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REVIEW ARTICLE

Constructed wetlands as sustainable ecotechnologies in decentralization practices: a review

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Abstract Recently, a range of novel and cost-effective engineered wetland technologies for decentralization practices of domestic wastewater treatment have been developed with ecological process modification, the use of functionalized plants, and advanced biofilm formation. However, selecting the one that can be more appreciated for on-site sanitation is still uncertain. This paper reviews the role of plants, media materials, microorganisms, and oxygen transfer in domestic wastewater purification through constructed wetlands (CWs). The effectiveness of traditional and recently developed CWs and the necessity of an induced biofilm attachment surface (BAS) in these systems for the treatment of domestic sewage are presented. This review also elucidates the idea of CWs for domestic wastewater characteristics highly stressed by total dissolved solids and the adaptive strategies in mitigating the cold climate impacts on their efficiencies. Further research needed to enhance the stability and sustainability of CWs is highlighted. By a more advanced investigation, BAS CWs can be specified as an ideal treatment process in decentralization.

Keywords Decentralization · Constructed wetlands · Wastewater treatment · High TDS · Cold climate

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Introduction

In recent years, constructed wetlands (CWs) have attracted considerable attention in urban wastewater treatment, particularly in decentralized sanitation, because they are affordable, reliable, simple in design and operation, and offer an environmental-friendly approach (Wu et al. 2011a). CWs are believed to have started in Germany based on research by Kathe Seidel in the 1960s and by Reinhold Kickuth in the 1970s (Kadlec and Wallace 2009). These ecological engineered systems are effective in removing many pollutants, such as organic compounds, suspended solids, pathogens, nutrients, and emergent pollutants. They are designed to take advantage of the same processes occurring in natural wetlands but within a more controlled environment. Gaining a better understanding of the mechanisms associated with CWs has led to a wide variety of designs and configurations to achieve more efficient domestic sewage treatment, e.g., single-stage modification (Chale 2012; Kumari and Tripathi 2014), multi-staged in series (Melian et al. 2010), and/or combination with other treatment technologies (Singh et al. 2009). Considerable research is being conducted on the use of CWs treating domestic wastewater under the specific influent conditions seriously stressed by total dissolved solids (TDSs) which could be faced in areas, such as coastal regions, where seawater applies to indoor activities (Valipour et al. 2014b). Many studies have also assessed the adaptation of CWs to cold climates through a sound operational approach (Ouellet-Plamondon et al. 2006). Accordingly, several authors have published review papers related to the use of CWs in the wastewater treatment (Vymazal 2002, 2005, 2013; Babatunde et al. 2008; Haynes 2015; Liua et al. 2015; Vymazal and Březinová 2015; Wu et al. 2015). Nevertheless, there are comparatively few reviews of the current knowledge aimed at the on-site sanitation of domestic wastewater. Still, there is a hesitation about the selection of the appropriate type of CW which is more suitable for domestic wastewater treatment in decentralized sanitation concepts. Therefore, it is imperative to briefly look into the applications of CWs for domestic wastewater treatment by paying more attention to decentralization practices. In addition, a valuable overview regarding the potential ability of wetland systems in dealing with TDS-stressed wastewater and operational strategies taken in a cold climate is also an important issue that would be a ready tool in the implementation of CWs for small-sized communities and isolated areas; this is rare in published literature reviews.

This paper provides a review of ecological CW technologies, including the essential components, the mechanisms of pollutant removal, and the process models (traditional and advanced) for treating domestic wastewater. The performance of various types of wetland processes on contaminant removal (focusing on organic matters, suspended solids, and nutrients) are also summarized. Through this study, the scope and issues at hand may be better defined to deal with environmental stressed conditions. This paper further highlights the potential areas worthy of the future sustainable application of CWs. The conceptual framework of this paper is to eventually create a context in which a new ideal could be inspired for decentralization practices.

Ecology in CWs

The main compounds employed in CWs are marshy vegetation, microbial communities, and media material (soil strata or any other material used as the matrix within the CWs). These systems utilize a combination of physical, chemical, and biological processes to remove contaminants from wastewater.

Marshy vegetation

Marshy plants play a crucial role in creating a pleasing landscape which can be incorporated into residential developments. They provide a valuable ecological habitat for wildlife. In considering the application of wetlands to treat wastewater, plants have several properties in relation to the treatment process that make them an essential biotic component in CWs. Depending on the plant species and dense coverage, plant effects have a strong influence on the treatment performance, based principally on the microbial communities, activities, and their population by providing ideal attachment sites (through roots, stems, and leaves), uptake capability, releasing oxygen, and filtration (Valipour et al. 2014b). However, beside the multi-role of wetland vegetation, contaminant uptake by plants has a minor role. Plant species used for phytoremediation should be possibly native and have a high growth rate, high biomass, adapt ecologically to diverse habitats, and the ability to accumulate the target pollutants in the aboveground parts. Four types of aquatic macrophytes, including free-floating, floating-leaved, submerged, and emergent, are typically used in CWs. Table 1 lists the features of some plant species commonly used in CWs.

Plants help filter suspended solids out of the effluent flowing through wetlands, whereas the retention time plays a significant role on the solid removal efficiencies. Nutrients (N and P) and other impurities are taken mainly up by wetland plants through the epidermis and vascular bundles of the roots and further transported upward to the plant (Valipour et al. 2014a, b). On the other hand, a small amount of N (<10 %) and P (<5 %) has been reported to be removed by macrophyte harvesting compared to their total removal in vegetated beds (Mander et al. 2003). The plant uptake efficiency differs in relation to the system configuration, loading range, pollutant concentration in wastewater, and environmental conditions. The rate of plant uptake is also limited by their net productivity (growth rate) and tissue nutrient concentrations (Table 1). Evapotranspiration plays an additional important role by increasing the hydraulic retention time significantly in wetland systems. The transpiration mechanism (reliant on species and environmental conditions) is positively related to impurity absorption, volatile compound emissions into the atmosphere, and the water purification capability index of plants (Valipour et al. 2015). Furthermore, age greatly influences the physiological activity of the plants, particularly its roots. The roots of younger plants can have greater ability to absorb impurities and release oxygen than older plants due to the increased lignification and suberization processes occurring with increasing age of the plants and tissues (Heers 2006).

Microorganisms/biofilms

Microorganisms, such as bacteria, fungi, and algae, play a key role in the transformation and mineralization of nutrients and organic compounds in CWs. The presence of bacteria, either in the form of a suspension or attached biofilm, is particularly more important because they have versatile metabolic pathways, high metabolic rates, and very short generation times. The rhizosphere has been reported to be associated predominantly with gram-negative bacteria (Valipour et al. 2009, 2011b, 2014b). This can be related to their ability to utilize efficiently the growth substrates available in the rhizosphere and to cope with polluted environments because of the presence of detoxifying enzymes (Valipour et al. 2014b). Rhizospheric bacteria can improve plant nutrition and growth, protect plants against diseases, and responses to external stress factors. Many rhizobacteria are capable of lowering the increased plant endogenous ethylene levels (which inhibit plant growth) to reestablish a healthy root system that needs to be faced with environmental stress (Martínez-Viveros et al. 2010; Gontia-Mishra et al. 2014). In addition, arbuscular mycorrhizal of the adapted fungi can sequester toxic substances and

Table 1 Properties of some aquatic plants used in CWs

Macrophyte	Type	Preferred temperature (°C)	Optimal pH	Root penetration (cm)	Nutrient uptake (kg/ha/year)	
					Nitrogen	Phosphorous
Bulrush (*Scirpus* sp.)	Emergent	16–27	4–9	75	125–1825	18–438
Cattail (*Typha* sp.)	Emergent	10–30	4–10	75	600–2630	75–403
Papyrus sedge (*Cyperus papyrus*)	Emergent	10–30	6–8.5	60	1100–3650	50–1059
Reed (*Phragmites* sp.)	Emergent	12–33	2–8	60	225–2500	35–120
Rush (*Juncus* sp.)	Emergent	16–26	5–7.5	25	800	110
Duckweed (*Lemna* sp.)	Free-floating	6–33	6.5–7.5	2	350–1200	116–400
Water fern (*Salvinia rotundifolia*)	Free-floating	10–30	6–7.7	–	350–1700	92–450
Water hyacinth (*Eichornia crassipes*)	Free-floating	12–35	65.–7.5	100	1950–5850	350–1125
Water lettuce (*Pistia stratiotes*)	Free-floating	15–35	6–6.8	80	900–3248	40
Water pennywort (*Hydrocotyle* sp.)	Free-floating	15–35	6.5–7.5	–	2957	730
Coontail (*Ceratophylum demersum*)	Submerged	15–35	6–10	–	100	10
Sago pondweed (*Potamogeton pectinatus*)	Submerged	10–37	6–10	–	500	40

() scientific name

promote plant growth by increasing nutrient absorption by the roots to enhance the host plant's resistance against environmental stresses (Kapoor et al. 2013).

