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Review

Removal of Emerging Contaminants from Wastewater Using Constructed Wetlands: A Critical Review of Removal Performance, Mechanisms, and Toward Sustainability

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Abstract

The presence of emerging contaminants (ECs) in aquatic environments poses significant risks to human and ecological health. Constructed wetlands (CWs) have emerged as cost-effective and sustainable alternatives to conventional wastewater treatment systems for EC removal. This review synthesizes the current understanding of the effectiveness of constructed wetlands in removing 42 commonly detected emerging contaminants from wastewater. It highlights the major removal mechanisms, including photodegradation, microbial biodegradation, phytoremediation, and sorption, and examines how their performance is governed by contaminant properties, CW configuration, operational conditions, and environmental factors. For selected ECs, constructed wetlands have demonstrated higher reported removal efficiencies than conventional wastewater treatment plants (WWTPs) for several compounds; however, these comparisons are study- and compound-specific and may be influenced by differences in influent concentrations, treatment configurations, and operating conditions. Their removal performance is strongly influenced by wetland configuration, hydrological conditions, substrate characteristics, and vegetation type. Research gaps persist regarding the fate and transformation pathways of ECs and the influence of operational conditions on their removal efficiency and treatment mechanisms in CW systems. A comprehensive understanding of these factors is essential for optimizing the design and operation of CWs. The literature review indicates that sorption, photodegradation, microbial biodegradation, and phytoremediation constitute the principal mechanisms governing EC removal; however, their relative contributions vary considerably with wetland configuration, operating conditions, and contaminant-specific properties. Strengthening the mechanistic understanding of EC fate and removal in CWs is therefore crucial for enhancing treatment efficiency, improving system predictability, and effectively addressing the increasing environmental challenges posed by these contaminants.

Keywords: constructed wetlands; emerging contaminants; mechanism; occurrence; removal efficiency; wastewater

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1. Introduction

Emerging contaminants (ECs) are synthetic or naturally occurring chemicals or any microorganisms that are not commonly monitored in the environment but have the potential to enter the environment and cause known or suspected adverse ecological and/or human health effects [1,2]. ECs comprise a diverse group of anthropogenic chemicals originating from industrial, agricultural, pharmaceutical, and personal care activities, etc. [3,4]. This group includes pharmaceutical and personal care products, endocrine-disrupting compounds (EDCs), artificial sweeteners, bisphenol A, etc., which are increasingly detected in freshwater environments worldwide [1–4]. ECs generally present at trace concentrations ranging from ng/L to μg/L, and many ECs are of considerable environmental and public health concern because of their persistence, bioaccumulation potential, and adverse effects on aquatic organisms and human health [5,6]. Accurate detection and quantification remain challenging because they often require sophisticated, costly analytical techniques with high sensitivity and selectivity [7]. ECs are introduced into aquatic systems through multiple pathways, including domestic wastewater, wastewater treatment plant (WWTP) effluents, industrial discharges, agricultural runoff, and other diffuse sources [8–10] (Figure 1). Continuous or intermittent inputs, particularly from WWTPs, can maintain the environmental presence of these substances and contribute to their pseudo-persistence, thereby increasing the likelihood of prolonged exposure and potential mixture-related or synergistic effects [9,11]. The occurrence of ECs in wastewater and receiving waters can exert selective pressure on microbial communities, potentially promoting the selection, enrichment, and dissemination of antimicrobial resistance and other adaptive traits. Conventional wastewater treatment processes are often not designed to completely remove these contaminants, allowing residuals and associated resistance determinants to persist in treated effluents [12,13]. About 120 ECs have been reported in raw wastewater from 13–15 WWTPs across Europe and Asia, with an average of approximately 46 compounds detected per treatment plant [14,15]. Their occurrence and concentrations vary considerably among geographic regions and are influenced by local industrial activities, agricultural practices, population density, consumption patterns, and treatment efficiency. For example, the total EC concentrations in raw wastewater reported from China and Japan ranged from 2466 to 7995 ng/L, whereas substantially lower concentrations of 15–247 ng/L were reported in Sweden, Germany, Italy, and Switzerland [13,15,16]. Despite their occurrence at relatively low environmental concentrations, several ECs exhibit significant biological activity and toxicity, raising concerns regarding chronic exposure and long-term ecological impacts.

The widespread occurrence of pharmaceutical and industrial contaminants further highlights the potential human-health implications of EC exposure. Veterinary antibiotics are released into the environment in substantial quantities, contributing to the development and spread of antimicrobial resistance, which represents a major global public-health challenge [13,15]. Polychlorinated biphenyls (PCBs), for instance, have been reported to undergo maternal transfer through breastfeeding, potentially exposing infants to compounds associated with developmental and carcinogenic risks [17]. Pharmaceuticals, personal care products, endocrine-disrupting compounds, antibiotics, and pesticides have been detected in human tissues and biological fluids and may accumulate over time, with reported associations with metabolic, endocrine, hepatic, and other adverse health effects [15,16]. The persistence, mobility, and bioaccumulative characteristics of several ECs facilitate their transport and long-term occurrence in drinking-water supplies, wastewater networks, surface waters, and coastal environments. Their continuous environmental release, combined with the limitations of conventional treatment technologies, underscores the need for efficient, sustainable, and advanced remediation strategies capable of minimizing EC exposure and protecting aquatic ecosystems and public health [11,15]. Traditional wastewater treatment

technologies are not designed to remove ECs, and therefore the treatment of ECs within municipal wastewater treatment facilities is often incomplete or inadequate. Advanced treatment technologies such as ozonation, membrane filtration, and ultraviolet radiation [13,18] are effective for the removal of some ECs [11], but they are energy-intensive and expensive, making them impractical for many communities. Therefore, more economically and environmentally friendly EC removal solutions have been developed as a result of interest in nature-based technologies [3,19]. Through natural processes like biodegradation, sorption, photodegradation, and phytoremediation, wetlands, both natural and artificial, can absorb and change nutrients and other elements [20]. Organic matter, suspended particles, total nitrogen, total phosphorus, biochemical oxygen demand, chemical oxygen demand, and organic xenobiotics such as ECs have all been shown to be effectively removed by these naturally occurring systems [21,22].

