions. This review highlights the present state of knowledge regarding the practice, applications, and research on CWs for treating runoff from agricultural irrigation and agro-industrial production. More specifically, the main objectives are to (i) assess the removal efficiencies of agricultural wetland systems for water quality parameters (e.g., TSS, organic matter, nitrogen, and phosphorus); (ii) evaluate the effectiveness of agricultural wetland systems for nutrient removal; (iii) discuss advances in optimizing the design and operational conditions of agricultural treatment wetlands; and (iv) address the challenges and perspectives for future research on agricultural treatment wetlands.

Wetland performance for treating agricultural runoff

Free water surface (FWS) CWs

FWS CWs are similar to many natural wetlands where the water surface is exposed to the atmosphere (Kadlec, 1996). They are typically used for treating agricultural runoff-impacted surface waters because of their ability to deal with pulse flows and changing water levels (Maniquiz et al., 2012). In general, removal efficiencies above 70% can be achieved for TSS, COD, BOD_5, and pathogens, including bacteria and viruses, in FWS CWs (Kadlec & Wallace, 2009). However, CWs often show limited capacity for nutrient (especially phosphorous) removal (Vymazal, 2007). Removal efficiencies typically range from 40 to 50% (for nitrogen), and from 40 to 90% (for phosphorous) (Vymazal, 2007). In the present review, a summary of wetland design, operational parameters, and mean removal efficiencies in FWS CWs is shown in Table 1. Figure 2 shows the mean concentration-based removal efficiencies in different types of wetland systems receiving agricultural runoff. The results indicate that FWS CWs can efficiently remove TSS (79.2 ± 24.1%), BOD_5 (77.0 ± 20.5%), COD (71.2 ± 25.4%), and NH_4^+–N (64.8 ± 20.8%). Despite significant variations, FWS CWs also show reliable removal efficiencies for TN (57.6%) and TP (54.7%). However, the average removal efficiency of NO_3–N (43.7 ± 20.8%) is rather low in the present study.

Subsurface flow constructed wetlands (SSF CWs)

SSF CWs may include substrate for bacterial growth and sedimentation, oxygen release, nutrient uptake and storage, and rhizosphere for microbial activity enhancement (Vymazal, 2011). Tables 2 and 3 summarize the wetland design, operational treatment parameters and removal efficiencies in HSSF and VSSF CW systems. Both HSSF (83.9%) and VSSF (81.8%) CWs exhibit efficient removal efficiencies for TSS. Efficient removal of BOD_5 (79.2% for HSSF CWs and 80.0% for VSSF CWs) and COD (72.1% for HSSF CWs and 78.7% VSSF CWs) is also observed. The high removal for both suspended solids and organic matter in SSF CWs may be attributable to filtration and/or sedimentation of suspended solids, and biodegradation by microorganisms (Sundaravadivel and Vigneswaran, 2010). HSSF CWs exhibit a higher removal for NH_4^+–N (72.0%) than VSSF CWs (66.1%), although intermittent feeding leads to increased transfer of oxygen in VSSF CWs, which promotes a more oxidizing environment for nitrification. In contrast, superior nitrate (NO_3–N) removal (85.7%) is observed in HSSF CWs, compared to VSSF CWs (67.3%), owing to promotion of conditions, which are conducive to denitrification. The average removal efficiency for TN in VSSF CWs (71.7%) is higher than that that in HSSF CWs (63.2%). The average TN removal efficiency in SSF CWs is higher than that in FWS CWs (57.6%). As for TP removal, both HSSF (63.9%) and VSSF CWs (63.7%) exhibit higher potential for TP removal.

Hybrid constructed wetlands

As many wastewaters may be difficult to treat in a single-stage system, hybrid systems, which consist of various types of CWs staged in series, have been introduced (Vymazal, 2011). In general, HSSF CWs can provide good conditions for denitrification, while their ability to nitrify ammonia is limited. In contrast, VSSF CWs can remove ammonia (NH_3–N) successfully, but denitrification hardly takes place in these systems. In this regard then, various types of CWs may be combined with one another to enhance TN removal (Vymazal, 2007). A summary of wetland design and operational parameters, as well as treatment efficiencies of hybrid systems is shown in Table 4. As expected, compared to other types of agricultural treatment wetlands, hybrid systems exhibit superior removal efficiencies for various contaminants and are found to be more efficient than single-stage systems in the removal of TSS (91.2%), BOD_5 (82.7%), NH_4–N

Table 1 A summary of the wetland design/operation and treatment efficiency of free water surface constructed wetlands (FWS CWs) for treating agricultural runoff

	Type of wastewater (WW) and stage of treatment	Removal performance							Wetland design and operation				Reference
		TSS	BOD_5	COD	NH_4–N	NO_3–N	TN	TP	Dimension (m × m × m) (L × W × D)	Plant species	Hydraulic loading rate (m^3 $days^{-1}$)	Hydraulic retention time (days)	
Legnaro, Italy													
Effluent value (mg l^{-1})	Agricultural WW	–	–	–	–	–	–	–	3,200 m^2	*Phragmites australis* (Cav.)	–	–	Borin et al. (2001)
Removal efficiencies (%)		–	–	–	–	90	–	–		*Typha latifolia* L.			
Atlantic, Canada													
Effluent value (mg l^{-1})	Dairy WW	38.6	18.2	–	8.1	–	–	4.0	25.0 × 1.0 × 0.6	*Typha latifolia* L.	1,000 l $days^{-1}$	15	Smith et al. (2006)
Removal efficiencies (%)		95	99	–	94	–	–	91		*Eichhornia crassipes* (Mart.)			
Load removal (g/m^2/d)		29.33	72.07	–	5.08		–	0.37					
Connecticut, USA													
Effluent value (mg l^{-1})	Milkhouse WW	130	611	–	–	0.1	73.48	14.07	400 m^2	*Typha angustifolia* L.	–	41	Newman et al. (1999)
Removal efficiencies (%)		94	85	–	–	60	53	68		*Phragmites australis* (Cav.)			
Ottawa, Canada													
Effluent value (mg l^{-1})	Potato farm WW	44.52	43.38	–	–	–	43.54	4.25	10 × 7 × 56	*Typha angustifolia L.*	–	0.3 months	Bosak et al. (2016)
Removal efficiencies (%)		99	96	–	–	–	86	90					
Kalmar Dämme, Sweden													
Effluent value (mg l^{-1})	Agricultural WW	–	13.4	–	0.4	3.2	4.3	0.06	48 km^2	*Elodea Canadensis* (Michx.)	–	–	Thorèn et al. (2004)
Removal efficiencies (%)		–	52.2	–	50	17.9	32.8	–		*Myriophyllum spicatum* L.			
Nyanza, Kenya													
Effluent value (mg l^{-1})	Sugar factory WW	11.0	–	–	2.9	–	–	4.1	3.0 × 20.0 × 0.4	*Cyperus papyrus* L.	75 mm $days^{-1}$	–	Bojcevska and Tonderski (2007)
Removal efficiencies (%)		76	–	–	36	–	–	29		*Echinochloa pyramidalis* (Lam.)			

Table 1 continued

	Type of wastewater (WW) and stage of treatment	Removal performance							Wetland design and operation				Reference
		TSS	BOD_5	COD	NH_4–N	NO_3–N	TN	TP	Dimension (m × m × m) (L × W × D)	Plant species	Hydraulic loading rate (m^3 $days^{-1}$)	Hydraulic retention time (days)	
USA													
Effluent value (mg l^{-1})	Swine WW	1.27	–	3.14	0.27	0.01	0.74	0.37	40 × 11		–	21	Reddy et al. (2001)
Removal efficiencies (%)		68.5	–	53.4	59.5	–	51.3	44.4		*Typha latifolia* L.			
Effluent value (mg l^{-1})		3.6		8.88	0.84	0.03	1.97	0.94		*Scirpus americanus* (Pers.)		10.5	
Removal efficiencies (%)		66.4		42.6	43.1	–	36.5	30.6					
Ireland													
Effluent value (mg l^{-1})	Dairy WW	–	–	12.4	1.3	0.2	–	6.7	4,800 m^2	*Typha latifolia* L.	3.6–18.5	4	Dunne et al. (2005a, b)
Removal efficiencies (%)		–	–	95.8	95.2	–	–	69.6		*Phragmites australis* (Cav.)			
Load removal (g/m^2/days)				1.09	0.11	–	–	0.06					
Korea													
Effluent value (mg l^{-1})	Agricultural WW	10	3	7.4	0.44	1.1	6	0.4	8,861 m^2	–	180 m^3 h^{-1}	16.8	Maniquiz et al. (2012)
Removal efficiencies (%)		60	53	50	64	21	28	67					
Load removal (g/m^2/days)		7.31	1.65	3.61	0.38	0.14	1.14	0.40					
NE, Italy													
Effluent value (mg l^{-1})	Agricultural runoff	–	–	–	0.24	–	–	–	3,200 m^2	*Phragmites australis* (Cav.)	–	–	Borin & Tocchetto (2007)
Removal efficiencies (%)		–	–	–	76.7	–	90	–		*Typha latifolia* L.			
Flanders, Belgium													
Effluent value (mg l^{-1})		–	–	–	–	–	–	–	–				Rousseau et al. (2004)
Removal efficiencies (%)		75	–	61	–	–	31	26		–	–	–	
Italy													
Effluent value (mg l^{-1})	Agricultural runoff	–	–	–	–	–	–	0.159	9.29 m^2	*Typha latifolia* L.	–	2.2	Yates & Prasher (2009)
Removal efficiencies (%)								41.9		*Phalaris arundinaceae* L.			