The biodegradation of organic matter is governed by either facultative or obligate aerobic/anaerobic bacteria in a wetland system, whereas aerobic respiration is more efficient and practical for the elimination of organic pollutants (Valipour et al. 2009, 2011b). Nutrients can be assimilated into microbial cellular biomass at the same level as plants (<10 % N and <5 % P) (Mander et al. 2003), but no net nutrient removal will result once the system reaches a steady state of biomass. This transformation is accepted to play a minor role in the removal of nutrients from a soluble phase, particularly when the influent wastewater contains a high nutrient concentration. The major biological processes responsible for the elimination of nitrogen in wetland systems include ammonification, nitrification, and denitrification. On the other hand, its removal efficiency can be attributed to dissolved oxygen available for nitrification, denitrification, the strength of organic carbon, and the nitrogen level in the influent sewage. Unlike nitrogen, phosphorus does not have an atmospheric component. Therefore, its removal is notoriously poor in any CW and can only occur efficiently by incorporating special matrix material with high sorption capacity (Ciria et al. 2005).

Media

Media materials (soil, sand, and gravel) strongly affect the movement of water through the bed (hydraulic conductivity) and macrophyte growth. These materials provide a huge surface area for microorganisms to attach additionally to plant biomass (roots, stems, and leaves) and also act either as filtration and/or adsorption medium for pollutants (Taleno 2012). Both the chemical soil composition and physical parameters, such as grain size distributions, interstitial pore spaces, effective grain sizes, degrees of irregularity, and the coefficient of permeability, are the key criteria influencing the treatment performance.

Ionized ammonia can be removed from wastewater through exchange with soil strata, detritus, humic substances, and organic and inorganic sediments or else fixed within the clay lattice in wetland systems. On the other hand, adsorbed ammonium binds loosely to the materials and can be released easily in response to changes in water chemistry (Kadlec 2009). Phosphorus is removed primarily through bed material rich cations (typically divalent or trivalent) by adsorption and ion exchange reactions. As these associated mechanisms have a finite capacity, the elimination of phosphorus will cease when that capacity is reached (Brix et al. 2001). Soil materials used in CWs normally do not have large quantities of ion cations, achieving generally very low phosphorus removal.

Alternative used media materials can be classified into natural materials (e.g., apatite, bauxite, dolomite, zeolite, laterite, limestone, opoka, and shale), industrial by-products (e.g., bauxsol, burnt oil shale, coal fly ash, ochre, red mud, and slag), and man-made products (e.g., alunite, filter P, filtralite, lightweight aggregates, norlite, oyster shell, and polonite) (Vohla et al. 2011). Most researchers employ natural zeolite exchangers (volcanic tuffs, usually clinoptilolite-rich tuff) as low-cost, effective, readily available materials to treat ammonium-containing wastewater (Copcia et al. 2010). Among various industrial by-products, the highest phosphorus removal capacities stated for some furnace slags up to 420 g P kg^{-1}. The natural and man-made media materials have been reported for maximum removal capacities of about 40 and 12 g P kg^{-1}, respectively (Vohla et al. 2011).

Oxygen transfer/diffusion

The amount of oxygen released from plant roots varies strongly according to the species-specific differences, seasonal variations, and different wetland techniques used in wastewater treatment. Aquatic plants transfer atmospheric oxygen to their root system through an internal gas space, called aerenchyma, and release a fraction of this oxygen (30–40 %) into the rhizosphere for aerobic microbial activity (Wiessner et al. 2006; Li et al. 2011a). Obvious processes associated with aboveground or above water gas exchange and gas transport inside the plants (through aerenchyma) are driven by photosynthesis, diffusion, thermoosmosis, convective flow induced by pressure gradient, and humidity pressurization (Allen 1997; Wiessner et al. 2006). In addition, dead and broken shoots in some plants, such as *Phragmites* sp., which has the ability to grow in variable water depths, can also allow for the transport of some oxygen to the root zone. This can be attributed to the venturi effects of wind flow, stagnation pressures of wind, pressure oscillations induced by turbulent wind flow, and the effects of heat tubes, e.g., evaporation and condensation at opposite ends of the tubes (Allen 1997); perhaps, these phenomena can provide a reliable prediction in wetlands operating under cold climates.

Traditional wetland systems

The wetland systems based traditionally on a water flow regime can be classified into surface flow (SF) and subsurface flow (SSF) CWs. The SF CWs often utilize free-floating, floating-leaved, emergent, and submerged macrophytes, whereas the SSF CWs are limited to emergent macrophytes.

SF CWs

In SF CWs, water is flowed horizontally aboveground and exposed to the atmosphere. SF CWs with free-floating macrophytes (50–100 cm depth) function as a horizontal trickling filter, where the submerged roots mainly provide physical support for a thick bacterial biofilm (Valipour et al. 2015). Moreover, the soil or any other suitable medium (at least 20–30 cm) can serve to support rooted vegetation (if they are dominant plantation) in the SF CWs (floodwater depth≤ 50 cm). The microbial activities (aerobic/anaerobic) primarily take place in the superior layer of the soil, on the surface of the immersed stems, and leaves of the plants.

The advantages of SF CWs include the following: relative ease to construct, simple designs, low maintenance, inexpensive operation, and habitat value. They can also be used for wastewater with higher levels of suspended solids. On the other hand, these constructed wetlands have lower removal rates of contaminants per unit volume. Therefore, they require more land space and are expensive to construct in terms of capital cost. Odors and insects are problems because of the free water surface and low treatment efficiency. These complications have limited the use of SF CWs for the treatment of wastewater, particularly in decentralization practices.

SSF CWs

In most of these systems, the flow path is horizontal, although some use vertical flow paths. In horizontal subsurface flow (HSF) CWs, the wastewater essentially flows horizontally (5 to 15 cm below the media surface) through the support media (50–100-cm active zone) (Tee et al. 2012). In vertical subsurface flow (VSF) CWs, the wastewater is typically discharged onto the entire surface via a distribution system and passes vertically into the media (having 50–100-cm active zone). VSF CWs, depending on whether the wastewater is fed onto the surface or to the bottom of the wetland, include up-flow (VUF CWs) and down-flow (VDF CWs). The latter is used more generally in wastewater treatment compared to the former system. This can be explain by the fact that they provide good oxygen transfer and are more suited for aerobic conditions resulting in the better elimination of pollutants (Zhao et al. 2011).

SSF CWs are believed to have several advantages over SF CWs. The soil matrix provides a large surface area available for attached microbial biofilms. As a result, the treatment responses are faster and less space is needed compared to SF CWs. The subsurface position of the water and the accumulated plant debris on the surface of the bed can also offer a greater thermal protection for cold climates in SSF CWs. Nevertheless, these systems can still be restricted by choking, clogging, odors and vectors, slow mass transfer, poor root penetration into the multi-layer soil column, high-area requirements, and capital investment. Therefore, applying these systems in practice may also not prove to be practicable in decentralization.

Process modification

A wide range of expended designs, configurations, and combinations with other technologies have been trialed in an attempt to prevail over the limitations of CWs and establish them as an effective tool for purifying municipal wastewater effluent.

Single-wetland systems

Single constructed wetland designs include systems with a shallow pond (Fig. 1a) (Valipour et al. 2011b), a tidal flow (Hu et al. 2014), a baffled flow (Fig. 1b) (Tee et al. 2012, 2015), a step-feed (Fig. 1c) (Stefanakis et al. 2011), artificial aeration (Fig. 1d) (Stefanakis et al. 2014), multi-level (two-layer) drop aeration (Fig. 1e) (Zou et al. 2012), and biofilm attachment surface, namely, as bio-rack (Fig. 1f) and bio-hedge (Fig. 1g) techniques (termed BAS process) (Valipour et al. 2009, 2015).