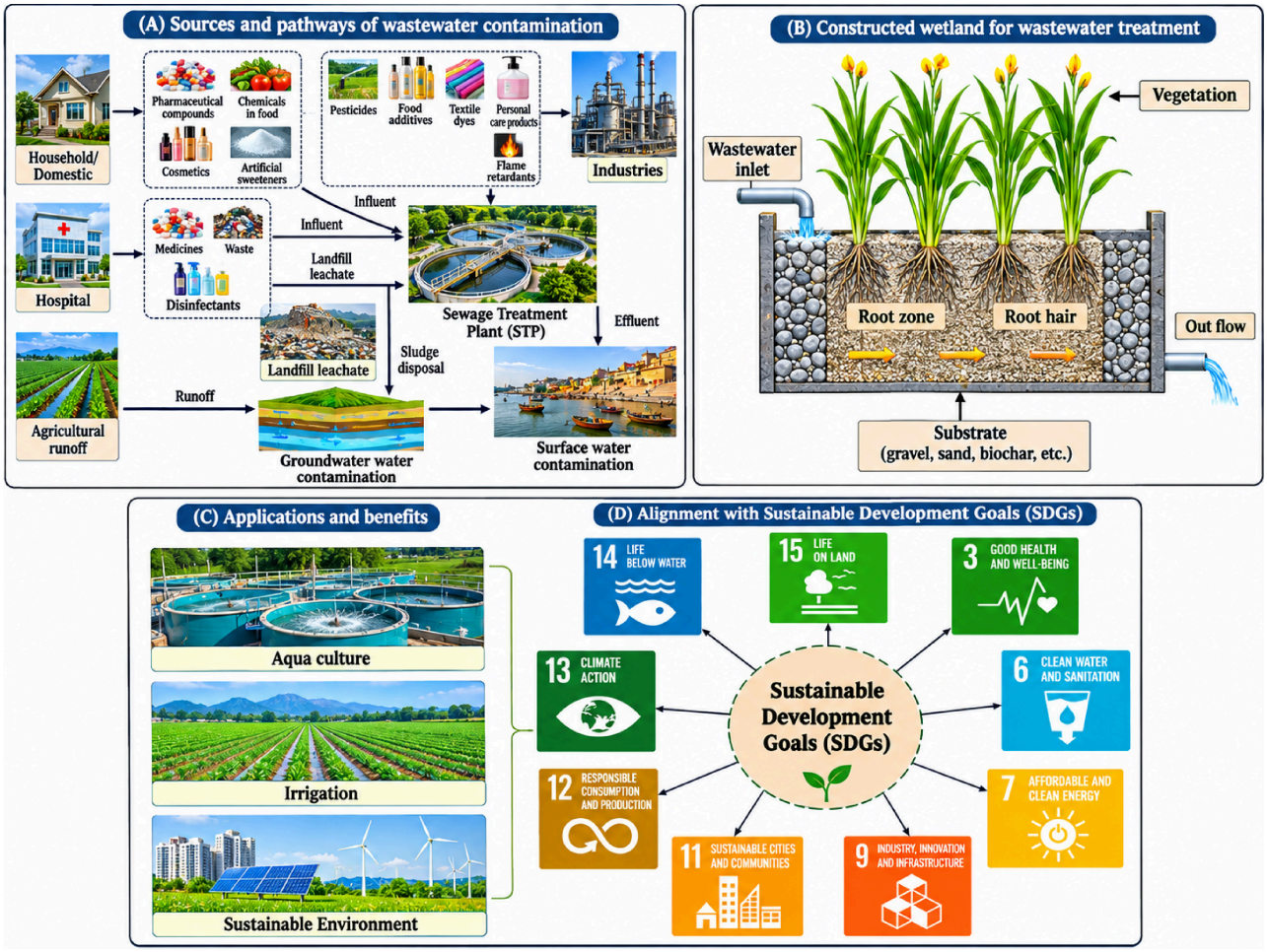


Figure 1. Schematic diagram showing the relationship between constructed wetlands (CWs) and the Sustainable Development Goals.

Matamoros et al. [23,24] compared removal efficiencies of six pharmaceutical compounds (PhCs) between HFCWs and conventional WWTPs, finding compound-specific differences (ibuprofen, naproxen > 70%, carbamazepine < 20%) and seasonal effects. Vymazal et al. [25] studied 32 PhCs in HFCWs, noting high removal for acetaminophen (91%), caffeine (84%), and furosemide (75%), but low for warfarin (31%) and gabapentin (14%). Hijosa-Valsero et al. [26] confirmed these trends, showing better CW performance, especially in planted systems. Previous studies focused on EC removal by CWs [11,23,25–29]. Imfeld et al. [30] and Zhang et al. [28] provided comprehensive overviews of CW removal

processes. Li et al. [10] examined design parameters influencing EC removal, while Verlicchi and Zambello [27] assessed CWs in primary, secondary, and tertiary treatment (Figure 2I). Carvalho et al. [31] explored plant–pharmaceutical interactions in the context of phytoremediation, while Gorito et al. [11] focused on pharmaceutical contaminants listed on the EU Watch List. However, further research is required to systematically synthesize and generalize findings across these interconnected thematic areas. Under appropriate conditions, the sustainability potential of constructed wetlands may be attributed to their relatively low energy requirements, reduced operational costs, and capacity to utilize solar energy and biological processes for contaminant attenuation [21,23]. In wastewater treatment, CWs can potentially provide multiple ecosystem services, including biodiversity enhancement, carbon sequestration, groundwater recharge, and the provision of green spaces in urban and peri-urban environments [23,27]. These potential benefits should be considered alongside trade-offs such as land requirements, variability in treatment performance, maintenance needs, and the possible accumulation or transformation of contaminants within wetland substrates and biomass [3,19,32]. From a circular economy perspective, constructed wetlands can provide opportunities for nutrient recovery, particularly nitrogen and phosphorus, biomass generation, and integration with complementary nature-based treatment systems, thereby contributing to resource recovery and closed-loop wastewater management [23,27]. The findings indicate that hybrid CW configurations integrating vertical and horizontal subsurface-flow systems (Figure 2I,II) can enhance the removal of both hydrophobic and hydrophilic emerging contaminants by exploiting complementary physicochemical, biological, and phytoremediation processes [32]. The integration of novel materials (e.g., biochar, zeolites, and nanomaterials) and innovative plant species (e.g., halophytes and hyperaccumulators) into CWs further enhance contaminant retention and system resilience [17]. Climate-resilient CW designs can adapt to variable hydraulic and pollutant-loading conditions, making them suitable for decentralised wastewater treatment in both developed and developing regions [17,19,32]. In terms of Sustainable Development Goals, CWs align with SDG 3 (Good Health and Well-being), SDG 6 (clean water and sanitation), SDG 7 (affordable and clean energy), SDG 11 (sustainable cities and communities), SDG 13 (climate action), SDG 14 (life below water), and SDG 15 (life on land) (Figure 1). To enhance long-term efficacy and sustainability, future studies should prioritize process optimization, life cycle assessment, and the scalability of treatment systems under realistic operating conditions. Constructed wetlands represent a promising pathway towards sustainable water management, offering an environmentally benign and socially acceptable solution for mitigating the impacts of emerging contaminants on water quality and ecosystem health [33–38]. Despite the potential of constructed wetlands for emerging contaminant removal, key gaps remain in understanding transformation products, mineralization, microbial–plant interactions, mixture effects, seasonal variability and effluent ecotoxicity. Future research should prioritize standardized performance metrics, long-term full-scale validation, and integrated life cycle, techno-economic, greenhouse-gas, resource-recovery, and climate-resilience assessments to establish the sustainability and scalability of CWs.

This study aims to systematically evaluate the removal efficiency of emerging contaminants in treated wastewater using constructed wetlands and elucidate the physicochemical, biological, and phytoremediation mechanisms governing their attenuation. Particular emphasis is placed on assessing the potential of CWs as sustainable, nature-based treatment systems for mitigating EC contamination, enhancing wastewater quality, and contributing to environmental protection and public health.