Table 1 continued

	Type of wastewater (WW) and stage of treatment	Removal performance							Wetland design and operation				Reference
		TSS	BOD_5	COD	NH_4–N	NO_3–N	TN	TP	Dimension (m × m × m) (L × W × D)	Plant species	Hydraulic loading rate (m^3 $days^{-1}$)	Hydraulic retention time (days)	
Romana lake, USA													
Effluent value (mg l^{-1})	Agricultural drainage	–	–	–	–	17.1	–	0.14	0.9 × 0.4 × 0.6	–	12–32 m^3 $days^{-1}$	–	Karpuzcu & Stringfellow (2012)
Removal efficiencies (%)		–	–	–	–	23–35	–	39.1					

(77.6%), TN (73.3%), and TP (69.9%). HSSF CWs show highest removal efficiencies for NO_3^-–N (85.8%), while VSSF CWs exhibit best COD removal (78.8%) among all the types of wetlands.

Floating treatment wetlands (FTWs)

Floating treatment wetlands (FTW) are relatively new and evolving treatment practices and employing a floating mat that sustains and supports rooted emergent macrophytes (Tanner & Headley, 2011). FTW systems may represent a significant opportunity to retrofit existing retention ponds by combining the functions of CWs and conventional retention ponds. One of the main advantages of FTWs over conventional sediment-rooted wetlands is their ability to cope with the highly variable nature of hydrologic and pollutant input that is typical for event-driven stormwater systems (Kerr-Upal et al., 2000). This feature also enables FTW systems to be designed as extended detention basins so that large runoff events can be captured and released slowly. As shown in Table 5, FTW systems can achieve comparable removal efficiencies for NH_4–N (61.9%), NO_3^-–N (64.5%), and TN (46.2%), as compared to FWS CW systems (64.8% for NH_4–N, 43.7% for NO_3^-–N and 53.9% for TN). However, they show relatively lower removal efficiencies for COD (33.7%) and TP (43.3%) as compared to FWS CWs (60.4% for COD and 57.1% for TP, respectively).

Nutrient removal in a wetland systems treating agricultural runoff

Nitrogen removal

Classic nitrogen removal mechanisms in wetland systems include sedimentation, adsorption, organic matter accumulation, ammonia volatilization, microbial assimilation, plant uptake, and nitrification/denitrification (Kadlec & Wallace, 2009). Nitrification/denitrification is considered as the predominant pathway for nitrogen removal (Vymazal, 2007). Novel nitrogen removal routes such as anaerobic ammonium oxidation (Anammox) where ammonium is directly oxidized to nitrogen gas by nitrite under anaerobic conditions, has been demonstrated in agricultural treatment wetlands (Tao et al., 2012; He et al., 2012). As shown in Fig. 2, the

Fig. 2 The mean concentration-based removal efficiencies in different types of wetland systems receiving agricultural runoff: **a** TSS and TP; **b** BOD_5 and COD; and **c** NH_4^-–N, NO_3^-–N, and TN. “FWS” denotes free water surface; “HSSF” denotes horizontal subsurface flow; “VSSF” denotes vertical subsurface flow; “HS” denotes hybrid system; “FTW” denotes floating treatment wetland. Data is expressed as the mean number of removal efficiencies ($n = 13$ for FWS CWs; $n = 17$ for HSS CWs; $n = 8$ for VSSF CWs; $n = 15$ for hybrid system; $n = 3$ for FTW). *Vertical bars* show standard deviation of the means

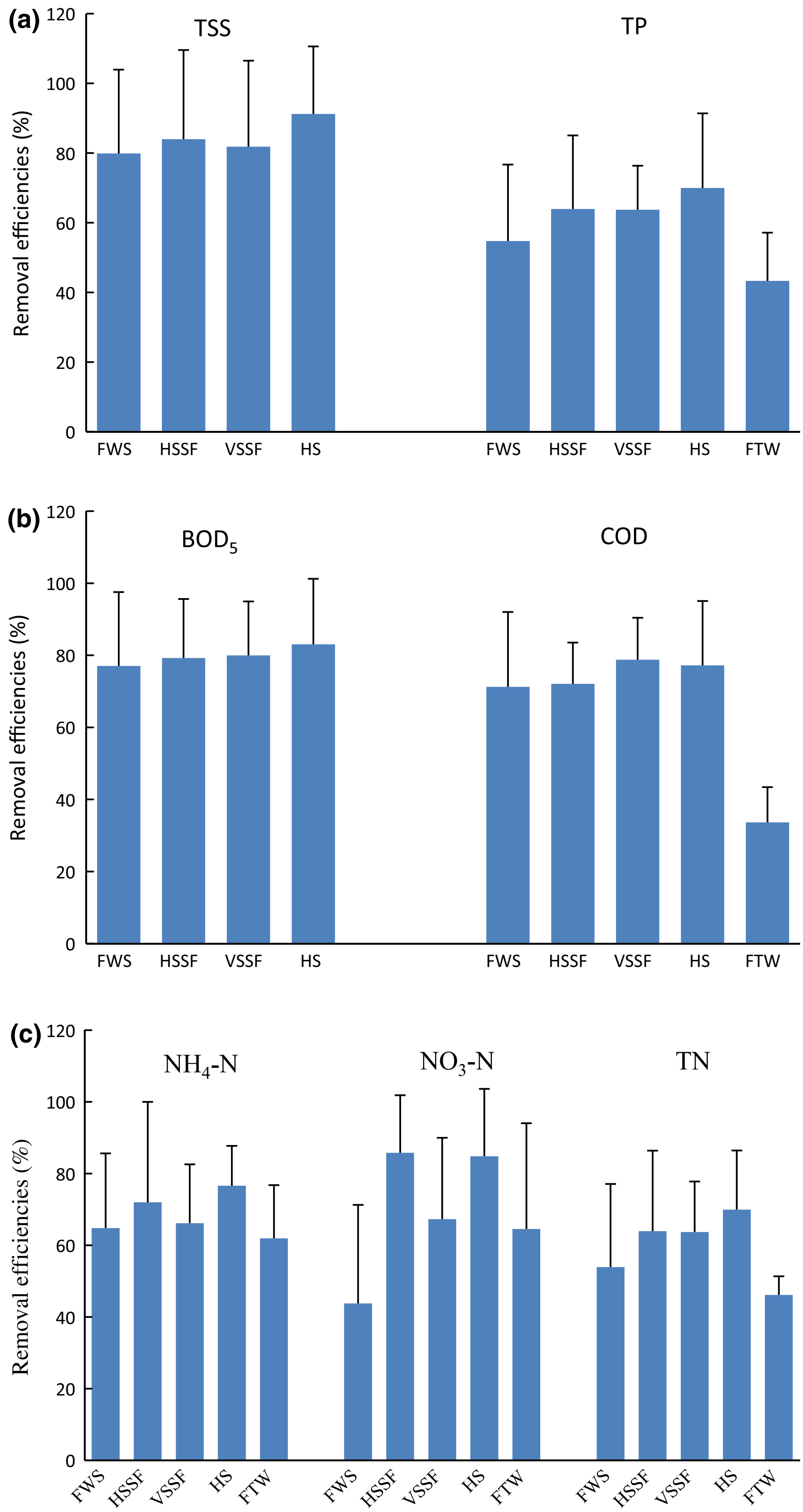

Table 2 A summary of the wetland design/operation and treatment efficiency of horizontal subsurface flow constructed wetlands (HSSF CWs) for treating agricultural runoff