Shallow pond is the system having a dense floating mat of vegetation (water hyacinth) and the water depth based on the fully matured plant root submerged to avoid the anaerobic zone. The tidal flow is based on the batch principal by multiple periodical flood and drain cycles per day. The treatment performance of this system could be attributed to many factors, including flood-to-drain time ratio, oxygen transfer efficiency, and media material characteristics. The baffled flow design, by inserting vertical baffles along the width of the wetland, is able to nurture sequential aerobic, anoxic, and anaerobic conditions within the same wetland bed. The main purpose of the step-feeding lies in the effective utilization of wetland area through uniform loading distribution (i.e., biochemical oxygen demand (BOD) and total suspended solids (TSS)) by discharging the wastewater at multiple input points along the length of the cell. In this strategy, the design/operation parameter might be great importance to avoid rapid clogging of subsurface flow systems and increase the life span of material used in the bed with step-feeding. Artificial aeration in CWs (particularly by intermittently aerated mode) can facilitate aerobic biodegradation of organic materials, nitrification, and consequently denitrification. The continuous artificial aeration results in the faster depletion of the influent carbon source and the lack of effective anoxic zone which both of them inhibit the subsequent denitrification step in CWs, and also, the high operation cost remains doubtful. In multi-level drop aeration design, the wastewater flowed to the top of the two-layer drop aeration units that are installed at above CW, and then being dropped into the next one by gravity. The amount of oxygen transferred into wastewater strongly depends on the designed flow rate and drop height of the aeration devices.

Foremost among these single-unit CWs, the BAS processes (especially bio-rack wetlands) can be strongly recommended for decentralization practices. In particular, BAS CWs are incorporated with the advantages of phytoremediation and engineered attached microbial growth processes. These systems (with effective depths of 0.15 and 0.5 m in bio-rack and bio-hedge, respectively) are free soil strata, and in lieu, a support matrix (BAS) is provided to enrich the microflora. They eliminate all the disadvantages of the traditional-type CWs, i.e., choking and clogging, odor- and insect-related problems, and slow biodegradation rate. These techniques have led to a low-space requirement; a low-capital investment; and most importantly, a high degradation of organic pollutants. In other modified single-wetland units, however, their performance over traditional wetland systems can provide quantitative advantages given the limited odors and insect vectors with smaller footprints and lower-capital costs (more or less depending on their configurations), but these issues can still be faced during their applications.

CWs combined with other technologies

CWs are being used as a post-treatment unit (typically for anaerobically treated wastewater) to avoid the clogging problems of porous media (particularly where soil strata are used), lower the organic load rate, and reduce the land area requirement and capital investment (Jamshidi et al. 2014). Yet, the behavior of the constraints is the same as in the other types of CWs, which are characterized by SSF (i.e., the bed with soil layers) and SF operation, as a post-treatment unit. For that reason, these integrated treatment processes may be associated with complications in decentralization practices. In contrast, the incorporation of the BAS wetland systems can be recommended.

Hybrid systems

Various types of CWs may be combined in order to complement each other, and are ideal for achieving higher treatment efficiency, particularly for nitrogen removal (termed hybrid system) (Tuncsiper 2009; Tee et al. 2012; Ávila et al. 2014). This design generally consisted of two stages of several parallel CWs in series, such as VDF-HSF, HSF-VDF, VDF-VUF, HSF-SF, and SF-HSF CWs, while the hybrids from VDF and HSF CWs arranged in a staged manner are most frequently used to achieve, in addition to BOD and SS removal, nitrification and denitrification. Besides, the multi-stage CWs that were comprised of more than two stages were also used (Abidi et al. 2009; Vymazal and Kropfelova 2011; Zhao et al. 2011; Ávila et al. 2014). Despite this, they often faced difficulties transitioning from traditional wetland treatment methods and a recycling system to enable the wastewater to undergo treatment under oxidation and reduction conditions repeatedly. Therefore, hybrid systems may also be restricted as a model for decentralization unless they engineered predominantly by BAS.

Process performance

A review of the literature has shown that the most treatment wetland studies deal primarily with organic, suspended solid, and nutrient pollutant removal. The performance of CWs depends on many factors, including the environmental conditions, degree of vegetative completeness within a wetland unit, types of plant, operational strategy taken, bacterial population, and oxygen concentration. In that order, emergent *Phragmites* sp. and free-floating water hyacinth have been found to have a great potential for use in phytoremediation

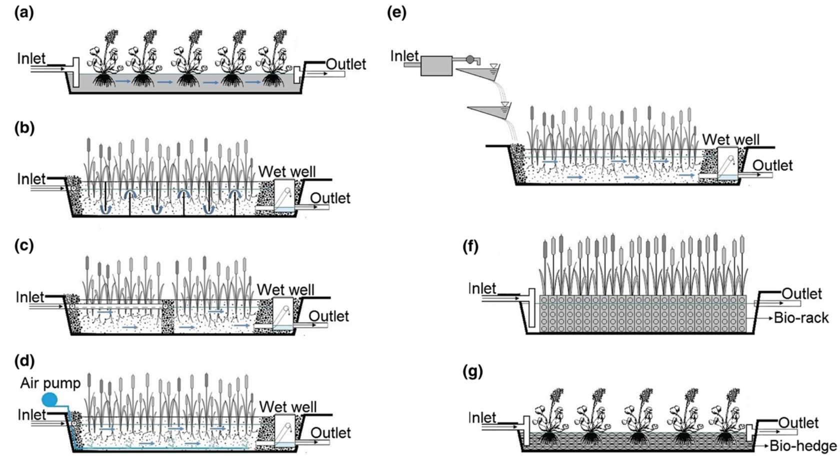

Fig. 1 Examples of advanced wetland systems: **a** shallow pond, **b** baffled follow, **c** step-feed, **d** artificial aeration, **e** multi-level drop aeration, **f** bio-rack, and **g** bio-hedge

of wastewater among other species. The potential abilities of these plants can be explained by their higher growth rate and extensive root systems responsible for greater microbial mass contribution, the desirable rate of the oxygen transfer efficiency, nutrient cycling, filter bed stabilization, and water quality improvement throughout the treatment unit. A mixed culture of plant species might not play an important role during wastewater treatment compared to monoculture systems. Agreeing the hydraulic retention time (HRT) is a noteworthy direct predictor in a performance evaluation of CWs. An excessively short HRT will result in low treatment efficiency, whereas an excessively long HRT will not be economically feasible and might cause clogging of the filter material.

Organic and suspended solid removal

Regardless of the traditional SF, which is operated in batch mode (see Table 2), high levels of organic matter (≥50 % chemical oxygen demand (COD) and ≥70 % BOD_5) are found in all types of CWs (Tables 2–5); despite this, the systems differ in terms of the operating conditions applied. Similarly, the removal of suspended solids is also high in all types of CWs (≥67 % TSS).

As shown in Table 2, high organic removal efficiencies can be adopted for traditional HSF CWs using *Phragmites* sp. at a minimum HRT of 1 day. According to Table 4, the traditional VDF CW shows better organic removal treatment performance from pre-treated domestic wastewater than traditional HSF CWs because they provide an ideal environment for aerobic bacterial respiration. Pre-treatment is a method used to remove anything which could hamper subsequent CWs. Generally, anaerobic processes are ideal for pre-treatment practices, while they are sometimes applied as stand-alone solution in decentralization. Anaerobically treated effluent is totally nutrient rich and containing high concentration of unacceptable pathogen, therefore needs a post-treatment step (as CWs). Yet, there is concern regarding the feasibility of traditional CWs as a cost-effective method because they typically require a long HRT to achieve efficient pollutant removal. There are restrictions involved with these traditional CWs as explained elsewhere.

The BAS wetland systems have proven effective in challenging applications of CWs in wastewater treatment. Among the cited modified processes applied in treating domestic wastewater (Table 3), the bio-rack system planted with *Phragmites* sp. offers an excellent opportunity for the high efficiency of organic removal at a shorter HRT (0.42 day). As an obvious effect of vegetation, the bio-rack system in the presence of *Typha* sp. can provide effective organic removal efficiency comparable to that planted with *Phragmites* sp. at a longer HRT of 0.71-day HRT; but, it can still operate at higher loading rates than other cited modified CWs. In fact, extensive root system of *Phragmites* sp. occupies thoroughly over the depth of the treatment unit and correspondingly enhances aerobic microbial mass contribution as well as organic and inorganic removal efficiencies. Similarly, the bio-hedge system (at low HRT of 0.6 day) appears to be an effective process in organic pollutant removal compared to other modified wetlands (Table 3). On the other hand, as shown in the Table 4, the bio-rack system planted with *Phragmites* sp. also appears to be a suitable alternative that can be used to remove organic pollutants from pre-treated domestic wastewater (at a HRT as low as 0.4 day). The bio-rack system planted with *Typha* sp. was correspondingly found to have a slightly longer retention time (0.52 day) to achieve an almost comparable organic effluent quality. The HRT of 0.52 day is still shorter than the other CWs (listed in Table 4). The influent wastewater discharged into the bio-rack wetland (with *Phragmites* sp.) contains somewhat higher concentrations of organic pollutants than that planted with *Typha* sp. The higher influent pollutant concentrations may cause a longer retention time in the CWs. But, by contrast, there would still be the added benefit of dealing with constraints associated either with traditional or other modified CWs when the bio-rack system (planted with *Typha* sp.) is used. Hereupon, it can be verified that the type of wetland design, efficient oxygen diffusion, and the high