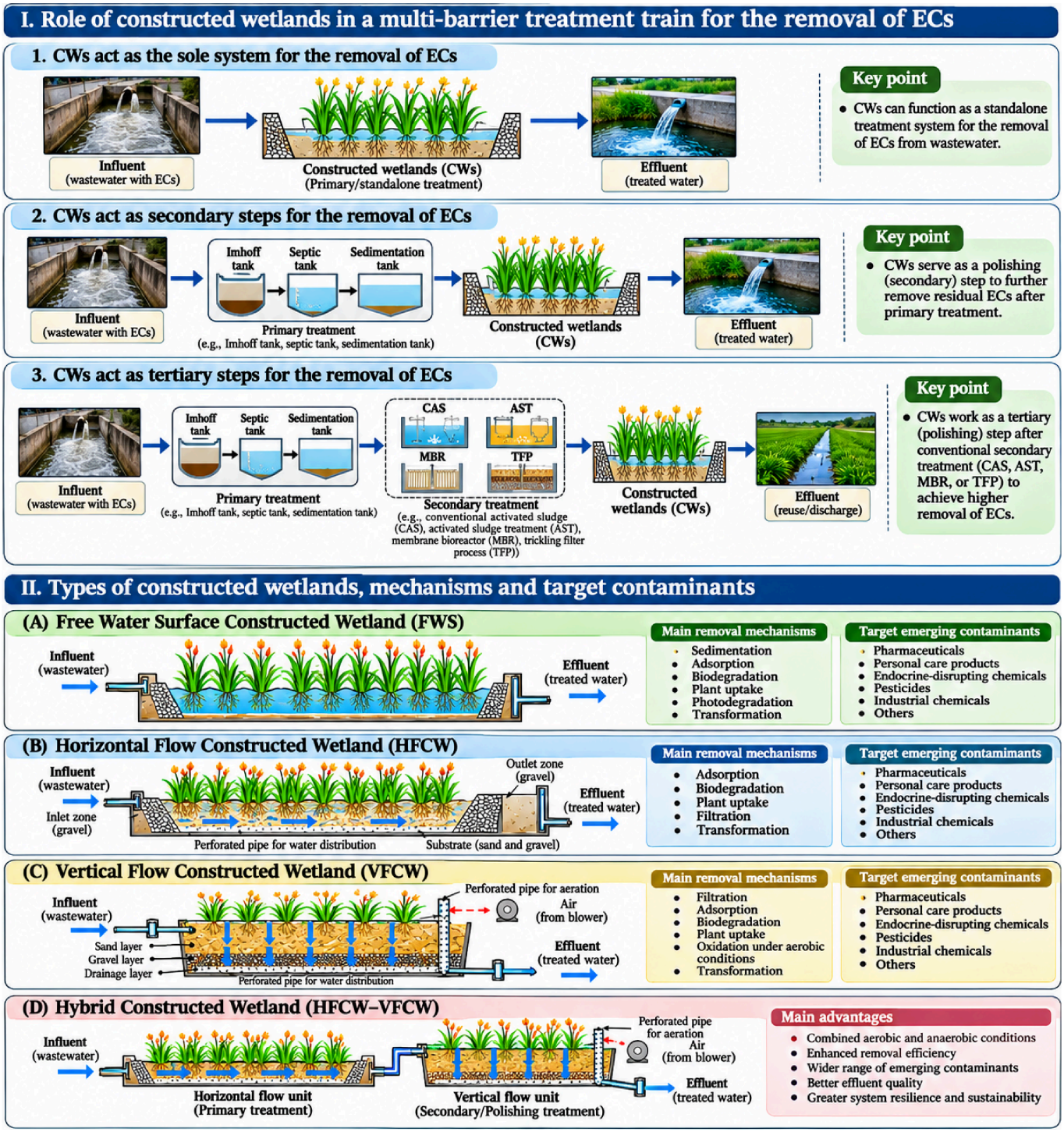


Figure 2. Schematic representation of CW applications for emerging contaminant removal as (**I**) sole, secondary, and tertiary treatment systems, and (**II**) the major CW types, including FWS, HFCW, VFCW, and hybrid systems, with their key removal mechanisms and target ECs.

2. Review Methodology

A structured literature review was conducted to identify, evaluate, and synthesize published research on the occurrence, fate, transport, transformation, removal, and treatment of emerging contaminants in water and wastewater, with particular emphasis on their removal using constructed wetlands. The literature search was performed using Web of Science, Scopus, Google Scholar, X-MOL, and Connected Papers, covering publications from 2010 to 2025. The literature search strategy employed combinations of relevant keywords and

search terms including "emerging contaminants" OR "pharmaceuticals" OR "pharmaceutical and personal care products" OR "personal care products" OR "endocrine-disrupting compounds" OR "artificial sweeteners" AND "water" OR "wastewater" OR "fate and transport" OR "removal mechanism" OR "treatment" OR "constructed wetlands". Studies were initially screened based on title and keyword relevance, followed by the assessment of abstracts and full texts where necessary. Quantitative data were extracted from multiple eligible studies rather than from one or two individual publications. For each EC, reported quantitative removal efficiencies from the relevant studies were compiled, and where comparable, averaged to obtain an overall representative removal value. This approach was adopted to provide a broader synthesis of the available evidence rather than relying on the findings of individual studies.

Peer-reviewed research articles and relevant review papers addressing EC occurrence, transformation, removal, or treatment, particularly through CWs, were included. Duplicate records were removed, and publications outside the 2010–2025 period were excluded from the core literature synthesis. Foundational studies were considered where necessary to provide context for fundamental concepts and well-established mechanisms. The selected literature was critically evaluated to synthesize removal performance, influencing factors, removal mechanisms, sustainability considerations, limitations, and research gaps. The ECs included in the review were selected based on their frequent occurrence in water and wastewater, environmental relevance, persistence, and the availability of evidence regarding their removal in CW systems. Particular emphasis was given to compounds for which sufficient information was available on their occurrence, quantitative removal efficiency, and associated removal mechanisms. This selection ensured that the quantitative synthesis was supported by multiple published studies and that comparisons among ECs were based on an adequately documented evidence base.

3. Occurrence and Environmental Concern of Emerging Contaminants (ECs)

The environmental occurrence, fate, and potential biological effects of ECs have attracted considerable global scientific attention, resulting in their recognition as an important class of emerging pollutants [3]. ECs are introduced into aquatic environments through both point and non-point sources [3,36]. Point sources include sewage treatment plants, hospitals, and industrial facilities, whereas non-point inputs arise from agricultural runoff, stormwater runoff, atmospheric deposition, landfill leachates, and septic systems [3,39] (Figure 3). Release of ECs may undergo a range of biotic and abiotic transformations governed by environmental conditions; however, many compounds are only partially metabolized, and unmetabolized parent compounds and transformation products can be excreted through urine and faeces and subsequently transported into wastewater streams [39–41]. Surface waters, including rivers, lakes, and ponds, receive ECs from diverse sources, with treated and untreated wastewater representing major pathways. The continuous discharge of wastewater introduces a complex mixture of pharmaceuticals, personal care products, industrial chemicals, and other micropollutants into receiving water bodies [27]. Several ECs, particularly those used in emulsifiers, adhesives, pesticides, and related commercial products, exhibit a strong tendency to associate with suspended matter and sediments because of their physicochemical properties [3,13,42]. Alkylphenols, EDCs, and PPCPs have been detected in surface waters, landfill leachates, and effluents from municipal and industrial wastewater treatment plants [16,19]. Estrogenic hormones have also been reported in surface waters and municipal wastewater effluents at concentrations capable of inducing endocrine disturbances and reproductive effects in aquatic organisms, particularly fish [43]. Their occurrence in wastewater is largely associated with human excretion