	Type of wastewater	Removal performance							Wetland design and operation				Reference
		TSS	BOD_5	COD	NH_4–N	NO_3–N	TN	TP	Dimension (m × m×m) (L × W × D)	Plant species	Hydraulic loading rate (m^3 $days^{-1}$)	Hydraulic retention time (days)	
Heilongjiang, China									–				Gao and Hu (2012)
Effluent value (mg l^{-1})	Rural sewage	–	–	22.4	8.58	–	–	2.09		*Iris tectorum (Maxim.)*	2,500 l $days^{-1}$	24	
Removal efficiencies (%)		–	–	85.0	71.1	–	–	36.5		*Oxalis violacea* L.			
Load removal (g/days)		–	–	317.3	52.5	–	–	3.03					
Dianchi, China									2,800 m^2				Lu et al. (2009a, b)
Effluent value (mg l^{-1})	Agricultural runoff	–	–	–	–	–	–	0.36		*Phragmites australis* (Cav.)	12.7 cm day^{-1}	2	
Removal efficiencies (%)		–	–	–	–	–	–	59.0					
Shanghai, China									1.5 × 0.3 × 0.2				
Effluent value (mg l^{-1})	Dairy WW	–	–	45.1	6.5	–	11.9	0.27		*Bryum muehlenbeckii (Rota.)*	5.1 l $days^{-1}$	6.5	
Removal efficiencies (%)		–	–	87	88	–	80	91					
Load removal (g/m^2/days)		–	–	0.34	0.05		0.05	0.003					

Table 2 continued

	Type of wastewater	Removal performance							Wetland design and operation				Reference
		TSS	BOD_5	COD	NH_4–N	NO_3–N	TN	TP	Dimension (m × m×m) (L × W × D)	Plant species	Hydraulic loading rate (m^3 $days^{-1}$)	Hydraulic retention time (days)	
Reggio Emilia, Italy									12 × 6 × 1				Mantovi et al. (2003)
Effluent value (mg l^{-1})	Dairy WW	60	28	98	24.5	0.5	33.3	5.0		*Phragmites australis* (Cav.)	6.3	–	
Removal efficiencies (%)		90.8	93.7	91.9	–	–	48.5	60.6					
Load removal (g/m^2/days)		51.82	36.4	97.28	–	–	2.74	0.68					
Pingtung, Taiwan									9.5 × 2.6 × 0.7				Lee et al. (2004)
Effluent value (mg l^{-1})	Swine effluent	21	39	190	144	1.7	156	21		*Eichhornia crassipes* (Mart.)	–	8.5	
Removal efficiencies (%)	Secondary	96	91	84	22	54	24	47					
Maryland, USA									0.055 ha				Schaafsma et al. (1999)
Effluent value (mg l^{-1})	Dairy farm WW	–	–	–	–	–	–	–		*Typha latifolia* L.	–	6 weeks	
Removal efficiencies (%)		96	97	–	98	82	98	96		*Lemma minor* L.			

Table 2 continued

	Type of wastewater	Removal performance							Wetland design and operation				Reference
		TSS	BOD_5	COD	NH_4–N	NO_3–N	TN	TP	Dimension (m × m×m) (L × W × D)	Plant species	Hydraulic loading rate (m^3 $days^{-1}$)	Hydraulic retention time (days)	
Montreal, Canada									1.2 × 0.8 × 0.3				Quellet-Plamondon et al. (2006)
Effluent value (mg l^{-1})	Trout farm effluent	–	–	–	6 ± 4	0.3 ± 0.3	24 ± 7	–		*Phragmites australis* (Cav.)	30 l m^{-2} $days^{-1}$	4	
Removal efficiencies (%)		84.2	–	56.2	–	–	39.3	–		*Typha angustifolia* L.			
Load removal (g/m^2/days)		–	–	–	–	–	0.47	–					
NSW, Australia									4 × 1 × 0.5				Headley et al. (2001)
Effluent value (mg l^{-1})		–	–	–	0.008	0.03	0.41	0.025		*Phragmites australis* (Cav.)	–	5	
Removal efficiencies (%)		–	–	–	93.8	97.7	86.8	94.4					
Effluent value (mg l^{-1})	Plant nursery runoff	–	–	–	0.008	0.001	0.38	0.13				4	
Removal efficiencies (%)		–	–	–	98.5	97.8	90.4	78.2					
Effluent value (mg l^{-1})		–	–	–	0.014	0.002	0.54	0.15				2	
Removal efficiencies (%)		–	–	–	97.3	97.4	84.4	65.9					

Table 2 continued

	Type of wastewater	Removal performance							Wetland design and operation				Reference
		TSS	BOD_5	COD	NH_4–N	NO_3–N	TN	TP	Dimension (m × m×m) (L × W × D)	Plant species	Hydraulic loading rate (m^3 $days^{-1}$)	Hydraulic retention time (days)	
New South Wales, Australia									0.6 × 0.37 × 0.3				Huett et al. (2005)
Effluent value (mg l^{-1})	Plant nursery runoff	–	–	–	–	–	–	–					
Removal efficiencies (%)		–	–	–	–	97	90	96		*Phragmites australis* (Cav.)	–	7.0	
New York, USA									55 m^2				Hill et al. (2000)
Effluent value (mg l^{-1})	Barnyard manure	–	–	–	–	–	–	6.1		*Kandelia candel L.*	8–30 cm day^{-1}	–	
Removal efficiencies (%)		–	–	–	–	–	–	52.7					
New Zealand									9.2 × 2 × 0.6				Tanner et al. (1995)
Removal efficiencies (%)	Dairy farm WW	78	80	–	42	–	52	45				3	
Removal efficiencies (%)		83	90	–	59	–	71	67		*Schoenoplectus validus* (Vahl.)	–	5.5	
Removal efficiencies (%)		76	92	–	71	–	75	74				7	
Vermont, USA									225 m^2 × 0.6 m				Tunçsiper et al. (2015)
Effluent value (mg l^{-1})	Swine effluent	–	–	–	–	–	–	–		*Schoenoplectus fluviatilis (Torr.)*	37.5	6	
Removal efficiencies (%)	Secondary	81–96	68–95										

Table 2 continued

	Type of wastewater	Removal performance							Wetland design and operation				Reference
		TSS	BOD_5	COD	NH_4–N	NO_3–N	TN	TP	Dimension (m × m×m) (L × W × D)	Plant species	Hydraulic loading rate (m^3 $days^{-1}$)	Hydraulic retention time (days)	
Germany									23.6 m^2				Sindilariu et al. (2008)
Effluent value ($mg\ l^{-1}$)	Trout farm effluent	0.71	0.78	4.66	0.13	5.31	5.53	0.15		–	0.9 l/s		
Removal efficiencies (%)		90.1	88.7	67.2	82.9	−9.7	10	40					
Nova Scotia, Canada									25 × 10 × 1.5				Vander Zaag et al. (2008)
Effluent value ($mg\ l^{-1}$)	Dairy WW	–	–	12.4	1.3	0.2	–	6.7		–	250	4	
Removal efficiencies (%)				95.8	95.2	–	–	69.6					
Load removal (g/m^2/days)		–	–	282.8	25.78			15.3					
Montreal, Canada									1.3 × 0.8 × 0.3	*Phragmites australis* (Cav.)	30, 60, 90 l m^{-2} day^{-1}		Maltais-Landry et al. (2007)
Effluent value ($mg\ l^{-1}$)	Fish farm WW	–	–	–	–	–	–	–		*Typha angustifolia L.*		–	
Removal efficiencies (%)		76–91	–	83–92	–	–	64–74	35–40					
Aosta Valley, Italy									200 m^2 × 1 m				Gorra et al. (2014)
Effluent value ($mg\ l^{-1}$)	Dairy WW	–	71	–	15	4	107	6		*Phragmites australis* (Cav.)	–	–	
Removal efficiencies (%)		–	92	–	17	–	27	40		*Typha latifolia L.*			

Table 2 continued

	Type of wastewater	Removal performance							Wetland design and operation				Reference
		TSS	BOD_5	COD	NH_4–N	NO_3–N	TN	TP	Dimension (m × m×m) (L × W × D)	Plant species	Hydraulic loading rate (m^3 $days^{-1}$)	Hydraulic retention time (days)	
Pontevedra, Spain									100 m^2		24.8 mm $days^{-1}$		De la Varga et al. (2013)
Removal efficiencies (%)		73.4	61.3	54.4						–	36.7 mm $days^{-1}$		
Removal efficiencies (%)	Winery WW	73.5	41.1	40.4	–	–	–	–	0.35–0.65 m (H)		36.2 mm $days^{-1}$		
Removal efficiencies (%)		76.3	44.8	43.2									

highest average TN removal efficiency is observed in hybrid systems (73.2%), followed by VSSF CWs (71.7%). Hybrid systems also exhibit high NH_4^+–N removal (76.6%). In contrast, CWs with HSSF flow have a high potential for NO_3^-–N removal and exhibit highest NO_3^-–N removal efficiency of 85.8%.