Table 2 Examples of design/operational parameters and treatment efficiency of traditional CWs

CW type	WT	Wastewater characteristics							HRT (day)	Flow rate (m^3/day)	HLR (m^3/m^2/day)	Vegetation	Reduction (%)					Study period (Months)	Country	Reference
		Temp. (°C)	pH	COD (mg/L)	BOD_5 (mg/L)	TSS (mg/L)	NH_4-N (mg/L)	TP (mg/L)					COD	BOD_5	TSS	NH_4-N	TP			
SF[a]	R	26	7.7	309	155	–	–	–	1.5	–	–	*Eichhornia* sp.	48	50	–	–	–	3	India	(Kumari and Tripathi 2014)
SF[a]	R	26	7.7	309	155	–	–	–	1.5	–	–	*Salvinia* sp.	37	38	–	–	–		India	(Kumari and Tripathi 2014)
SF[a]	R	26	7.7	309	155	–	–	–	1.5	–	–	*Eichhornia* sp. and *Salvinia* sp.	48	51	–	–	–		India	(Kumari and Tripathi 2014)
FSF	R	24–27	7.4	363	212	144	30	–	1.8	0.03	0.27	*Eichhornia* sp.	76	87	67	69	–	22	India	(Valipour et al. 2014a)
HSF	R	28–30	7.2	445	235	152	33	–	1	0.026	0.23	*Phragmites* sp.	80	90	69	73	–		India	(Valipour et al. 2014a)
HSF	R	19–42[b]	6.5	477	167	159	–	20	2	0.0125	0.014	*Phragmites* sp.	69	95	90	–	96	6	India	(Baskar et al. 2009)
HSF	R	–	7.69	–	105	33	16.6	5.03	1.5	0.198	0.34	*Typha* sp.	–	80	87	59	53	3	Thailand	(Sirianuntapiboon and Jitvimolnimit 2007)
HSF	R	–	7.69	–	105	33	16.6	5.03	1.5	0.198	0.34	*Canna* sp.	–	77	84	56	58		Thailand	(Sirianuntapiboon and Jitvimolnimit 2007)
HSF	R	–	7.69	–	105	33	16.6	5.03	1.5	0.198	0.34	*Typha* sp. and *Canna* sp.	–	79	86	63	64		Thailand	(Sirianuntapiboon and Jitvimolnimit 2007)
HSF	R	–	7.69	–	105	33	16.6	5.03	3	0.099	0.17	*Typha* sp.	–	86	90	69	77		Thailand	(Sirianuntapiboon and Jitvimolnimit 2007)
HSF	R	–	7.69	–	105	33	16.6	5.03	3	0.099	0.17	*Canna* sp.	–	84	91	68	79		Thailand	(Sirianuntapiboon and Jitvimolnimit 2007)
HSF	R	–	7.69	–	105	33	16.6	5.03	3	0.099	0.17	*Typha* sp. and *Canna* sp.	–	87	87	72	82		Thailand	(Sirianuntapiboon and Jitvimolnimit 2007)
HSF	R	–	7.69	–	105	33	16.6	5.03	6	0.0495	0.086	*Typha* sp.	–	90	92	86	86		Thailand	(Sirianuntapiboon and Jitvimolnimit 2007)
HSF	R	–	7.69	–	105	33	16.6	5.03	6	0.0495	0.086	*Canna* sp.	–	92	93	84	86		Thailand	(Sirianuntapiboon and Jitvimolnimit 2007)
HSF	R	–	7.69	–	105	33	16.6	5.03	6	0.0495	0.086	*Typha* sp. and *Canna* sp.	–	91	90	88	90		Thailand	(Sirianuntapiboon and Jitvimolnimit 2007)
HSF	R	5–22[b]	–	536	340	272	36	23	4.7	2	0.05	*Typha* sp.	79	97	92	20	36	24	Spain	(Ciria et al. 2005)
HSF	R	5–22[b]	–	744	485	359	43	29	4.7	2	0.05	*Typha* sp.	81	97	92	36	12		Spain	(Ciria et al. 2005)

SF surface flow; *HSF* horizontal subsurface flow; *WT* means wastewater type, including raw sewage (*R*)

[a] Operated under batch mode

[b] Air temperature

Table 3 Examples of design/operational parameters and treatment efficiency of modified CWs

CW type	WT	Wastewater characteristics							HRT (day)	Flow rate (m^3/day)	HLR (m^3/m^2/day)	Vegetation	Reduction (%)					Study period (Months)	Country	Reference
		Temp. (°C)	pH	COD (mg/L)	BOD_5 (mg/L)	TSS (mg/L)	NH_4-N (mg/L)	TP (mg/L)					COD	BOD_5	TSS	NH_4-N	TP			
Aerated SF[a, b]	R	26	7.7	309	155	–	–	–	1.5	–	–	*Eichhornia* sp.	82	83	–	–	–	3	India	Kumari and Tripathi 2014)
Aerated SF[a, b]	R	26	7.7	309	155	–	–	–	1.5	–	–	*Salvinia* sp.	77	77	–	–	–		India	Kumari and Tripathi 2014)
Aerated SF[a, b]	R	26	7.7	309	155	–	–	–	1.5	–	–	*Eichhornia* sp. and *Salvinia* sp.	83	85	–	–	–		India	Kumari and Tripathi 2014)
SPWHS	R	26–28	7.2	429	222	152	32		0.88	0.023	0.16	*Eichhornia* sp.	81	91	70	74	–	18	India	(Valipour et al. 2011b)
Bio-hedge	R	26–30	7.4	418	215	165	33	–	0.6	0.036	0.25	*Eichhornia* sp.	79	86	73	72	–	12	India	(Valipour et al. 2015)
Bio-rack	R	27–29[f]	7.3	421	218	154	33	–	0.71	0.071	0.72	*Typha* sp.	78	88	69	73	–	5	Iran	(Valipour et al. 2014b)
Bio-rack	R	25–26	7.3	340	207	171	31	–	0.42	0.127	1.14	*Phragmites* sp.	75	87	73	70	–	19	India	(Valipour et al. 2009)
Step-feed HSF[c]	S	16	6.8	491	373	–	31	10	14	–	–	*Phragmites* sp.	84	83	–	39	60	8	Greece	(Stefanakis et al. 2011)
Step-feed HSF[d]	S	19	6.8	443	379	–	23	14	6	–	–	*Phragmites* sp.	87	89	–	69	78		Greece	(Stefanakis et al. 2011)
Aerated VDF[a, b]	S	14–15[f]	–	352	–	–	46	–	3	–	–	*Phragmites* sp.	97	–	–	99	–	4	China	(Fan et al. 2013a)
Aerated VDF[a, e]	S	14–15[f]	–	352	–	–	46	–	3	–	–	*Phragmites* sp.	96	–	–	97	–		China	(Fan et al. 2013a)
Aerated VDF[a, e]	S	–	7.5	113	–	–	40	3.9	3	–	–	*Phragmites* sp.	91	–	–	99	65	5	China	(Fan et al. 2013b)
Aerated VDF[a, e]	S	–	7.4	217	–	–	40	3.9	3	–	–	*Phragmites* sp.	95	–	–	99	79		China	(Fan et al. 2013b)
Aerated VDF[a, e]	S	–	7.4	429	–	–	40	3.9	3	–	–	*Phragmites* sp.	96	–	–	99	92		China	(Fan et al. 2013b)
Aerated VDF[a, e]	S	–	7.4	836	–	–	40	3.9	3	–	–	*Phragmites* sp.	97	–	–	96	99		China	(Fan et al. 2013b)
TF	S	20–25[f]	7.7	590	167	297	42	9.6	(6.75:0.5)	0.003	0.44	*Phragmites* sp.	49	71	67	43	84	5	Ireland	(Hu et al. 2014)
TF	S	20–25[f]	7.7	436	181	107	46	5.1	(5.75:1.5)	0.003	0.44	*Phragmites* sp.	65	83	81	70	85		Ireland	(Hu et al. 2014)
TF	S	20–25[f]	7.5	552	244	168	51	10.8	(4.75:2.5)	0.003	0.44	*Phragmites* sp.	83	94	92	96	95		Ireland	(Hu et al. 2014)
TF	S	20–25[f]	7.7	207	61	69	55	8.2	(4.75:2.5)	0.003	0.44	*Phragmites* sp.	62	79	86	94	88		Ireland	(Hu et al. 2014)
TF	S	20–25[f]	7.6	224	105	65	52	8.9	(4.75:2.5)	0.003	0.44	*Phragmites* sp.	70	87	85	96	88		Ireland	(Hu et al. 2014)
TF	S	20–25[f]	7.6	464	182	157	50	8.6	(4.75:2.5)	0.003	0.44	*Phragmites* sp.	82	92	93	95	91		Ireland	(Hu et al. 2014)
TF	S	12–25[f]	6.4	–	193	–	37	–	(3:3)	0.022	0.9	–	–	84	–	82	–	–	China	(Wu et al. 2011b)
TF	S	12–25[f]	6.5	–	193	–	75	–	(3:3)	0.022	0.9	–	–	82	–	74	–		China	(Wu et al. 2011b)