and may be influenced by physiological and hormonal conditions, including pregnancy, oral contraceptive use, and hormone replacement therapy [8,44]. PPCPs originate from numerous anthropogenic activities, including pharmaceutical manufacturing, domestic consumption, healthcare facilities, and municipal wastewater generation. Industrial discharges, hospital effluents, and municipal wastewater therefore constitute important pathways for their transport into aquatic ecosystems [3,32,41]. Agricultural activities provide another significant source of micropollutants, particularly through the extensive use of pesticides for crop protection and disease and pest management. These compounds may be mobilized from agricultural land and transported to adjacent water bodies through surface runoff and leaching [3,39]. Veterinary pharmaceuticals and feed additives administered to livestock can be excreted and subsequently enter surrounding soils and water bodies through runoff, infiltration, and leaching processes [3]. The distribution of ECs in freshwater environments, together with their persistence, biological activity, and potential adverse effects, represents an increasing concern for ecological integrity and public health [43]. Steroid hormones are of particular concern because of their potent endocrine-disrupting activity and ability to interfere with physiological processes at very low concentrations. Even at ng/L levels, certain steroid hormones can affect the development, growth, reproduction, ageing, and survival of aquatic organisms, potentially resulting in population-level and ecosystem-scale consequences [43,45,46]. Estrogenic compounds present in aquatic environments have been implicated in reproductive abnormalities in both humans and wildlife, including the feminization of male fish [19,41,43,47] (Figure 3). Their environmental persistence may contribute to the proliferation of antibiotic resistance genes in soil microbial communities [43]. Among the PPCPs, degradation products such as nonylphenol (NP) and octylphenol (OP) exhibit significantly greater toxicity than their parent compounds and are recognized EDCs. Experimental studies have demonstrated that both NP and OP can induce vitellogenin (VTG) production and feminization in male fish at concentrations as low as parts per billion, with OP exhibiting approximately 20-fold higher estrogenic activity than NP [6,45,48–53]. The physicochemical characteristics and classification of ECs highlight their potential adverse effects, particularly on human health and aquatic organisms in water and wastewater (Table 1 & Figure 3). Reported health concerns include cardiovascular disorders, hypertension, anaemia, reduced IQ, memory impairment, neurological disorders, and reproductive dysfunction (Figure 3). These effects suggest that prolonged or repeated exposure to ECs may adversely affect multiple physiological systems. Figure 3 underscores the environmental and public-health implications of EC contamination and highlights the importance of effective monitoring, risk assessment, and appropriate treatment strategies for their removal.

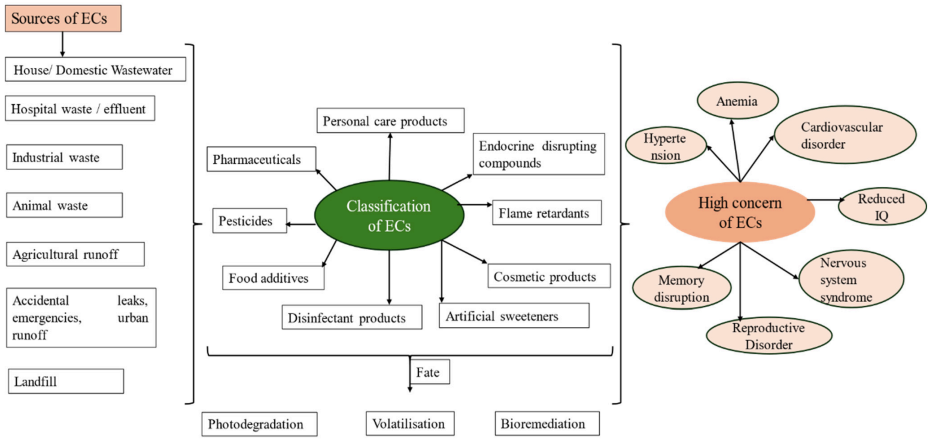


Figure 3. Sources, types, and health concerns of ECs.

Table 1. Physicochemical properties and environmental behaviour of selected emerging contaminants (ECs) in water and wastewater.

Sl. No.	Compounds of ECs	CAS No.	Physicochemical Properties						Behaviour of ECs	References
			Mol. Formula & Mol. Wt.	Henry Constant (H) ($Pa \cdot m^3\ mol^{-1}$)	pKa	Log K_{ow}	S^w	pH	Tempera Ture (°C)	
1	Paracetamol/ Acetaminophen	103902	$C_8H_9NO_2$ 151.16	6.42×10^{-13}	9.38	0.46	1.40×10^4	5.5–6.5	22.15	Polar	[15,22]
2	Carbamazepine	298-46-4	$C_{15}H_{12}ON_2$ 236.27	1.08×10^{-10}	13.9	2.45	17.7	7	25	Polar	[3,27]
3	Caffeine	58082	$C_8H1_0O_2N$ 194.19	1.90×10^{-19}	10.4	−0.07	2.16×10^4	7.8	23.12	Polar	[4,25,32]
4	Ibuprofen	15687-27-1	$C_{13}H_{18}O_2$ 206.28	1.50×10^{-7}	4.91	3.97	21	7	25	Non-polar	[37,45]
5	Ketoprofen	22071-15-4	$C_{16}H_{14}O_3$ 254.28	2.12×10^{-11}	4.45	3.12	51	7	25	Non-Polar	[19,31]
6	Diclofenac	15307-86-5	$C_{14}H_{11}C_{l2}O_2N$ 296.15	4.73×10^{-12}	4.15	4.57	2.37	8.1	21	Non-Polar	[31,52–54]
7	Naproxen	22204-53-1	$C_{14}H_{14}O_3$ 230.26	3.39×10^{-10}	4.15	3.18	15.9	6.5	24.2	Polar	[48,49,55]
8	Clofibric acid	882-09-7	$C_{10}H_{11}O_3Cl$ 214.64	NA	NA	2.57	NA	NA	NA	Polar	[49,56–58]
9	Amoxicillin	26787-78-0	$C_{16}H_{19}O_5N_3S$ 365.4	2.49×10^{-21}	NA	0.87	3.43×10^3	NA	25	Polar/ hydrophilic	[51,59,60]
10	Erythromycin	114-07-8	$C_{37}H_{67}O_{13}N$ 733.93	NA	8.8	2.83	2.0×10^3	NA	25	Polar	[50,61–63]
11	Salicylic acid	69-72-7	$C_7H_6O_3$ 138.12	7.34×10^{-9}	2.97	2.26	2240	6.2–7.8	24.6	Polar	[48,50,52,55]
12	Atenolol	29122-68-7	$C_{14}H_{22}O_3N_2$ 266.34	1.37×10^{-18}	NA	0.16	1.33×10^{-4}	NA	25	Polar	[49,57,64]