Lu et al. (2009b) investigated nitrogen removal from agricultural runoff by a full-scale CW in China, and reported removal efficiencies of NH_3^+–N, NO_3^-–N, and TN of 63.6, 60.9, and 65.4%, respectively. 14% of nitrogen input was incorporated into the plant biomass, while 47% of nitrogen input was removed by nitrification/denitrification/ammonia adsorption/bacteria. The authors concluded that the function of plants, the warm climate and the intermittent inflow in the wetland played significant roles in the good nitrogen removal capacities; inflow load significantly affected both TN and ammonia removal efficiencies. Lin et al. (2005) evaluated the performance of a CW consisting of a FWS and a HSSF CW treating aquaculture effluent under high HRTs, and reported that both total ammonium nitrogen (TAN) and nitrite (NO_2–N) levels were not considerably reduced from the influent to effluent of FWS cell ($P > 0.05$). However, significant decrease ($P < 0.05$) in TAN and NO_2–N level was consistently observed across the SSF cell. Consequently, overall TAN and NO_2–N reduction percentage of the FWS-HSSF wetland averaged to 66 and 94%, respectively, leading to average removal rates of 0.34 g m^{-2} $days^{-1}$ for TAN and 0.58 g m^{-2} $days^{-1}$ for NO_2–N, respectively. Poach et al. (2004) investigated the ability of marsh-pond-marsh CWs to treat swine wastewater in North Carolina, USA, and reported that TN removal efficiency ranged from 37 to 51% at TN loading rates of 2–51 kg Ha^{-1} $days^{-1}$, and ammonia volatilization contributed to greater than 50% of TN removal.

Phosphorus removal

Phosphorus removal in CWs is closely associated with physicochemical and hydrological properties of the filter materials (Dunne et al., 2005b; Healy et al., 2007). Phosphorus sorption and retention in wetland soils is considered as long-term mechanisms for phosphorus removal; in acid soils, phosphorus is fixed as Al and Fe phosphates, while in alkaline soils, phosphorus sorption is governed by Ca and Mg components (Reddy et al., 1999). In general, microbial removal and plant uptake

Table 3 A summary of the wetland design/operation and treatment efficiency of VSSF CWs for treating agricultural runoff

	Type of wastewater	Removal performance							Wetland design and operation				References
		TSS	BOD_5	COD	NH_4–N	NO_3–N	TN	TP	Dimension (m × m×m) (L × W × D)	Plant species	Hydraulic loading rate (m^3 day^{-1})	Hydraulic retention time (day)	
Wuxi, China									2.0 × 2.0 × 1.0				He et al. (2006)
Effluent value ($mg\ l^{-1}$)	Livestock WW	96	61.8	–	32.9	41.3	–	–		*Phragmites communis* (Trin.)	0.4	–	
Removal efficiencies (%)	Secondary	77.1	81.3	–	61.7	66.6	48.9	–		*Phragmites australis* (Cav.)			
Load removal ($g/m^2/d$)		32.32	46.9	–	5.3	8.23	–	–					
Wuxi, China													Li et al. (2007)
Effluent value ($mg\ l^{-1}$)	Aquaculture WW	–	–	–	–	–	–	–		*Ictalurus punctatus (Rafinespue)*			
Removal efficiencies (%)		81.9	70.5	–	61.5	68.0	–	20		*Megalobrama amblycephala (Yih.)*	–	–	
USA													Behrends et al. (2000)
Effluent value ($mg\ l^{-1}$)	Aquaculture WW	–	–	–	–	–	–	–		–	–	–	
Removal efficiencies (%)		–	99	99	86	–	96	84					
USA													Travis et al. (2012)
Effluent value ($mg\ l^{-1}$)	Farm WW	–	–	–	–	–	–	–	60 m^2 × 0.6 m	*Canna indica* L.	–	5	
Removal efficiencies (%)		97	96	94	–	–	73	73	4 m^2 × 0.4 m	*Salvia arizonica* A.			
Flanders, Belgium													

Table 3 continued

	Type of wastewater	Removal performance							Wetland design and operation				References
		TSS	BOD_5	COD	NH_4–N	NO_3–N	TN	TP	Dimension (m × m×m) (L × W × D)	Plant species	Hydraulic loading rate (m^3 day^{-1})	Hydraulic retention time (day)	
Effluent value (mg l^{-1})	Dairy WW	–	–	–	–	–	–	–					Rousseau et al. (2004)
Removal efficiencies (%)		98	–	94	–	–	52	70		–	–	–	
Canada													
Effluent value (mg l^{-1})	Aquaculture WW	–	–	–	–	–	–	–		–	–	–	Snow et al. (2010)
Removal efficiencies (%)		39	53	56	–	–	–	58					
Ontario, Canada									11.8 × 8.6 × 1.2				Rozema et al. (2016)
Effluent value (mg l^{-1})	Winery	2.7	–	6	0.18	2.03	0.45	–		*Typha latifolia* L.	22.3 mm/d	–	
Removal efficiencies (%)		98	–	98.9	72.7	–	88.7	83		*Schoenoplectus tabernaemontani* (*C.C.Gmel*)			
Ankara, Turkey									1.0 × 0.5 × 0.4				Yalcuk & Ugurlu (2009)
Removal efficiencies (%)	Agricultural runoff	–	–	27.3	62.3	–	–	52.6		*Typha latifolia* L.	10 l day^{-1}	11	
Removal efficiencies (%)		–	–	30.6	48.9	–	–	57.9				8	

Table 4 A summary of the wetland design/operation and treatment efficiency of hybrid systems for treating agricultural runoff

	Type of waste-water (WW) & stage of treatment	Removal performance							Wetland design and operation				Reference
		TSS	BOD_5	COD	NH_4–N	NO_3–N	TN	TP	Dimension (m × m × m) (L × W × D)	Plant species	Hydraulic loading rate (m^3 $days^{-1}$)	Hydraulic retention time (days)	
Hokkaido, Japan													Zhang et al. (2016)
Effluent value (mg l^{-1})	Piggery and dairy WW	23 ± 10	106 ± 81	382 ± 164	14 ± 10	–	34 ± 13	13 ± 5	1472 m^2	*Phragmites australis* (Cav.)	0.7 cm $days^{-1}$	–	
Total removal (%)		84–97	94–98	91–96	40–85	–	70–86	71–90	3048 m^2		1.0 cm $days^{-1}$		
Taiwan									5.0 × 1.0 × 0.8				Lin et al. (2002)
Effluent value (mg l^{-1})	Agricultural WW	–	–	–	0.11	0.07	0.18	3.53		*Phragmites australis* (Cav.)	1.35 cm $days^{-1}$	–	
Total removal (%)		–	–	–	86.3	97.4	95.4	32.0					
Pontevedra, Spain													Serrano et al. (2011)
Effluent value (mg l^{-1})	Winery WW	17	279	448	12.5	–	25.2	1.9	8.3 × 6.0 × 1.4	*Phragmites australis* (Cav.)	17.6	–	
Total removal (%)		87.0	67.5	71.6	–	–	64.0	57.6	10 × 10 × 0.35	*Juncus effuses* L.			
Load removal (g/m^2/day)													
Hokkaido, Japan													Sharma (2013)
Effluent value (mg l^{-1})	Dairy WW	0.1	1.32	2.8	0.16	–	0.26	0.04	160 m^2	*Typha* sp.	7.3–7.9 cm $days^{-1}$	–	
Total removal (%)		97.9	89.3	89.8	72.4	–	78.5	77.8	336 m^2	*Lemna* sp.			