Table 3 (continued)

CW type	WT	Wastewater characteristics							HRT (day)	Flow rate (m^3/day)	HLR (m^3/m^2/day)	Vegetation	Reduction (%)					Study period (Months)	Country	Reference
		Temp. (°C)	pH	COD (mg/L)	BOD_5 (mg/L)	TSS (mg/L)	NH_4-N (mg/L)	TP (mg/L)					COD	BOD_5	TSS	NH_4-N	TP			
TF	S	12–25[f]	6.6	–	366	–	75	–	(3:3)	0.022	0.9	–	–	86	–	67	–		China	(Wu et al. 2011b)
TF	S	12–25[f]	6.2	–	366	–	34	–	(3:3)	0.022	0.9	–	–	91	–	33	–		China	(Wu et al. 2011b)

SF surface flow; *SPWHS* shallow pond water hyacinth system; *HSF* horizontal subsurface flow; *VDF* vertical down-flow; *TF* tidal flow; *WT* means wastewater type, including raw sewage (*R*) and synthetic sewage (*S*)

[a] Operated under batch mode

[b] Operated under continuous aeration

[c] Inflow distributed at three points by percentage of 33:33:33

[d] Inflow distributed at three points by percentage of 60:25:15

[e] Operated under intermittent aeration

[f] Air temperature; ( ) flood-to-drain time ratio (h: h)

accumulation of the attached microbial biofilm are the major factors for high level of removal efficiency in the BAS practices. These exceptional processes identified to overcome the limitations of any other CWs and permit treatment of effluent in the most cost-effective method.

The shallow pond water hyacinth process is less efficient than BAS wetland systems (as bio-rack and bio-hedge techniques) by increasing 20–50 % HRT but more effective in removing organic pollutants than other modified processes (Table 3). The type of design being used in this system ensures the optimal interactions between the wastewater effluent and microbial biomass in the pond. It could offer the better oxygen diffusion efficiency through the roots and the accumulation of a larger aerobic bacterial population. Nevertheless, it may be thought to require a large land area and have a potential to become a mosquito problem.

The step-feed HSF CWs have been operated for long retention times (6 and 14 days) (Table 3). In full-scale CWs, the main purpose of the step-feeding strategy lies in the effective utilization of wetland area through uniform loading distribution. This strategy ensured uniform distribution of influent wastewater through the reactor bed, prevents overloading of the influent, and enhances the treatment performance (Stefanakis et al. 2011). While, this technique may still causes choking and clogging dilemma.

According to Table 3, a comparison of batch-type SF CWs with continuous aeration (at 1.5-day detention time) with the traditional operation (presented in Table 2), it appears that artificial aeration can compensate for the lack of plant-mediated oxygen supply and enhance the organic removal efficiencies in CWs. On the other hand, a comparison of continuously aerated SF with continuously aerated VSF CWs (Table 3) showed that the type of wetland system and HRT can play a key role in these systems. Moreover, based on the evidence of aerated batch-fed VDF CWs (at 3-day detention time), the systems that operated under intermediate aeration showed comparable removal efficiencies of organic pollutants to that operated under continuous aeration. Besides, as a performance improvement of wetland, the artificial aeration increases the aerobic microbial activities and enhances activation of the plant rhizomes and root systems (Tao et al. 2010). But, generally, the greater microbial activity could lead to a change of both microbial community structure and diversity and higher microbial biomass yield. In view of that, the possible long-term effect of artificial aeration on SSF CWs, such as choking and clogging difficulties, is lacking and should investigate the case further (Chazarenc et al. 2009). Artificial aeration anyhow requires energy input and additional cost.

The experimental results from tidal flow (TF) CWs (Table 3) showed that the system produced the highest organic pollutant removal efficiency with relatively short-flood and long-drain periods, highlighting the importance of oxygen transfer into the bed matrix. In fact, these types of wetlands

Table 4 Examples of design/operational parameters and treatment efficiency of post-treatment CWs

CW type	WT	Wastewater characteristics							HRT (day)	Flow rate (m^3/day)	HLR (m^3/m^2/day)	Vegetation	Reduction (%)					Study period (Months)	Country	Reference
		Temp. (°C)	pH	COD (mg/L)	BOD_5 (mg/L)	TSS (mg/L)	NH_4-N (mg/L)	TP (mg/L)					COD	BOD_5	TSS	NH_4-N	TP			
HSF	PE	–	–	266	219	–	–	2.43	1.64	0.79	0.2	*Typha* sp.	74	76	–	–	85	24	Indonesia	(Hendrawan et al. 2013)
HSF	PE	–	7.2	123	–	54	–	8.9	1	0.44	0.22	*Canna* sp.	59	–	93	–	12	–	Thailand	(Konnerup et al. 2009)
HSF	PE	–	7.23	136	–	65	–	9.2	2	0.22	0.11	*Canna* sp.	73	–	95	–	23		Thailand	(Konnerup et al. 2009)
HSF	PE	–	7.26	135	–	47	–	9.8	4	0.11	0.055	*Canna* sp.	83	–	96	–	35		Thailand	(Konnerup et al. 2009)
HSF	PE	–	7.2	123	–	54	–	8.9	1	0.44	0.22	*Heliconia* sp.	58	–	92	–	6		Thailand	(Konnerup et al. 2009)
HSF	PE	–	7.23	136	–	65	–	9.2	2	0.22	0.11	*Heliconia* sp.	72	–	95	–	7		Thailand	(Konnerup et al. 2009)
HSF	PE	–	7.26	135	–	47	–	9.8	4	0.11	0.055	*Heliconia* sp.	79	–	97	–	13		Thailand	(Konnerup et al. 2009)
Baffled HSF	PE	23–34	–	156	–	–	33	–	2	0.25	0.25	*Typha* sp.	59	–	–	74	–	12	Malaysia	(Tee et al. 2012)
Baffled HSF	PE	23–34	–	150	–	–	33	–	3	0.16	0.16	*Typha* sp.	67	–	–	84	–		Malaysia	(Tee et al. 2012)
Baffled HSF	PE	23–34	–	160	–	–	37	–	5	0.1	0.1	*Typha* sp.	79	–	–	99	–		Malaysia	(Tee et al. 2012)
HSF	AnE	–	–	–	116	62	–	13.4	3.5	1	0.063	*Phragmites* sp.	–	82	77	–	47	–	Korea	(Ham et al. 2007)
HSF	AnE	–	–	197	120	79	52	4.5	3	0.146	0.073	*Phragmites* sp.	71	83	94	89	33	–	Saudi Arabia	(El-Khateeb and El-Bahrawy 2013)
HSF	AnE	11–30	–	355	–	–	34	12	8	0.6	0.03	*Phragmites* sp.	70	–	–	21	52	12	Italy	(Mietto and Borin 2013)
VDF	AnE	11–30	–	355	–	–	34	12	2.7	1.2	0.21	*Phragmites* sp.	92	–	–	94	27		Italy	(Mietto and Borin 2013)
Drop-aerated VDF	AnE	12–16[a]	7.7	164	69	41	–	–	3.6	0.144	0.192	–	80	91	84	–	–	–	China	(Zou et al. 2012)
Bio-rack	AnE	25–29[a]	7.1	420	150	245	–	–	0.4	0.126	1.31	*Phragmites* sp.	77	84	75	–	–	12	Iran	(Jamshidi et al. 2014)
Bio-rack	AnE	25–29[a]	7.15	290	110	237	–	–	0.52	0.096	1	*Typha* sp.	66	76	70	–	–		Iran	(Jamshidi et al. 2014)
HSF	SPE	21[a]	7.5	276	112	–	–	16	6	0.28	0.1	*Typha* sp.	64	72	–	–	31	5	Brazil	(dos Santos et al. 2013)

HSF horizontal subsurface flow; *VDF* vertical down-flow; *WT* means wastewater type, including primary effluent (*PE*), anaerobic effluent (*AnE*), and stabilization pond effluent (*SPE*)

[a] Air temperature

are hydraulically efficient because they resist formation of preferential flow paths, and they can build up as deep as 1.2 m without losing hydraulic efficiency. As a result, deeper systems occupy less land area, thereby broadening the number of potential wetland treatment sites (Austin 2006). Whereas, in cold climates, wetland beds may face with problem of freeze solid (Austin and Nivala 2009). Clogging could also build up over a period of time. This can result in limiting available voids in the bed for atmospheric air diffusion and treatment efficiency.