Table 1. *Cont.*

Sl. No.	Compounds of ECs	CAS No.	Physicochemical Properties							Behaviour of ECs	References
			Mol. Formula & Mol. Wt.	Henry Constant (H) ($Pa \cdot m^3\ mol^{-1}$)	pKa	Log K_{ow}	S^w	pH	Tempera Ture (°C)		
13	Metoprolol	51384-51-1	$C_{15}H_{25}O_3N$ 267.3665	1.40×10^{-13}	NA	1.88	1.69×10^4	6.5–7.7	NA	Polar	[56,59,61]
14	Sulfamethoxazole	723-46-6	$C_{10}H_{11}O_3N_3S$ 253.28	6.42×10^{-13}	NA	0.89	610	NA	25	Polar	[20,65–67]
15	Sulfadimethoxine	122-11-2	$C_{12}H_{14}O_4N_4S$ 310.329	NA	NA	1.63	343	5.8–6.1	21	Polar	[19,68–72]
16	Triclosan	3380-34-5	$C_{12}H_7O_2Cl_3$ 289.54	4.99×10^{-9}	NA	4.76	10	8.1	20	Non-polar	[41,55,59,62,65,73]
17	Triclocarban	101-20-2	$C_{13}H_9Cl_3N_2O$ 315.59	NA	NA	4.90	NA	8.5	20	Non-Polar	[41,44,62,67,74,75]
18	Mehylparaben	99-76-3	$C_8H_8O_3$ 152.15	1.72×10^{-7}	8.17	1.96	2500	6.5	23	Polar	[26,76–80]
19	Bisphenol A (BPA)	80-05-7	$C_{15}H_{16}O_2$ 228.29	2.87×10^{-6}	9.8	3.6	210	6.2	NA	Non-polar	[63,67,81–86]
20	Iopromide	73334-07-3	$C_{18}H_{24}I_3N_3O_8$ 791.12	5.18×10^{-18}	NA	−2.30	2.4	7	25	Polar	[57,67,87–93]
21	Furosemide	54-31-9	$C_{12}H_{11}ClN_2O_5S$ 330.74	5.07×10^{-12}	3.9	2.03	10	7.2	24.7	Polar	[67,87,93–95]

ECs: emerging contaminants; CAS No.: Chemical Abstracts Service Number; pKa: acid–base dissociation constant; Log K_{ow}: water partition coefficient; S^w: water solubility (mg/L).

4. Removal Performance of Different Constructed Wetland Configurations

The removal performance of constructed wetlands varies substantially with hydraulic configuration, substrate characteristics, vegetation, and operating conditions. Vertical-flow constructed wetlands generally provide enhanced oxygen transfer and nitrification due to intermittent loading and unsaturated conditions, facilitating the aerobic degradation of emerging contaminants [4]. Horizontal-flow constructed wetlands predominantly support anaerobic and anoxic processes, including reduction and sorption mechanisms. Hybrid constructed wetlands, integrating vertical- and horizontal-flow units, can provide complementary aerobic and anaerobic conditions and consequently improve overall contaminant removal [4,21,23]. Substrate composition, hydraulic retention time, plant species, redox conditions, and microbial activity further regulate removal efficiency. Thus, configuration-specific optimization is essential for achieving effective and sustainable EC removal. Wetlands, sorption, adsorption, transformation, degradation, and mineralisation represent distinct but interconnected processes governing the attenuation and removal of emerging contaminants (Table 2). They should be clearly differentiated because the disappearance of an EC from the aqueous phase does not necessarily indicate its destruction.

Table 2. Removal processes of emerging contaminants in constructed wetlands.

Process	Definition	Role in EC Removal in CWs
Sorption	The overall partitioning of an EC from the aqueous phase to or into a solid phase, including adsorption and absorption.	Retains ECs within substrates, sediments, biofilms, and plant-derived organic matter, reducing their aqueous concentration and increasing their residence time for subsequent transformation.
Adsorption	Accumulation of an EC primarily at the surface of a solid material through physical or chemical interactions.	Facilitates the retention of ECs on sand, gravel, biochar, zeolite, organic matter, and biofilms. It is particularly relevant for hydrophobic and ionizable compounds.
Absorption	Partitioning of an EC from the aqueous phase into the internal matrix of another material rather than merely onto its surface.	Contributes to EC retention within organic matter, plant tissues, biofilms, and organic-rich substrates and can influence subsequent release or transformation.
Plant uptake	Transfer of dissolved or particle-associated ECs from wastewater into plant tissues through roots, and in some cases, subsequent translocation to shoots.	Removes or temporarily retains ECs within plant biomass and may facilitate phytotransformation, although uptake does not necessarily indicate complete degradation.
Biotransformation	Biochemical alteration of the molecular structure of an EC by microorganisms or plants, producing one or more transformation products (TPs).	Microbial and plant-mediated reactions convert parent ECs into structurally modified products and constitute an important pathway for EC attenuation in CWs.
Abiotic transformation	Chemical alteration of ECs without direct biological mediation, including photolysis, hydrolysis, oxidation, and reduction.	Contributes to EC transformation, particularly in zones exposed to sunlight or possessing suitable pH, redox, and chemical conditions.
Biodegradation	Biological breakdown of ECs or their transformation products by microorganisms into simpler compounds through metabolic or co-metabolic processes.	Microbial communities associated with substrates, roots, and biofilms can substantially reduce EC concentrations under aerobic, anoxic, or anaerobic conditions.

Table 2. *Cont.*

Process	Definition	Role in EC Removal in CWs
Photodegradation	Transformation of ECs induced directly by light or indirectly through reactive species generated by light-driven processes.	Can contribute to the attenuation of light-sensitive ECs, particularly in shallow, surface-flow, and exposed wetland zones.
Phytotransformation	Enzymatic conversion of ECs within plants or in the rhizosphere into structurally modified metabolites or transformation products.	Complement microbial processes and can contribute to EC attenuation through plant-associated enzymatic reactions.
Volatilisation	Transfer of a contaminant from the aqueous or solid phase to the atmosphere as a volatile compound.	Generally, a secondary pathway for most ECs because many pharmaceuticals and personal-care products have low volatility; however, it may be relevant for selected volatile compounds.
Transformation-product formation	Generation of secondary compounds following the structural modification of a parent EC.	Determines the subsequent environmental fate of ECs; transformation products may be less persistent, similarly persistent, more mobile, or potentially more toxic than their parent compounds.
Degradation	Breakdown of an EC or its transformation products into compounds of progressively lower molecular complexity.	Represents a greater degree of contaminant destruction than simple phase transfer and can occur through microbial, enzymatic, photochemical, or chemical processes.
Mineralisation	Further conversion of organic ECs and/or their transformation products into inorganic end products such as CO_2, H_2O, inorganic nitrogen, phosphate, and halide ions, depending on molecular composition.	Represents the most complete degradation pathway and indicates substantial destruction of the original organic contaminant rather than merely its transfer or structural modification.
Desorption	Release of a previously sorbed EC from a solid or organic phase back into the aqueous phase.	Can reduce the apparent removal efficiency and represents a potential pathway for the remobilisation of retained ECs, particularly when environmental conditions change.
Retention/ sequestration	Temporary or prolonged immobilisation of ECs within wetland substrates, sediments, biofilms, or plant biomass.	Reduces aqueous-phase concentrations and may protect against immediate downstream transport, although long-term retention does not necessarily mean permanent removal.