Table 4 continued

	Type of waste-water (WW) & stage of treatment	Removal performance							Wetland design and operation				Reference
		TSS	BOD_5	COD	NH_4–N	NO_3–N	TN	TP	Dimension (m × m × m) (L × W × D)	Plant species	Hydraulic loading rate (m^3 $days^{-1}$)	Hydraulic retention time (days)	
Nanjing, China									0.3 × 0.4 × 0.2				Zhu et al. (2012)
Effluent value (mg l^{-1})	Livestock WW	–	–	–	–	–	–	–		*Cyperus alternifolius* L.	0.125 m $days^{-1}$	–	
Total removal (%)		86.5	–	62.4	95	–	–	80		*Phragmites* spp.			
Anne Valley, Ireland									1208–2435 m^2				Zhang et al. (2009)
Effluent value (mg l^{-1})	Farmyard runoff	15.1	11.2	55.2	0.37	–	–			–	–	–	
Total removal (%)		91.1	98.0	96.2	99.1	–	–	92.7					
UK									0.76 ha				Mustafa et al. (2009)
Effluent value (mg l^{-1})	Farmyard runoff	16.3	12.9	75.5	0.37	0.99	–	0.94		–	–	–	
Total removal (%)		93.7	97.6	94.9	99	74	–	91.8					
Ontario, Canada													Gottschall et al. (2007)
Effluent value (mg l^{-1})	Dairy WW	–	–	–	–	–	–	–	–	*Typha latifolia* L.	–	–	
Total removal (%)		–	–	–	82	–	72	58	–	*Typha angustifolia* L.			
Thailand													Kantawanichkul (2005)

Table 4 continued

	Type of waste-water (WW) & stage of treatment	Removal performance							Wetland design and operation				Reference
		TSS	BOD_5	COD	NH_4–N	NO_3–N	TN	TP	Dimension (m × m × m) (L × W × D)	Plant species	Hydraulic loading rate (m^3 $days^{-1}$)	Hydraulic retention time (days)	
Effluent value (mg l^{-1})	Pig farm WW	–	–	–	–	–	–	–	–	*Cyperus flabelliformis (Rottb.)*	2–8 cm $days^{-1}$	–	
Total removal (%)		92	–	75	90	–	85	75	–				
Northern Ireland									12,510 m^2				Forbes et al. (2011)
Effluent value (mg l^{-1})	Farmyard water	9	–	–	0.06	–	0.49	1.0		*Phragmites australis* (Cav.)	–	–	
Total removal (%)		99	–	–	99	–	92.8	95		*Typha latifolia* L.			
Michigan, USA									0.84 × 0.66 × 0.3				Adhikari et al. (2015)
Effluent value (mg l^{-1})	Dairy	–	–	–	–	–	–	–		*Schoenoplectus*	–	22.5	
Total removal (%)	Wastewater	–	–	52	–	–	39	32		*Tabernaemontani (C.C.Gmel.)*			
Vietnam									3.7 × 0.9 × 0.3				Konnerup et al. (2011)
Effluent value (mg l^{-1})	Fishpond effluent	–	15–27	–	–	–	4.5–8.2	1.0–2.5		*Canna generalis L.H. Bailey*	750–3000 mm $days^{-1}$	–	
Total removal (%)		–	50	50	–	–	41	44				–	
Flanders, Belgium									4500 m^2				Boets et al. (2011)
Effluent value (mg l^{-1})	Manure WW	–	–	–	–	–	–	–		–	1.0 m^{-2} $days^{-1}$		
Total removal (%)		–	–	90.8	–	–	88.5	92.8					
Morgex, Italy													Comino et al. (2011)

Table 4 continued

	Type of waste-water (WW) & stage of treatment	Removal performance							Wetland design and operation				Reference
		TSS	BOD_5	COD	NH_4–N	NO_3–N	TN	TP	Dimension (m × m × m) (L × W × D)	Plant species	Hydraulic loading rate (m^3 $days^{-1}$)	Hydraulic retention time (days)	
Effluent value (mg l^{-1})	Cheese factory	719	1160	2870	35.4	5.2	39.6	13	$1472m^2 \times 0.9m$	–	0.05 m $days^{-1}$	5	
Total removal (%)	WW	80.3	80.3	80.4	39.1	81.7	39.8	–	$344m^2 \times 0.95m$				
Carolina, USA									3.7 × 33.5				Poach et al. (2003)
Effluent value (mg l^{-1})	Swine manure	–	–	–	22	4	33	–		*Scripus cyperinus* L.	–	12	
Total removal (%)					73	88	78			*Typha angustifolia* L.			
Effluent value (mg l^{-1})					8	6	19			*Typha latifolia* L.			
Total removal (%)					91	80	85			*Juncus effuses* L.			

Table 5 A summary of the wetland design/operational parameters and treatment efficiency of floating treatment wetlands (FTW) for treating agricultural runoff

	Type of wastewater (WW)	Removal performance							Wetland design and operation				References
		TSS	BOD_5	COD	NH_4–N	NO_3–N	TN	TP	Dimension (m × m × m) (L × W × D)	Plant species	Hydraulic loading rate (m^3 $days^{-1}$)	HRT (days)	
Taihu Lake, China									2.0 × 1.0 × 0.75				Yang et al. (2008)
Effluent concentration (mg l^{-1})	Agricultural runoff	–	–	36.70	0.54	1.42	2.86	1.34		*Oenanthe javanica (Blume) DC.*	–	1	
Total removal (%)		–	–	47	60	71	64	13					
Effluent concentration (mg l^{-1})		–	–	40.2	1.31	0.05	2.95	1.16					
Total removal (%)		–	–	24	–	97	35	15			–	2	
Yixing, China									1.2 × 0.8 × 1.2				Yang et al. (2008)
Effluent concentration (mg l^{-1})	Agricultural runoff	–	–	–	–	–	2.42	–		*Oenanthe javanica (Blume) DC.*	–	1–3	
Total removal (%)		–	–	14–47	71–91	–	31–64	8–15					
Jiaxing, Zhejiang, China									–				Zhao et al. (2012)
Effluent concentration (mg l^{-1})	River water	–	–	–	1.66	3.05	4.92	–		*Hydrocharis dealbata (Miq.)* *Eichhirnia crasslpes (EC.)*	–	–	
Total removal (%)					44.8	25.6	39.6	43.3					

are responsible for phosphate removal, while precipitation and adsorption are responsible for the removal of all phosphorus forms (Kadlec & Knight, 1996). As shown in Fig. 2, the highest average TP removal efficiency is observed in hybrid systems (69.9%), followed by HSSF CWs (63.9%). SSF CW exhibits better TP removal efficiencies (63.9% for HSSF CWs and 63.7% for VSSF CWs) than those in FWS CWs (54.7%). However, compared to FTW systems (43.3%), FWS CWs (54.7%) show a high potential for TP retention and adsorption.

Large variations in phosphorous reduction in farm wetland system have been observed in previous studies. Forbes et al. (2011) evaluated performance of a CW for treating farmyard wastewater and reported overall phosphorus reduction of 95% despite the high phosphorus loading rate of 8.7-31.8 g m^{-2} $year^{-1}$, while Braskerud (2002) reported low TP retention of 21–44% in in small FWS wetlands treating agricultural runoff in Norway. Healy et al. (2007) concluded that 65–95% of phosphorus may be removed at loading rates of less than 5 g TP m^{-2} $year^{-1}$. The authors indicated that phosphorus uptake by macrophytes provided an initial removal mechanism but only provided short-term phosphorus storage, while 35–75% of phosphorus stored was eventually released back into the water upon dieback of algae and microbes, as well as plant residues. Zhu et al. (2012) investigated phosphorous removal efficiencies in multi-level mineralized refuse-based CWs and reported TP removal rates by ladder-type CWs (horizontal flow) and tower-type CWs (vertical flow) were as high as 86.7 and 98.6%, respectively. The authors indicated that the high phosphorus adsorption capacities were attributed to the high CaO, Al_2O_3 and Fe_2O_3 contents of the mineralized refuse, as phosphorus removal is basically a function of the adsorption of ions in the mineralized refuse matrix.