Based on the data in Table 4, the baffled HSF CW requires a long HRT (≥2 days) to reach the desired organic pollutant removal efficiency in treating domestic sewage (after pre-treatment), while the pollutant influent concentration was comparatively low in the system. In fact, the low influent pollutant concentrations and the longer HRT in the cited CWs can be due to the possible long-term effects, such as choking and clogging difficulties. However, wetland systems with long HRT also had adverse effects on effluent quality. Baffled HSF could be provided by a longer flow path provided by the up-flow and down-flow conditions, sequentially in the wetland resulting in more contact of the wastewater with the rhizomes and microaerobic zones (Tee et al. 2012). Yet, the major disadvantage of traditional SSF CWs includes their tendency for clogging, and overall system costs would exist in this type of wetland system.

The drop-aerated VDF CW found to be efficient in the treatment of pre-treated domestic wastewater (having relatively low pollutant concentration) under long HRT above 3 days (Table 4). In this system, there is no additional cost of energy consumption due to supplementary aeration. But, the low temperature would freeze the influent dropping device in cold climates. Nuisance odors and insect problems may occur because of the exposure of wastewater to the atmosphere in the influent dropping device. Clogging failure could also be a problem due to soil strata in this type of CWs.

As shown in Table 5, the use of hybrid systems does not eliminate the need for pre-treatment. For the treatment of domestic wastewater (100–500-mg COD/L), the hybrid systems were operated optimally in HRT with a range of 3–6 days. Accordingly, a higher organic pollutant removal rate can be obtained at a higher organic loading rate compared to those operated under a low organic loading rate. Zhao et al. (2011) reported that the average organic pollutant removal efficiencies ranged from 73 to 93 % by varying the influent COD/N ratios (2.5, 5, and 10) in two-stage hybrid systems (Q= 0.04 m^3/day, HLR=0.07–0.07 m^3/m^2/day) (Fig. 2). On the other hand, the highest organic matter removal occurred when the COD/N ratios were 5–10 for VDF-VUF CW. In hybrid systems, the advantages of various wetland techniques can be combined and improve the wastewater treatment. Despite this, the limitations can be derived from each individual CW as their initial state depending on the type of the process. Caution should be taken to avoid clogging problems, particularly when they are operated under high organic loading rates.

Nitrogen removal

As shown in Tables 2–5, the removal efficiency of ammonium nitrogen (NH_4-N) varied (20–99 %), which can be due to the input organic and nitrogen concentrations, oxygen transfer efficiency of the systems, aerobic microbial population within the wetland beds, process condition (i.e., hydraulic retention time), and a certain adsorption capacity of the materials in the wetland bed. According to Tables 2 and 3, the nitrogen removal rate (nitrification) could be limited by the high organic matter concentrations (≥300-mg BOD_5/L) presented in the influents in a single-unit CWs, while the organic removal efficiency remains constant. This might be because the available oxygen in wastewater with a high BOD and nitrogen content is utilized quickly by heterotrophic bacteria for the metabolism of organic carbon (Ciria et al. 2005). As a result, clogging of the bed can be observed due to the excessive heterotrophic biofilm growth. Based on Table 3, the low nitrogen removal efficiencies may also be due to the high nitrogen levels presented in the influent (≥70 mg NH_4-N/L).

Nevertheless, in step-feed CWs (Table 3), the nitrogen removal efficiency may be altered depending on the step-feeding scheme. The gradual wastewater inflow at three points across the wetland length with sequence percentages of 60, 25, and 15 % of the total influent volume is more effective in enhancing the level of nitrogen elimination in the step-feed HSF CWs. Fan et al. (2013a) stated that the classic route of denitrification in CWs with step-feeding strategy is generally not completed and the TN removal remains at a low level. However, the additional carbon source supply in the new influent may believe to improve the denitrification. In the cited aerated batch-fed VDF CWs (3-day HRT) (Table 3), the improved simultaneous removal of organic matter and nitrogen for high-strength nitrogenous organic wastewater can also be realized by incorporating supplementary aeration. Therefore, the intermittent aeration could be an appropriate method for enhancing the TN removal efficiency than continuous aeration due to the nitrification and denitrification conditions that occur simultaneously. Fan et al. (2013a, b) reported intermittently aerated VSF CWs with a removal efficiency of 96–99 % NH_4-N and 26–94 % TN depending on the influent COD/N ratio (from 2.5 to 20). Based on the tidal CWs cited (Table 3), a total bed rest time of 2.5 h in an approximately 8-h cycle (i.e., short-flood and long-drain periods) was adequate for complete nitrification (≥94 %). Under this condition, the system achieved effective denitrification when the influent COD/N ratio was above 7 (≥85 % TN reduction) (Hu et al. 2014). By considering the results of the cited CWs that received pre-treated domestic sewage (Table 4), it can be verified that the traditional-type VDF CW provides higher nitrification activity compared

Table 5 Examples of design/operational parameters and treatment efficiency of hybrid CWs

CW type	WT	Wastewater characteristics							HRT (day)	Flow rate (m^3/day)	HLR (m^3/m^2/day)	Vegetation	Overall Reduction (%)					Study period (Months)	Country	Reference
		Temp. (°C)	pH	COD (mg/L)	BOD_5 (mg/L)	TSS (mg/L)	NH_4-N (mg/L)	TP (mg/L)					COD	BOD_5	TSS	NH_4-N	TP			
VDF-HSF	R	17–29	–	462	310	80	124	–	3	0.07	0.33–0.1	*Phragmites* sp. and *Scirpus* sp.	83	87	95	85	–	8	Spain	(Melian et al. 2010)
VDF-HSF	R	17–29	–	274	162	72	122	–	6	0.036	0.17–0.05	*Phragmites* sp. and *Scirpus* sp.	74	85	95	91	–		Spain	(Melian et al. 2010)
VDF-VUF	S	–	7.21	289	–	–	–	3	1.2	0.5	0.5–0.5	*Typha* sp.-*A. donax*	61	–	–	–	51	6	China	(Chang et al. 2012)
VDF-VUF	S	–	7.21	289	–	–	–	3	1.2	0.5	0.5–0.5	*Canna* sp.-*Pontederia* sp.	64	–	–	–	50		China	(Chang et al. 2012)
HSF-VDF	AnE	10–27	7.5	115	41	26	15	5.1	3	27	0.17–0.15	*Phragmites* sp.	94	95	85	–	94	17	Italy	(Masi and Martinuzzi 2007)
VDF-HSF-SF	AnE	15–25	7.8	189	133	25	20	–	4	0.2	0.06–0.1–0.1	*Phragmites* sp.	74	90	84	96	–	11	Spain	(Ávila et al. 2013a)
VDF-HSF-SF	AnE	18–24	7.5	294	204	98	43	6.8	–	14	0.04–0.06–0.06	*Phragmites* sp.-*Phragmites* sp.-six species	83	97	94	95	22	14	Spain	(Ávila et al. 2013b)
VDF-HSF-SF	AnE	13–15	7.8	549	–	27	31	–	4	0.2	0.06–0.1–0.1	*Phragmites* sp.	91	–	93	97	–	–	Spain	(Ávila et al. 2014)
VDF-HSF-SF	AnE	15–16	7.8	828	–	53	47	–	2	0.4	0.13–0.2–0.2	*Phragmites* sp.	89	–	96	72	–		Spain	(Ávila et al. 2014)
VDF-HSF-SF	AnE	18–20	7.5	868	–	60	43	–	1.5	0.55	0.18–0.28–0.28	*Phragmites* sp.	91	–	95	86	–		Spain	(Ávila et al. 2014)
THCW	AnE	−4.2–39[a]	7.11	320	–	124	46	6	2.7	43	0.32	Five species	85	–	89	83	64	8	China	(Ye and Li 2009)