Based on the available evidence, the contribution of key removal processes to the attenuation of emerging contaminants in constructed wetlands can be attributed to multiple interacting mechanisms, including photodegradation, biodegradation, sorption, and phytoremediation [4,21,23]. These processes do not necessarily operate independently or contribute equally; rather, two or more mechanisms may simultaneously participate in the removal of individual ECs. The relative contribution of each mechanism cannot be generalized across CW systems, as it varies with wetland configuration, contaminant characteristics, and operational and environmental conditions [46–48,96]. Therefore, the removal of ECs in CWs should be considered a multifactorial process involving the combined effects of physical, chemical, and biological pathways, with their relative importance being compound- and system-dependent. Several interacting factors influence the relative contribution of individual removal processes, including substrate characteristics, vegetation species, and the physicochemical properties of the target ECs [49–53,97]. The effectiveness and significance of specific removal mechanisms may vary considerably among different CW configurations

and operating conditions [54,55]. Photodegradation may contribute substantially to the removal of readily photodegradable pharmaceuticals, such as naproxen and diclofenac, whereas biodegradation, sorption, and phytoremediation can collectively contribute to EC removal in subsurface-flow constructed wetlands (SSF-CWs) [56–60]. Therefore, the removal of EC in CWs should be regarded as a multifactorial process, in which the relative importance of individual mechanisms depends on contaminant characteristics and wetland-specific conditions [61,62].

4.1. Removal Performance of ECs in CWs

Removal efficiency is an important criterion for evaluating the performance of constructed wetlands (CWs) at the macroscopic level. This review focuses on frequently occurring ECs and examines their removal and fate within CW systems. These contaminants originate from diverse wastewater sources, including domestic, industrial, and hospital effluents [3,63,64]. The physicochemical properties of commonly investigated ECs are summarized in Table 1. Based on hydrological flow patterns, CWs used for EC removal can be broadly classified into three major types: vertical-flow constructed wetlands (VFCWs), horizontal-flow constructed wetlands (HFCWs), and hybrid constructed wetlands (HCWs) [35,38,65] (Figure 2I,II). In HFCWs, wastewater enters the system at the inlet and flows horizontally through the porous substrate beneath the filter-bed surface before reaching the outlet [26]. In VFCWs, wastewater is intermittently applied and percolates vertically through the substrate layers before being collected at the bottom [28–30]. Hybrid CWs integrate two or more configurations, such as HFCW–VFCW or VFCW–HFCW, arranged sequentially to enhance treatment performance. This integrated configuration exploits the complementary treatment mechanisms of different wetland types, thereby improving overall pollutant removal efficiency [32,35,67,68]. CWs are increasingly recognized as effective nature-based solutions for removing ECs, including pharmaceuticals, personal care products, pesticides, and industrial chemicals, from water and wastewater [69,70]. Their removal performance results from the combined action of physical, chemical, and biological processes, such as filtration, sedimentation, sorption, microbial degradation, plant uptake, and photodegradation [35,36,71–73,75] (Table 2). Pharmaceuticals like diclofenac and ibuprofen are removed efficiently in CWs through microbial degradation and adsorption onto substrate materials. The rhizosphere zone plays a pivotal role, as it enhances oxygen transfer and provides a habitat for microbial communities capable of degrading complex organic compounds [32,72,75,76]. Personal care products such as triclosan and bisphenol A are primarily removed through sorption onto organic matter and biodegradation pathways. A key mechanism for EC removal is microbial metabolism, wherein bacteria enzymatically break down contaminants into less harmful by-products [3,73]. Sorption onto substrate media like gravel, sand and biochar aids in immobilizing hydrophobic compounds, while plant uptake removes a fraction of ECs, particularly those with high water solubility. Photodegradation of light-sensitive ECs, facilitated by UV radiation, adds another layer of contaminant reduction in surface-flow CWs [4,11,74].

Figure 4 shows the average removal efficiency of the selected ECs in WWTPs and CWs. The results demonstrate that CW systems are capable of removing a wide range of ECs from wastewater. It was observed that CWs exhibit considerable variability in their treatment performance for frequently studied ECs, including pharmaceuticals, personal care products, endocrine-disrupting chemicals, and others. CWs demonstrated a high removal potential for acetaminophen, paracetamol, salicylic acid, sulfadimethoxine, and caffeine, with average removal efficiencies exceeding 85%. The average removal efficiencies of tramadol, gemfibrozil, and iopromide in CW systems were relatively low, i.e., below 20%. The average removal efficiencies of ECs, including acetaminophen, ibuprofen, naproxen, sulfadimethox-

ine, atenolol, and caffeine, exceeded 65% in CW treatment systems, demonstrating their considerable potential for effective EC removal (Figure 4).

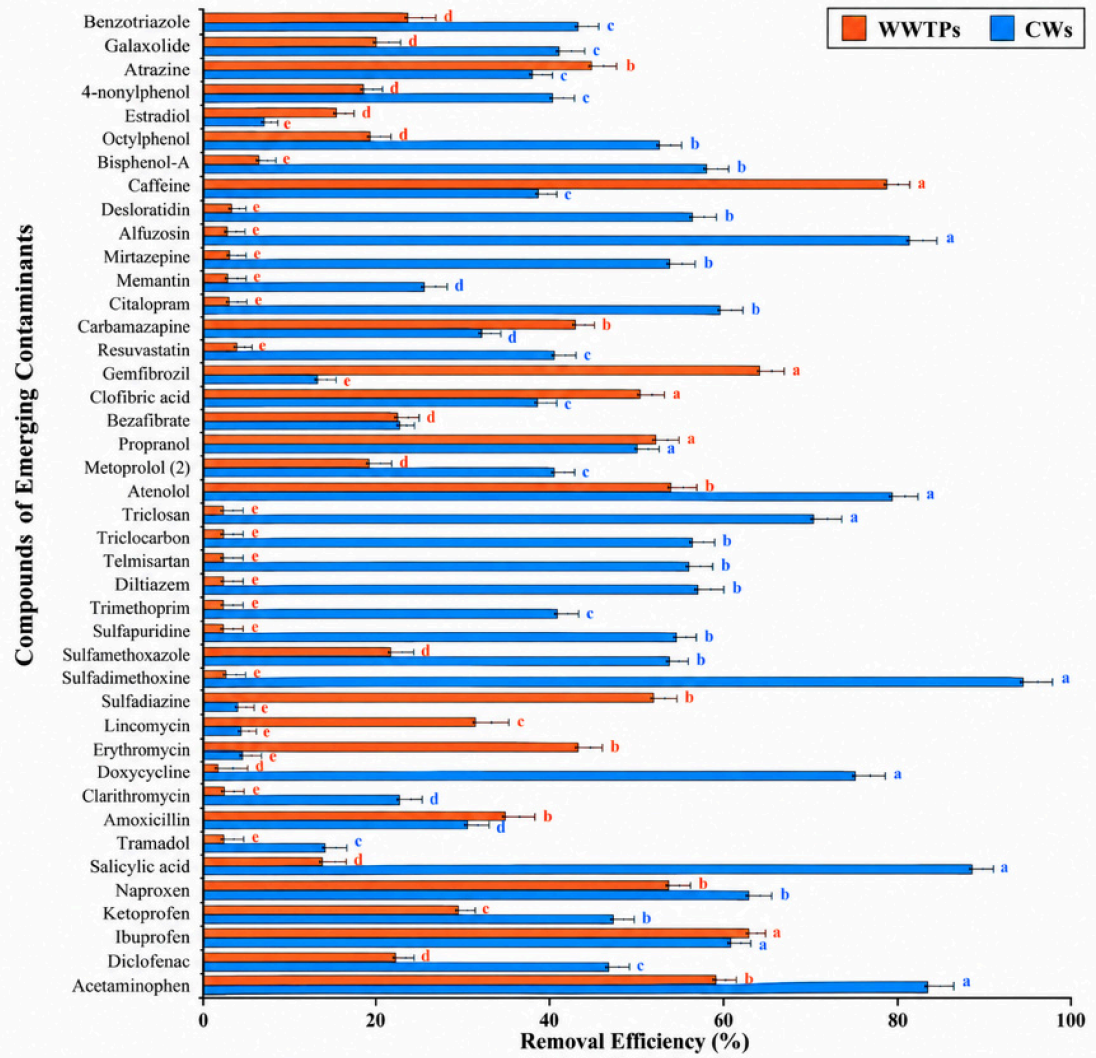


Figure 4. Removal efficiency of selected emerging contaminants in wastewater treatment plants (WWTPs) and constructed wetlands. The removal data for emerging contaminants in CWs were collected from multiple published studies, as follows [3–6,9,11,18–21,23,25,26,35,41,43,47–54,67,73,81–83,85], and Different lowercase letters (a, b, c, d, and e) indicate statistically significant differences in removal efficiencies among treatment systems, while identical letters indicate no significant difference ($p < 0.05$).