Optimization of design and operational conditions

Hydraulic loading rate (HLR)

The determination of a proper HLR/HRT is crucial for treatment performance in wetland systems (Kadlec and Wallace, 2009). In general, for a given CW geometry, a higher hydraulic loading rate (HLR) implies a lower HRT. Sindilariu et al. (2008) examined the effects of different HLR on the treatment efficiency of a SSF CW treating trout farm effluent, and revealed that the treatment efficiency for particulate nutrients and ammonia nitrogen increased with decrease in HLRs. The authors indicated that the highest treatment efficiencies were detected at the lowest HRT of 0.9 l s^{-1} for TN (10.0%), BOD_5 (88.7%), COD (67.2%), and TSS (90.1%). The treatment efficiencies for the dissolved nutrients (NO_2^-–N, NO_3^-–N, PO_4^{3-}–P) showed no differences among the HRTs. At the highest hydraulic load of 3.9 l s^{-1}, the treatment efficiency for NH_4–N (61.2%), BOD_5 (71.5%), and TSS (84.6%) was the lowest compared to the other two hydraulic loads (1.8 and 0.9 l s^{-1}). As for TN (5.5%) and COD (54.6%), a significant difference was found only in comparison to the lowest load of 0.9 l s^{-1}. In a study investigating the removal of nutrients from plant nursery irrigation runoff in HSSF CWs, Headley et al. (2001) observed a declining distinct trend, with mean TP reductions of 94.4, 78.2, and 65.9% for the 5, 4, and 2 days HRTs, respectively. With respect to nitrogen removal, the authors stated that HRT significantly ($P < 0.05$) affected TN and NH_4^+–N removal, and the reduction in TN was greater ($P < 0.05$) for 4-days HRT (90.4%) than 2-days HRT (84.4%). The authors further concluded that with the lowest removal for NH_4^+–N being achieved at the longest HRT (5-days), the additional time allowed further breakdown of organic N into NH_4–N. Yang et al. (2008) investigated the purification of nitrate-rich agricultural runoff by a hydroponic system, and stated that the variation in nitrate removal performance was attributed to the different HLR levels. An appropriate HLR of 0.18–0.27 m $days^{-1}$ could achieve mean NO_3^-–N removal of 91–97% at 2 and 3-days HRT, while a higher HLR of 0.54 m $days^{-1}$ resulted in mean nitrate removal of 71% at 1-day HRT. This hydroponic system exhibited an average TN removal percentage of 31–35% at 1- and 2-days HRTs that increased sharply up to 64% at 3-days HRT.

Seasonal variation

CWs have been successfully applied to mitigating environmental pollution in warm climates; however, their performance in cold climates is still questionable.

Previous studies indicated that the contaminant treatment performance in wetlands commonly declines as temperature decreases (Newman et al., 2000; Travis et al., 2012), because some key microbial processes in CWs, such as nitrification/denitrification and organic matter mineralisation, usually depend on temperature (Tanner et al., 2005). In general, the efficiency of treatment decreases at low temperature primarily due to reduced biotic activity. Truu et al. (2009) indicated that tropical conditions could enhance the removal of contaminants, as microorganisms living in the CWs usually reach their optimal activity at temperatures of 15–25°C. Vymazal (2005) reported that the optimum temperature for nitrification ranges from 25 to 35°C in pure cultures and from 30 to 40°C in soils.

Newman et al. (2000) investigated the seasonal performance of a wetland constructed to process dairy wastewater in Connecticut, USA, and indicated that the mean removal efficiencies for TP (54%), NH_3–N (7%), NO_3–N (54%) and total Kjeldahl nitrogen (TKN) (29%) during winter were significantly lower than during summer (68, 31, 70, and 55%, respectively). Travis et al. (2012) evaluated the seasonal variation on treatment of farm wastewater in a VSSF CW in Israel, and reported that winter temperatures of 10–15°C were typically associated with COD removal of 40–60%, whereas summer temperatures of 25–30°C were associated with 80–90% COD removal. Furthermore, nutrient fluxes may be associated with plant growth and senescence varies markedly with season. Reyes and Vidal (2015) assessed the effect of variation in the seasonality on the operation of a FWS CW for treatment of swine wastewater, and reported that plant uptake accounted for 14.9% of the TN removed, with the vegetative peak in summer at a nitrogen loading rate of 25.3 ± 0.3 kg TN ha^{-1} -days^{-1}. Borin et al. (2001) investigated biomass and seasonal nitrogen dynamics in a FWS CW for treating agricultural wastewater in Italy and showed that, in both *Phragmites australis* (Cav.) and *Typha latifolia* L., nitrogen reached maximum levels in summer and minimum levels in winter. With respect to phosphorous removal, Dunne et al. (2005a) reported that mass retention of TP and SRP in a wetland varied, ranging from 5 to 84% owing to seasonal variations.

The presence of macrophytes

Plants play a significant role in contaminant removal and can enhance treatment efficiency by aiding settling of particulates, adsorption of solutes, transporting gases and solutes from shoots to roots, uptake and storages of inorganic/organic pollutants, release of oxygen and exudates, and promoting microbial population growth and diversity (Greenaway & Woolley, 2001; Stottmeister et al., 2003). It has been generally accepted that planted wetlands outperform unplanted filters, and there is a positive correlation between plant uptake and nutrient removal from agricultural treatment wetlands (Comin et al., 1997; Huett et al., 2005; Gottschall et al., 2007). Moreover, harvesting of the emergent macrophytes has a pronounced effect on the growth and nutrient uptake rates, possibly because nutrient uptake and growth rates are higher in young vegetation stands (Greenaway & Woolley, 2001).

Huett et al. (2005) compared nitrogen and phosphorus removal from plant nursery runoff in vegetated SSF CWs and unvegetated filters in Australia, and reported that removal efficiency for planted wetlands increased from 63.4 and 69.1% for TN and NO_3–N, respectively, after 30 days to >90 and >97% after 140 days, while unplanted filters had low nitrogen removal efficiency (<10% for TN and NO_3–N) after 140 days. In addition, removal efficiency for planted wetlands varied from 84.5 and 87.5% for TP and PO_4–P, respectively, after 30 days to >96% for both after 155 days. Unplanted filters removed similar amounts of TP after 30 days (60.4% for TP and 63.9% for PO_4–P) which declined to 38% after 140 days. Similarly, Comin et al. (1997) explored uptake capacity of dissolved inorganic nitrogen in wetlands planted with *Typha latifolia* in Spain, and reported that plant uptake accounted for over 66% of nitrogen removal. Another study by Greenaway and Woolley (2001), found that 27–47% of TN removal was due to plant uptake. With respect to phosphorous removal, Huett et al. (2005) stated that phosphorous removal varied from 78.4 to 99.5% (mostly >90%) in planted wetlands, compared with –76.4 to 51.7% in unplanted filters. Plant uptake accounted for 86% of P removal, while roots and rhizomes were the dominant sink (67%).

Pretreatment

Successful agricultural wetland design must include adequate pretreatment to protect the health of wetland biota and to meet water quality goals (Knight et al., 2000). The primary objective of pretreatment is aimed at screening, skimming, and settling of suspended solids, fats, and oil and grease (O&G) and other floatable substances, to avoid clogging that leads to a reduction of the infiltration capacity of the gravel bed (Travis et al., 2012; De la Varga et al., 2013). Thus, it is generally accepted that the application of proper wastewater pretreatment is essential for sustainable long-term operation of treatment wetlands. The most common form of pretreatment is a settling basin or anaerobic lagoon. Previous published results indicate that 50–75% of the total BOD_5 and TSS in raw livestock wastewaters is typically removed through pretreatment (Knight et al., 2000). Furthermore, according to the level of organic matter removal, anaerobic pretreatment provided a 30–60% reduction in the wetland area requirement (Álvarez et al., 2008).

Travis et al. (2012) evaluated the effect of seasonal variations on treatment of oil-rich farm wastewater in a VSSF CW in Israel. The system included a series of anaerobic baffled tanks as pretreatment to facilitate solid retention and initial organic matter degradation. The authors reported that TSS was reduced 85% from 1100 to 170 mg l^{-1}, and O&G was reduced 92% from 520 to 44 mg l^{-1}. Accordingly, it was calculated that approximately 700 kg $year^{-1}$ of dry solids were settled in the anaerobic tanks annually. TN was reduced approximately 21% in the anaerobic treatment to 77 mg l^{-1}. Similarly, De la Varga et al. (2013) carried out a long-term study of a hybrid wetland system including an anaerobic digester for winery wastewater treatment in Spain, and reported that the anaerobic digester removed 76.4% TSS, 26.3% COD, and 21.3% BOD_5 on average, reducing the maximum loading rate to the subsequent CWs. The authors revealed that the concentrations of suspended solids in the effluent were nearly constant and independent of the influent concentrations, while the TSS and volatile suspended solid (VSS) removal efficiencies increased with the influent concentration.