HSF horizontal subsurface flow; *VDF* vertical down-flow; *VUF* vertical up-flow; *SF* surface flow; *THCW* towery hybrid CW; *WT* means wastewater type, including raw sewage (*R*), synthetic sewage (*S*), anaerobic effluent (*AnE*), and stabilization pond effluent (*SPE*)

[a] Air temperature

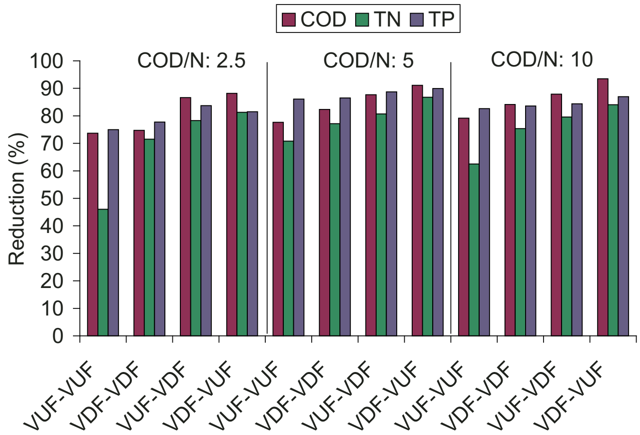

Fig. 2 Pollutant removal efficiencies of four types of two-stage hybrid CWs fed with synthetic domestic wastewater (*study period* 9 months, *vegetation A. calamus*, *influent temperature* 8–41 °C) (Zhao et al. 2011)

to that of traditional HSF CWs. In hybrid systems (Table 5), such as the cited VDF-HSF, nitrification could be limited slightly when they are operated under a high organic loading rate. On the other hand, the high nitrogen concentration in wastewater might not have an effect on the ammonia nitrogen removal performance in hybrid systems (as VDF-HSF). Tower and three-stage (VDF-HSF-SF) hybrid systems can achieve a TN>80 % (Ávila et al. 2013b; Ye and Li 2009). Tuncsiper (2009) reported the highest nitrogen removal efficiency (≥80 %) in a two-stage hybrid system (HSF-VSF) by 100 % recycling and HLR of 0.3 m^3/m^2/day (Table 6). Therefore, the removal efficiency has been increased by increasing the recycling ratio and decreasing the HLR. Zhao et al. (2011) reported average removal efficiencies of 46–87 % for TN by varying the influent COD/N ratios (2.5, 5, and 10) in two-stage hybrid systems, and VDF-VUF CW showed the highest TN reduction at a COD/N ratio of 5 (Fig. 2). As a result, the two hybrid systems of VUF-VUF and VDF-VDF have the lowest TN removal efficiencies because of their inability to achieve the aerobic/anaerobic conditions for the nitrification/denitrification.

Phosphorous removal

As reported in previous studies (Tables 2–5), the phosphorus removal efficiency in CWs varies considerably (6–99 %), depending on the wetland design, environmental condition, and loading rate. The elevated phosphorus elimination in the cited CWs can be due directly to the plant uptake process, more stable temperature during wetland operation, and long contact time within the wetland units. Similarly, the high removal of phosphorus could be due to ligand exchange reactions within the bed (Vohla et al. 2011). In a proposed design system, increasing the hydraulic retention time can ameliorate the treatment performance. The removal of phosphorus is normally low, and typically, amounts of only 40 to 60 % are observed during the treatment of domestic sewage (Vymazal 2004). In particular, Zhao et al. (2011) reported an average reduction of 75 to 90 % TP by varying the influent COD/N ratios (2.5, 5, and 10) in two-stage hybrid systems (Fig. 2). Therefore, VDF-VUF CW with COD/N ratio of 5 was verified for the highest TP reduction.

Environmental stress condition

High TDS concentration and desalination

Domestic wastewater in coastal areas may be contaminated with high concentrations of total dissolved solids ranging from 400 to 3000-mg TDS/L (Valipour et al. 2014b). The tolerance to high TDS stresses in aquatic macrophytes is a coordinated action in the treatment performance. Caution must be taken for plant selection when dealing with high TDS-contaminated wastewater because they should be stress-tolerant plants to achieve a resilient and effective wetland system. Under high TDS stress wastewater, plants' stand using several physiological mechanisms include the sequestration of TDS in the vacuoles of the cells, TDS exclusion from the transpiration stream, excreting excess TDS through the TDS glands, and preventing TDS uptake into the roots (Tuteja 2007). Note that TDS affects plants in different ways, such as osmotic effects, specific ion toxicity, and/or nutritional disorders. The extent to which one mechanism affects the plant over the others depends upon many factors, including the species, genotype, plant age, ionic strength, and composition of the solution (Läuchli and Grattan 2007). In that order, halophytic plants have long been suggested for treating wastewater contaminated with high TDS; however, the use of halophytes to reduce TDS (desalination) is a novel strategy. To act as an accumulator of TDS, the plant needs to tolerate a wide gradient of TDS, grow in wetlands, and accumulates enough ions within its tissues to reduce the TDS of wastewater (Shelef et al. 2013). In general, the systematic data regarding phytoremediation of TDS-contaminated wastewater are scarce.

An investigation of the use of bio-rack and shallow pond systems in treating TDS-contaminated wastewater showed that the *Phragmites* sp., *Typha* sp., and water hyacinth can tolerate TDS up to 9000, 2500, and 2000 mg/L, respectively (Valipour et al. 2010, 2011a, 2014b). Beyond these concentrations, the plants were highly damaged and the COD level in treated effluent reached above 100 mg/L. At these detention limits, the removal efficiencies were close to 80 % in COD and 20 % in TDS (Table 7). These results suggest that aquatic macrophytes (such as *Phragmaties* sp., *Typha* sp., and water hyacinth) play a role in desalination by the accumulation of TDS in their tissues because a decrease in TDS was observed in these biological processes. *Phragmaties* sp. may be referred

Table 6 Mean nitrogen removal at different recycling ratios and HLR in a two-stage hybrid CW fed with pre-treated domestic sewage

Recirculation ratio	Nitrogen components	HLR 0.1 $m^3/m^2/day$		HLR 0.8 $m^3/m^2/day$		HLR 0.06 $m^3/m^2/day$		HLR 0.03 $m^3/m^2/day$	
		HSF (%)	VSF (%)	HSF (%)	VSF (%)	HSF (%)	VSF (%)	HSF (%)	VSF (%)
0	TN	35	60	45	63	48	50	63	83
	NH_4-N	37	60	45	65	47	50	62	87
50	TN	40	60	50	63	57	52	73	93
	NH_4-N	37	63	50	68	53	53	73	97
100	TN	53	62	57	50	63	60	80	95
	NH_4-N	50	65	53	52	60	63	80	98

Vegetation Iris sp.-*Phragmites* sp., *influent temp*erature −20 °C, *study period* 15 months

Source: Tuncsiper (2009)

more appropriately to a TDS-tolerant plant due to the osmotic adjustment with attribution of a high K^+ content in shoot tissues. In addition, the same desalination phenomenon has also been reported to occur in halophyte *Bassia indica* by the accumulation of 10 % Na of its dry weight (Shelef et al. 2013).

Cold climate operation

The best prospects for successful wetland treatment should be in the warmer regions. On the other hand, the application of this system under cold climate has also been reported. From the North America Treatment System Database, 176 wetland treatment sites in Canada and the USA have been listed, of which approximately 60 % of these are located in cold climate regions. Almost all the cold climate wetlands are SF, and 90 % of them are treating domestic wastewater (Mæhlum 1999).

During the cold season, the microbial metabolism and their bioactivity are rather low; whereas with increasing temperature, the plant biomasses and activities of the microorganisms increase at high speed, which results in a higher removal rate of the pollutant. For a multi-stage pond-wetland system in Dongying City, China, in the cold season, the removal efficiencies of COD, BOD_5, and NH_4-N were approximately 85, 40, and 20 %, respectively, while in the warm season, 92, 73, and 71 % of these pollutants were removed, respectively (Peng et al. 2005). A long hydraulic retention time could enable microorganisms to grow and reduce organic matters and nitrogen at low temperatures, but the capital investment in this system is higher.

In fact, the CW technologies need to be adapted to subfreezing environments, which would enhance the treatment performance. The plant species that provide structure year-round perform better than those species that die below the waterline after the onset of cold temperatures. For this reason, fast-growing emergent species (such as *Phragmites*, *Typha*, and *Scirpus*), which have high lignin contents and are adaptable to variable water depths, are most ideal for CWs during the cold seasons. In addition, under cold climate conditions, greater oxygen leakage from the roots of some plants can sometimes appear due to plant dormancy and reduced internal oxygen consumption. This may offset the possible oxygen limitation and improve the aerobic microbial respiration in the root zone throughout the cold condition operation (Stein and Hook 2005). The selection of appropriate plants may have added benefits in protecting the water surface from freezing, trap falling and drifting snow, enhance oxygen transfer, and reduce the heat loss effects of wind. In addition, some types of insulation strategies can also apply as a response of wetland system to cold climates.