Comparison of emerging contaminant removal efficiencies (%) in conventional wastewater treatment plants (WWTPs) and constructed wetlands (CWs). Orange and blue bars represent WWTPs and CWs, respectively. Different lowercase letters (a, b, c, d, and e) indicate statistically significant differences in removal efficiencies among treatment systems, while identical letters indicate no significant difference ($p < 0.05$).

Several studies have reported that constructed wetlands can achieve higher removal efficiencies for emerging contaminants comparable to those of conventional wastewater treatment plants (WWTPs) [4,26,55,75]. Figure 4 and Table S1 shows that CWs exhibited higher reported removal efficiencies than WWTPs for several emerging contaminants (ECs), including benzotriazole, galaxolide, 4-nonylphenol, octylphenol, metoprolol, bisphenol A, desloratadine, alfuzosin, mirtazapine, memantine, citalopram, rosuvastatin, atenolol, triclosan, triclocarban, telmisartan, diltiazem, trimethoprim, sulfamethoxazole, sulfadimethox-

ine, doxycycline, salicylic acid, naproxen, ketoprofen, and diclofenac [21,23,25,26,35,41]. WWTPs demonstrated higher reported removal efficiencies for several compounds, including caffeine, carbamazepine, gemfibrozil, clofibric acid, bezafibrate, propranolol, ibuprofen, amoxicillin, erythromycin, lincomycin, and acetaminophen [5,9,11,18]. These differences suggest that EC removal is compound-specific and influenced by the physicochemical properties of the contaminants, their biodegradability, sorption potential, and susceptibility to degradation, as well as treatment-system characteristics. However, the observed differences should be interpreted cautiously because the figure compares the reported percentage removal efficiencies across studies that may differ in influent and effluent concentrations, mass loading rates, hydraulic retention times (HRTs), treatment configurations, substrate composition, microbial communities, and operational conditions [76–78]. The higher removal efficiency reported for a particular compound in CWs or WWTPs does not necessarily indicate the inherent superiority of one treatment technology over the other under comparable operating conditions [47,48,79]. Therefore, the higher percentage removal efficiencies reported for CWs should not be considered conclusive evidence of their superior treatment performance relative to WWTPs. This is because removal efficiency is influenced not only by treatment technology, but also by initial contaminant concentrations, loading rates, and operational conditions. Differences in influent concentrations can substantially affect the calculated removal percentages, as the same absolute reduction in contaminant concentration may yield different percentage removal values depending on the initial concentration [81–83]. Comparison of treatment performance requires the consideration of influent and effluent concentrations, contaminant mass removal, and relevant operational parameters under comparable treatment conditions. CWs have demonstrated effective treatment performance for real-world wastewater matrices, particularly municipal wastewater generated from small villages and communities, as well as landfill leachate, highlighting their potential as decentralized and nature-based treatment systems [49,50,84]. CW systems have been increasingly adopted as decentralized wastewater treatment technologies in several developed countries, demonstrating considerable cost-effectiveness for the removal of ECs from wastewater generated by rural settlements and local communities [50,85]. Landfill leachate represents an important source of diverse ECs, including pharmaceuticals and personal care products (PCPs), endocrine-disrupting chemicals (EDCs), artificial sweeteners (AWSs), bisphenol A, and pesticides [4,21,46]. Previous studies have demonstrated that full-scale hybrid constructed wetlands (CWs), incorporating multiple treatment units such as aerobic lagoons, reed beds, and polishing ponds, achieved EC removal efficiencies of approximately 85–95% or higher from landfill leachate [51,52,86]. These systems showed superior treatment performance compared with conventional wastewater treatment plants (WWTPs) in Singapore [21].

Hybrid CWs have been widely utilized as a secondary treatment step for EC removal [48,56]. Hybrid CWs could offer spatial zone environments, providing diverse redox conditions for microbial growth. Aerobic environments were dominant in VFCWs, while an anaerobic environment was favourable in HFCWs [56]. The integration of VFCWs with HFCWs can be beneficial for EC removal due to the diverse redox environments within a hybrid CW bed [56]. According to the removal characteristics of different kinds of ECs, various hybrid systems could be integrated to enhance the removal efficiencies of multiple ECs in CWs [57]. In this study, one hybrid CW system consisting of ponds, FWSCWs and HFCWs in series or VFCWs-HFCWs showed higher removal performance or efficiency than conventional WWTPs in the removal of ECs, i.e., ketoprofen (81.25%), diclofenac (87.47%), and caffeine (96.64%) [55]. Hybrid constructed wetlands generally achieve higher removal efficiencies for emerging contaminants than HFCWs or VFCWs because they combine complementary treatment mechanisms and redox conditions [20].

The VFCW stage provides better oxygen transfer and promotes aerobic biodegradation, filtration, and sorption, whereas the HFCW stage provides prolonged contact with saturated media and favourable anoxic/anaerobic conditions for further biodegradation and sorption [21,22,53]. The sequential configuration also enhances hydraulic contact time, microbial diversity, rhizosphere interactions, and contaminant–substrate interactions. Thus, the integration of aerobic, anoxic, and anaerobic processes enables more effective removal of chemically diverse ECs than a single wetland configuration [20–22,27,52,57,67,83,87].

CW treatment systems can serve as an effective tertiary treatment step for further improving the quality of municipal wastewater effluents. Conventional wastewater treatment plants are generally not specifically designed to effectively remove a broad range of ECs, resulting in their persistence in treated effluents and subsequent discharge into receiving aquatic environments [55,57,88]. Therefore, the incorporation of an additional tertiary treatment step is essential to further reduce residual ECs and minimize their potential environmental impacts [57]. Cost-effective and sustainable CW technologies offer a promising option for polishing secondary WWTP effluents [56]. Among the various CW configurations, free-water-surface constructed wetlands (FWS-CWs), VFCWs, HFCWs, and hybrid CW systems are among the most commonly applied approaches for the tertiary treatment and removal of ECs from municipal wastewater [58]. This might be ascribed to the larger storage capacity and natural landscape enhancing effects of FSCWSs compared to SSFCWs [46,56]. A recent full-scale study that adopted a reclamation pond-constructed wetland (CW) system for the tertiary treatment of practical secondary effluent from WWTPs demonstrated a good average removal efficiency of 72–75% for several ECs (e.g., acetaminophen, ibuprofen, ketoprofen, diclofenac, naproxen, and triclosan) [58]. The CW systems were as efficient in the removal of ECs as other tertiary treatment technologies, including photo-Fenton and membrane filtration, advanced oxidation, etc., in SSFCWs [46,56]. Despite the effective removal of many emerging contaminants, negative removal efficiencies have occasionally been reported for compounds such as carbamazepine, paracetamol, ibuprofen, and sulfamethoxazole. Negative removal, characterized by a higher measured concentration in the effluent than in the influent, should not necessarily be interpreted as contaminant generation within the CW [20,52,89]. It may result from a combination of analytical and sampling variability, temporal and spatial concentration heterogeneity, desorption of previously retained compounds from wetland substrates or biomass, and transformation processes [55]. Deconjugation of EC metabolites and the subsequent regeneration of parent compounds can increase the apparent concentration of target ECs in the effluent. Transformation or back-conversion of precursor compounds may contribute to apparent negative removal [55]. Changes in pH, redox conditions, dissolved organic matter, and substrate saturation may further influence sorption–desorption equilibria and contaminant release [20,55]. Therefore, negative removal should be interpreted cautiously, and where possible, evaluated using mass balances, paired sampling, replicate analyses, and simultaneous monitoring of relevant metabolites and transformation products. CW systems demonstrate considerable potential as sustainable treatment technologies for the removal of a wide range of ECs, however, their treatment performance varies substantially among individual ECs.