Effects of pH

pH value can also affect nitrogen removal in ammonium-rich agricultural treatment wetlands. Since the conventional nitrification-denitrification process is usually restricted by limited availability of oxygen and organic carbon (Kadlec and Wallace, 2009; Vymazal, 2007), anaerobic ammonium oxidation (Anammox), which uses nitrite to oxidize ammonium under anaerobic conditions, can significantly reduce the requirement for oxygen and organic carbon. In the Anammox process, partial nitrification (nitritation) of the ammonium to nitrite by ammonia-oxidizing bacteria (AOB) occurs first, and then the resulting ammonium and nitrite are converted to dinitrogen gas by Anammox bacteria (Strous et al., 1999; Bae et al., 2001).

Tao et al. (2012) evaluated the effects of pH on coupling nitritation and Anammox in biofilters treating dairy wastewater, and revealed that ammonium removal rate was significantly higher in the biofilter at pH 8.1 (7.8 g N m^{-3} days) than that in the other biofilter at pH 7.6 (6.4 g N m^{-3} days). The average TIN removal rate was higher in biofilters with pH value of 7.60 (7.7 g N m^{-3} days) than the one with pH value of 8.08 (7.0 g N m^{-3} days), demonstrating that a pH value of 8.1 enhanced nitritation–Anammox compared to a pH value of 7.6. The authors also indicated that the relative abundance of AOB and Anammox together was found to be higher in the former than in the latter accordingly. He et al. (2012) examined the effect of pH on simultaneous partial nitrification and Anammox in two FWS CWs receiving dairy wastewater, and revealed that the wetland with higher effluent pH values had a higher AOB abundance than the one with lower effluent pH. However, the authors further indicated that free ammonia concentration in wetland with higher pH was increased to 2.3–10.8 mg N l^{-1}, which could inhibit Anammox. A lower relative abundance of Anammox bacteria was detected in CW with higher pH, since free ammonia concentrations at 1.7–8.3 mg N l^{-1} have been reported to inhibit Anammox.

Artificial aeration

Available oxygen in CWs is an important factor for biodegradation of organic matter and transformation of ammonium–nitrogen, both of which are oxygen limited processes (Vymazal, 2007). Due to the insufficient oxygen supply through the surface air and the plant-mediated oxygen transfer, incomplete nitrification has been reported in CWs (Tanner and Kadlec, 2003; Huett et al., 2005). Artificial aeration has been used as an alternative solution to enhance treatment performance and represents a promising approach to improve removal efficiency in HSSF CWs for treating agricultural runoff, especially in winter when plants are dormant (Quellet-Plamondon et al., 2006; Maltais-Landry et al., 2007). Furthermore, artificial aeration prevents partially degraded organic matter from accumulating in the bed matrix (Maltais-Landry et al., 2007). However, injecting air into the SSF CW matrix requires energy input and leads to additional costs for operation and maintenance of the facility and is generally not regarded as desirable. Thus, aeration is only justified when its life cycle cost is sufficiently offset by the reduction in the capital cost by the net saving of reduced wetland area size. Nevertheless, aeration which has been widely applied to fish farms to maintain a high oxygen level is readily available for CWs to save capital cost (Quellet-Plamondon et al., 2006; Maltais-Landry et al., 2007).

Quellet-Plamondon et al. (2006) evaluated the contribution of artificial aeration on pollutant removal in CWs treating a reconstituted fish farm effluent. Artificial aeration reduced TKN effluent as mass loading for unplanted units (23 ± 19 mg days^{-1} in summer and 1 ± 8 mg days^{-1} in winter), compared a non-aerated system (57 ± 21 mg days^{-1} in summer and 54 ± 11 mg days^{-1} in winter). Artificial aeration also improved NH_4–N removal in both planted (4 vs. 1 mg days^{-1}) and uplanted (49 vs. 13 mg days^{-1}) units. However, the authors indicated that although artificial aeration exhibited a promising approach to improve removal efficiency in HSSF CWs especially in cold climates, the additional aeration did not fully compensate for the absence of plants, suggesting that the role of macrophytes goes beyond the sole addition of oxygen in the rhizosphere. Maltais-Landry et al. (2007) evaluated the effects of artificial aeration on removal efficiency in CWs treating fish farm wastewater and reported that units with artificial aeration removed significantly more TKN than non-aerated unit: 1.86 ± 0.55 g m^{-2} days^{-1} (aerated) versus 1.17 ± 0.49 g m^{-2} days^{-1} (non-aerated). However, there was no significant effect of artificial aeration on TSS and COD removal on percentage basis. In contrast, Macphee et al. (2009) evaluated a diffused air aeration system for a CW (~ 100 m^2) receiving dairy wastewater in Nova Scotia, Canada, and revealed that artificial aeration significantly increased TKN and NH_3–N mass reduction. However, aeration did not significantly affect the removal of BOD_5, TSS, NO_3–N, and TP. The authors suggested that the benefits of wetland aeration were not great enough to warrant its widespread adoption for small-scale agricultural systems.

Effluent recirculation

Effluent recirculation has been considered as an operational modification to improve treatment efficiency in wetland systems by taking part of the effluent and transferring it back to the inflow (Sun et al., 2003, 2005). This operation can bring benefits to wetland treatment performance (e.g., nitrification) by enhancing interactions between pollutants in the wastewater and microorganisms attached on the roots and gravel, particularly during oxygen depletion (Sun et al., 2003; Lavrova and Koumanova, 2010). Effluent recirculation can also improve denitrification mechanisms in wetland systems, due to the enhanced mass transfer of oxygen and substrates to biofilms, and dilution of influent which distributes substrates more evenly through the system (Zhao et al., 2004). As the recirculation increases hydraulic loading, it may not be suitable for horizontal flow systems. However, in gravel-based vertical flow reed beds the hydraulic conductivity is much greater, making effluent recirculation a practicable operation (Sun et al., 2003).

Sun et al. (2003) investigated the effect of effluent recirculation on the performance of a vertical reedbed system treating agricultural runoff, and reported that recirculation considerably improved the removal of BOD_5, COD, TSS, and NH_4–N. Recirculation reduced the BOD_5 by 96.7% from an average of 427–14 mg l^{-1}, whereas the average percentage reduction was only 71.8% before recirculation was used. The removal percentage of NH_4–N was increased by 51% after effluent recirculation was adopted. A large amount of NO_3–N was generated with recirculation, but without recirculation there was virtually no increase in the NO_3–N and NO_2–N levels. The authors indicated that without recirculation, the contact time between the wastewater and the biofilms inside the reed bed matrices may not be adequate for the nitrifying bacteria to function. Zhao et al. (2004) employed 1:1 recirculation ratio in four-stage tidal flow VSSF wetlands for treating heavily loaded (1,055 g COD m^2 $days^{-1}$) pig slurry wastewater, and revealed excellent removal efficiencies of 77, 78, 66, and 62% for COD, BOD_5, TSS and NH_4^+–N, respectively. The authors attributed the high removal efficiencies even at high HRT to the effluent recirculation that enhanced oxygen transport due to the re-distributing of the wastewater.

In-series design

It has been acknowledged that pollutant removal is efficient by multi-stage CWs than single-stage systems (Vymazal, 2005, 2011), and hybrid systems have gained increased attention in treating agricultural runoff (Sharma et al., 2013; Zhang et al., 2016). In the HSSF–VSSF systems, nitrification takes place in the vertical flow stage at the end of the process sequence. If nitrate removal is needed, it is then necessary to recirculate the effluent back to the front end of the system where denitrification can take place in the anoxic HSSF bed (Vymazal, 2011). In order to achieve higher TN removal or to treat more complex agricultural wastewaters, hybrid systems can also include a FWS stage (Lim et al., 2001; Lin et al., 2002).