In CWs (dominated by emergent plants), ice and snow can be used as an insulating layer. As an example, these types of insulation have been carried out in northern China (Li et al.

Table 7 Performance of shallow pond and bio-rack CWs treating domestic sewage seriously stressed with TDS at the optimal tolerance limit of plant species

CW type	HRT (day)	Flow rate (m^3/day)	HLR ($m^3/m^2/day$)	Inlet (mg/L)		Reduction (%)		Study period	Country
				COD	TDS	COD	TDS		
Bio-rack system (*Phragmiters* sp.)	0.42	0.127	1.14	456	8760	80	14	3	India
Bio-rack system (*Typha* sp.)	0.71	0.071	0.72	388	2512	76	21	3	Iran
Shallow pond system (Water hyacinth)	0.88	0.023	0.16	442	1992	81	19	3	India

Source: Valipour et al. (2010, 2011a, 2014b)

2011b). In practice, an ice-air blanket is often generated purposefully by raising the effluent levels to allow it to freeze and subsequently lower it, leaving an air gap between the ice layer and the effluent (Horváth 2012). A pilot-scale CW planted with *Phragmites* sp. (in Tianjin, northern China) under ice layers when the average air temperature is lower than −4 °C has shown a better effluent quality than a secondary treatment level, e.g., $BOD_5<20$ mg/L, SS<20 mg/L, TN<15 mg/L, and TP<0.6 mg/L (Yin and Shen 1995). Ice begins to form on a water surface when the water temperature reaches temperatures as low as 3 °C because of the density differences and convective losses. If the wastewater entering wetlands has a temperature >10 °C, aeration can be used to modify the surface water temperature near zero. The presence of an ice layer can be a benefit for providing insulation and decreasing the cooling of the underlying water, but the water flow will be reduced as the ice layer thickens. As a result, the constriction of flow beneath the ice layer leads to subsequent flooding, freezing, and hydraulic failure (Mæhlum 1999). Therefore, an increased depth up to 1 m is also recommended for SF (water depth) and SSF (effective depth) CWs. A Canadian FWS CW operated successfully over 4 years by raising the water level at freezing time (Mæhlum 1999; Heers 2006). On the other hand, in SSF CWs, the dead vegetation or mulch (such as poplar bark, wood chips, and reed-sedge peat) can be used as appropriate heat insulation materials to prevent freezing and resulting hydraulic failure throughout the cold season (Wallace et al. 2000).

The amount of mulch material required to protect a wetland bed from freezing damage is strongly dependent on the timing and amount of snowfall because snow cover itself is a significant insulator (Wallace and Nivala 2005). The use of well-decomposed organic materials to avoid degrading treatment efficiency is recommended. To be effective, the insulation must be uniform in coverage, which requires that it should be designed as an integral part of a wetland system (Wallace et al. 2000). Besides, the wetland surface can also be covered with a porous media having low thermal conductivity (such as expanded clay aggregates) which should be kept unsaturated during the cold season (Mæhlum 1999; Horváth 2012).

In a SSF CW covered with harvested vegetation, the average removal efficiencies obtained were 31 % for TP, 27 % for NH_4-N, and 10 % for TN during winter (−3 to 6 °C), which were 16, 10, and 5 % higher than that of the control wetland, respectively (Shen et al. 2007). An integrated household VSF CWs (using *Salix* sp.) in rural villages in northern China suggested that a 0.4-m insulating biomass layer (sawdust) maintained a bed temperature above 6 °C in the face of freezing temperatures (even at a very low temperature of −8 °C) during winter. The average removal efficiencies were 95 % BOD_5, 96 % TSS, 85 % NH_4-N, and 88 % TP during the winter period. A negligible increase in the average removal efficiencies for BOD_5, TSS, and NH_4-N out of winter (1.3, 1.1, and 5.4 %, respectively) was observed, whereas a 0.6 % increase was achieved for TP removal in the winter period (Wu et al. 2011a).

Moreover, artificial aeration can be used to overcome the effects of oxygen limitation in CWs caused by the winter dormancy and the mortality of vegetation and the need to use an insulating mulch layer in the bed. As a result, organic material and nitrogen pollutant removal are enhanced during the cold season. The oxygen solubility is higher in colder water, but gas exchange in HSF CWs might be reduced by the additional insulation layer and the fact that the plants are dormant (Ouellet-Plamondon et al. 2006; Horváth 2012). In SF CWs, artificial aeration can also create an ice-free zone during the cold season (Wallace et al. 2011). Another alternative option that has already been used in northern USA and Canada is to store discharged wastewater in a tank during the cold season and reflow it again through the wetland unit during spring time (Pries et al. 1996). The advantage of this practice is the use of a design for warm weather conditions, whereas the disadvantage is the cost of the storage lagoons. A seasonal storage for the winter waste load might be necessary in locations where extended periods of air temperature <−10 °C are experienced. A number of CWs in South Dakota and northwest Canada operated this way (Mæhlum 1999).

Future sustainability of CWs

Future research and development work will be needed for the successful and sustainable application of CWs, particularly in decentralization practices. In summary,

(1) Knowledge of traditional CWs has been moving to development, but critical studies to overcome the constraints (such as problems of clogging, odors, insect vectors, high land area, and capital investment) according to the types of advanced treatment systems ("Process modification" section) and achieve sustainable wastewater quality improvement will still be required. Accordingly, several localization frameworks in advance CWs should also be developed to provide a sustainable model for decentralization practices.
(2) In BAS CWs, plastic-type materials are used primarily for attached biofilm growth. The supporting media can be regarded as a critical step to provide an obligatory surface area for biomass concentration within the wetland unit. The greater surface area to volume ratio of the expended bed reactor can enhance the treatment performance. More research is needed on the application of a more sustainable material with a higher surface area in BAS CWs.
(3) Hybrid systems are mainly a combination of different types of traditional CWs. Further research should assess

the possible combination of various types of modified wetland processes to achieve high pollutant removal efficiencies.

(4) This is important for achieving a comparative assessment on the use of different aquatic weeds in each modified wetland processes under the same environmental conditions. Additionally, predicting under what circumstances the plant contribution to the wetland units will be more remarkable in the phytoremediation of TDS-contaminated wastewater will be the major challenge. Further analysis will also be needed to identify potential plant species either for TDS accumulation (in desalination practices) or for cold climate adaptation.

(5) During the cold season, the mulch materials can strongly affect the system performance and the establishment of introduced plants. Therefore, future research will need to verify the appropriate mulch layer over the main media. In addition, the research may also need to be extended beyond a focus on plant species that can tolerate the mulch layer.

(6) Research on the predominant microbial species and macrophytes with a specific gene for nitrogen removal may help optimize nitrogen removal in CWs.

Conclusions

The following conclusions can be drawn from the review:

(1) The interrelationship between vegetation, media materials, and living organisms is a major mechanism of pollutant removal in CWs.

(2) The types of wetland system, plant species, and operating condition are crucial to achieve a sustainable treatment performance. *Phragmites* sp. and water hyacinth can be strongly recommended in treating wastewater among other species.

(3) All CWs are found efficient in removing organic matters and suspended solids, but the required retention times differ considerably. The nutrient (N and P) removal efficiency can be influenced by plant species, oxygen availability, COD/N ratio, temperature, hydraulic retention time, and high sorption capacity media of the wetland bed.

(4) Besides the advantages in designing modified CWs, most of these systems are associated with technical and operational limitations. Among the various types of wetland systems, the BAS CWs tend to give a more consistent performance in the pollutant elimination (by lowering 29–97 % HRT), depending on the types of plant species and system configuration. The low HRT of system can reduce the overall capital cost due to low-footprint-area requirement in comparison with other CWs.

(5) With some simple operating guidelines, CWs can be operated successfully either in coastal regions, where domestic sewage is contaminated with high TDS concentration or in cold climates during winter seasons. The selection of plant species plays a major role in the adaptation of CWs for TDS tolerance and desalination practices. Operational strategies of CWs for cold seasons can initiate by selecting plant species, prolonging the hydraulic retention time, deeper the wetland bed, thermal insulation, artificial aeration, and wastewater storage.

(6) Overall, BAS CWs (especially bio-rack using *Phragmites* sp.) can be promising for decentralization. On the other hand, to reach fruition, further research will be needed for the successful and sustainable application of CWs.

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