Lee et al. (2010) evaluated the efficiency of hybrid CWs for treating dairy wastewater in Vermont, USA, and reported that both VSSF–HSSF and HSSF–HSSF achieved high treatment efficiencies for BOD_5 (89 and 86%) and TSS (94 and 95%). HSSF–HSSF exhibited higher removal efficiencies for dissolved reactive phosphorus (DRP) (75%) and NH_4^+–N (64%), compared to the VSSF–HSSF systems (68% for DRP and 64% for NH_4^+–N). Lin et al. (2005) evaluated the performance of a wetland treatment unit, mainly consisting of FWS and HSSF wetland cells for controlling water quality of an aquaculture system for intensive shrimp culture. The authors indicated that FWS–HSSF wetland cells effectively removed TSS (55–66%), BOD_5 (37–54%), total ammonia (64–66%), and NO_3^-–N (83–94%) even under high HLR of 1.57–1.95 m $days^{-1}$. Comino et al. (2011) explored the capacity of a VSSF–HSSF wetland system in a cold climate region in Italy for treating mountain cheese factory wastewater. Overall removal rates were reported as 28–88% for TSS, 53–80% for COD, 31–80% for BOD_5, 25–80% for TOC, 10–73% for TP, and 40–51% for TN, during the monitored periods. The authors stated that although the outlet concentrations never satisfied the national limits for discharge in superficial water for industrial facilities, this result did not indicate a failure of this hybrid wetland system as the removal rates were still high.

Microbial community dynamics

Efficient wastewater treatment in CWs depends upon not only the wetland design and system capacity, but also microbial community dynamics, and the interactions between biogenic compounds and particular contaminants in filter beds (Stottmeister et al., 2003). The activities of microbial communities involved in biogeochemical cycles of wetland soils are crucial for the functions of wetlands, because they play a significant role in energy flows and nutrient transformations (Scholz & Lee, 2005). Moreover, pollutant removal and microbial activities are closely associated to the cycling of pollutants, and some microorganisms may play an important role in nitrogen transformations in CWs, such as Achaea nitrifies, denitrifying fungi, aerobic denitrifying bacterial, and heterotrophic nitrifying bacteria (Truu et al., 2009). Additionally, operational conditions (e.g., operational mode and hydraulic loading) and wetland configurations (e.g., wetland type, plant species) that influence the observed variation in effluent quality may lead to shifts in the structure and diversity of the microbial community (Faulwetter et al., 2013). A better understanding of the microbial activities and functions involved in biogeochemical processes, their distribution within CWs and the microbial structure shifts in relation to environmental variations are important for wetland application.

Ibekwe et al. (2003) characterized microbial composition in two HSSF CWs receiving dairy washwater using denaturing gradient gel electrophoresis (DGGE) technology, and revealed that the most predominant bacterial abundance was affiliated to *Bacillus*, followed by *Clostridium*, *Mycoplasma*, *Eubacterium*, and *Proteobacteria*. The abundance of ammonia-oxidizing bacteria *Nitrosospira* was found to be higher in wetland effluent samples, while a higher percentage of *Nitrosomonas* sequence was found in raw washwater and a facultative pond. Subsequently, Ibekwe et al. (2016) assessed bacterial composition within a SFW

CW receiving swine wastewater using high-throughput pyrosequencing technology, and revealed that different bacterial groups were responsible for the composition of different wetland nutrients and decomposition processes. The results of principal coordinate analysis (PCoA) showed that about 54% of the variations in the wetland microbial community structures were explained by NH_4^+–N and PO_4^{3-}–P, implying that these two nutrient sources were strongly correlated with the distribution of bacteria species and contributed the most to microbial community dynamics in this wetland. The authors also indicated that the techniques could detect greater percent operational taxonomic units (OTUs) from *Nitrosospira*, *Nitrospira,* *Nitrosomonas*, *Nitrosovibrio*, and *Nitrosococcus*, compared to their previous study (Ibekwe et al., 2003) using denaturing gradient gel electrophoresis (DGGE) that could only detect *Nitrosospira* and *Nitrosomonas* using the same wetland. This study also demonstrated that high levels of nutrient status in different sections of CWs were correlated with the diversity and structure of bacterial communities.

Conclusions and perspectives

Constructed wetlands appear as a technically and environmentally sustainable alternative to conventional wastewater treatment. This review integrates knowledge of treatment performance and design protocols, which may be adopted in wetland systems receiving agricultural runoff or agro-industrial wastewater. Some conclusions and perspectives are also included here:

(1) The evaluation of the treatment performance in wetlands constructed for treating agricultural runoff and agro-industrial wastewaters indicates that hybrid systems, which combine different types of wetlands, can achieve the highest removal efficiency for TSS (91.2%), BOD_5 (82.7%), NH_4–N (77.6%), TN (73.3%), and TP (69.9%), while VSSF and HSSF CWs perform best for COD (78.7% for VSSF CWs) and NO_3^-–N (85.7% for HSSF CWs) removal among all types of wetlands. SSF CWs exhibit higher TN and TP removal efficiencies than those in FWS CWs.

(2) Nitrogen loading rates are expected to be high in agricultural treatment wetlands, compared to wetlands constructed for municipal wastewater treatment. The classical nitrification–denitrification pathway is still considered to be the predominant mechanism for nitrogen removal. Plant uptake may not be the main pathway for nitrogen transformation, but certainly the presence of macrophytes is fundamental for establishing a heterogeneous environment that facilitates the physical, biochemical and photochemical processes for contaminant removal. Optimizing the operational conditions, such as sequential aerobic-anaerobic conditions and availability of organic carbon, demands further in-depth research. The novel Anammox process that does not require organic carbon is an attractive option and can offer significant potential for nitrogen removal improvement in agricultural treatment wetlands. However, this novel biodegradation route for nitrogen removal has not been consistently implemented, and more research is needed to explore this process in agricultural CWs. In particular, identification of growth conditions for Anammox bacteria (e.g., temperature, pH) is crucial for the determination of design and operational parameters. In addition, to optimize nitrogen removal, mass balance analysis for components of nitrogen transformation occurring within treatment wetlands can provide a better understanding for nitrogen transformations.

(3) Assimilation and sorption of phosphorus by soil is considered as long-term phosphorus retention in agricultural wetlands. Phosphorus removal from wetlands exemplifies the combination of physical, chemical, and biological mechanisms, and is significantly related to the contents of Al, Fe, Ca, and Mg in the soil. Although macrophytes also play a role in phosphorus assimilation and storage, phosphorus uptake and storage by macrophytes provides a short-term removal and phosphorus stored in the plants is eventually released back into the water upon dieback of algae and microbes. Most previous studies evaluated only short-term phosphorus assimilation/sorption capacity and batch soil microcosms such as kinetic studies using sediment–water columns. Further investigation of

phosphorus removal may include long-term monitoring and assessment of storage/sorption capacity, development of scale-up technology and full-scale testing based on the lab-scale experimental results.

(4) Properly designed and operated agricultural treatment wetlands can effectively reduce or eliminate contaminant loads to downstream waters. A definitively one-for-all design does not exist. The treatment performance of wetland systems varies with climate conditions and operational parameters. Optimization of design criteria of wetland systems, such as pretreatment facilities, effluent recirculation, forced aeration, and in-series design, can substantially enhance contaminant removal (particularly with high inflow loads), allowing for a more holistic approach to agricultural wastewater management. It is also worth noting that there is a crucial inter-dependency between the modification of operation conditions and many parameters (e.g., inflow loading rates, wastewater chemistry, wetland configurations, plant species, etc.).

(5) A further amelioration and improvement is feasible by focussing on hydrology of CWs. Agricultural treatment wetlands appear to be limited by the organic and nitrogen loads. Moreover, the response of agricultural treatment wetland systems to precipitation events also has impact on hydrological fluctuation. An in-depth analysis of the wetland hydraulics would be useful in understanding the effect of retention times and flow paths on treatment performance, and would facilitate evaluation of their hydraulic treatment efficiencies. Studies on the correlation between wetland loading and treatment efficiency would be conducive to determine the threshold of organic/nitrogen loading to achieve a more effective treatment performance.

(6) There is dearth of knowledge on the microbial structure, diversity, and function in treatment wetlands receiving agricultural runoff. Recent high-throughput pyrosequencing technologies (e.g., 454 pyrosequencing, Illumina sequencing) have demonstrated an excellent capacity for providing profound insights into an overall microbial community and making an accurate phylogenetic affiliation assessment for microbes in complex environmental systems. This application can significantly enhance the understanding of the dynamics of microbial community structure in wetlands. Increasing knowledge on the relationship between the treatment performance and the role of functional bacteria in CWs, the bacterial community shifts in relation to environmental (e.g., soil and water properties, plant species) and operational (e.g., hydraulic loading, effluent circulation) factors, and the influence of wetland variables on the composition and structure of microbial communities, should be the main objectives in future research.

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