4.2. Mechanisms of EC Removal in CWs

The removal of ECs in CWs is governed by a complex interplay of physical, chemical, and biological processes, with their relative contributions strongly influenced by wetland configuration, design characteristics, and operational conditions [3,27]. Within wastewater treatment systems, EC removal can occur through several pathways, including physicochemical transformation, photodegradation (photolysis), microbial biodegradation, and sorption onto wetland substrates and plant-associated organic matter [35]. These processes

may act synergistically to facilitate the transformation, retention, and subsequent removal of ECs from the wastewater matrix. The removal mechanisms generally depend on the physicochemical properties of ECs [15,35,46,90].

These processes are closely interrelated and collectively contribute to the removal of ECs in CWs, primarily through solar photodegradation, substrate sorption/adsorption, sedimentation, microbial biodegradation, and plant-mediated phytoremediation [21,46]. A comprehensive understanding of individual removal processes and their underlying mechanisms is therefore essential for optimising EC removal efficiency in CW systems [21,55]. The mechanisms involved in EC removal in CWs can be classified into biotic processes (e.g., microbiological degradation, biofilm, root and plant uptake) and physicochemical/abiotic processes (evaporation, photodegradation, oxidation, hydrolysis, retention or root sorption into the gravel/sand bed) (Figure 5A,B). CWs are highly complex systems with multiple environments and microenvironments, where different physicochemical conditions can prevail, influencing the removal processes mentioned above [20,21,48,57]. The internal environment of CWs varies significantly depending on their specific design, encompassing distinct zones such as pore water, the sunlit surface layer, macrophytes, biofilms on the substrate, and root-associated biofilms. This structural and functional heterogeneity contributes to the current limited understanding of the complex mechanisms governing contaminant removal in CWs [19–21] (Table 3).

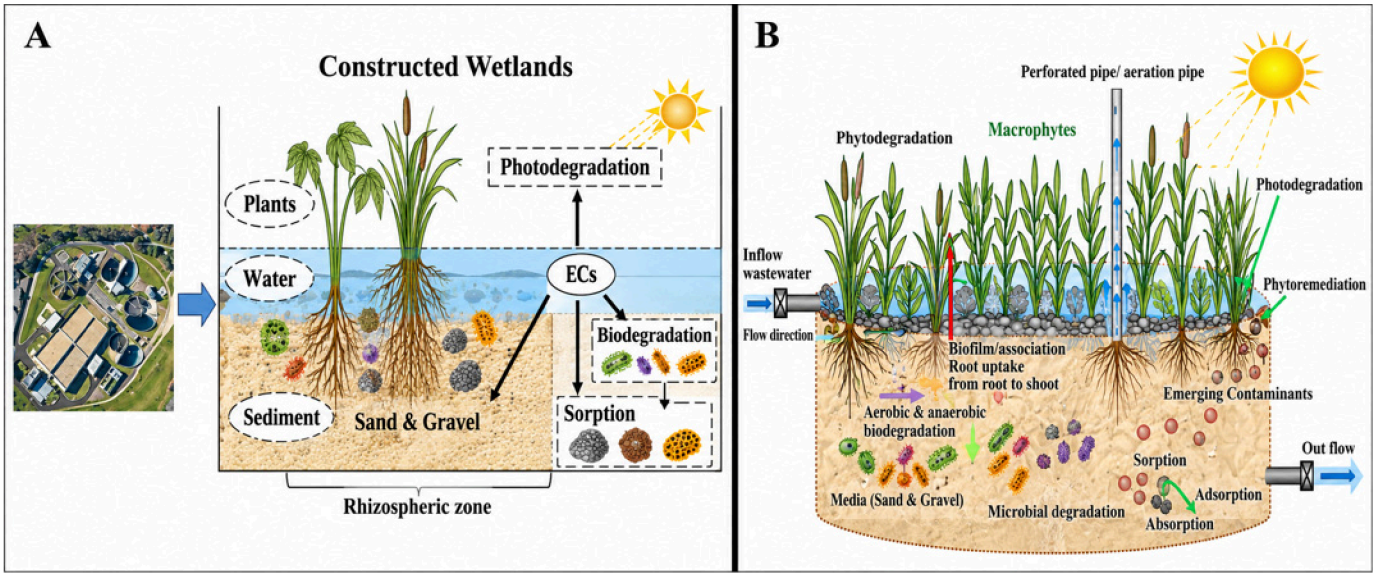


Figure 5. Schematic representation of CWs for the removal of ECs: (**A**) major components and principal removal mechanisms, and (**B**) detailed pathways of EC removal through CWs.

Table 3. Removal mechanisms of selected emerging contaminants in constructed wetlands.

Sl. No.	Emerging Contaminants	Dominant Removal Process	References
1	Caffeine	Biodegradation Microbial remediation Phytoremediation	[10,17,26,32]
2	Carbamazepine	Sorption Phytoremediation Photodegradation	[22,26,59,60,66]

Table 3. *Cont.*

Sl. No.	Emerging Contaminants	Dominant Removal Process	References
3	Erythromycin	Biodegradation Sorption Phytoremediation	[4,26,59,60,65]
4	Clarithromycin	Sorption Photodegradation	[21,22,57,67,71,75]
4	Bisphenol-A	Adsorption Biodegradation Photodegradation	[61,67,82,86,92]
5	Iopromide Norephedrine Azathioprine	Adsorption Biodegradation Microbial degradation	[62,87,93,94,96,97]
6	Benzophenone-3 Benzophenone-4	Photodegradation Phytoremediation Biodegradation	
7	Methylparaben Ethylparaben Propylparaben Butylparaben	Photodegradation Biodegradation	
8	Diclofenac	Biodegradation Photodegradation	[20,21,63,67,81,87,91]
9	Ibuprofen	Biodegradation Photodegradation	[23,41,44,64,67,83]
10	Ketoprofen	Photodegradation	[21,26,64,67,81,87]
11	Naproxen	Biodegradation Photodegradation	[21,26,28,64,67,87,93]
12	Acetaminophen	Biodegradation Phytoremediation	[20–22,57,62,63,67,75]
13	Triclosan	Biodegradation Sorption Photodegradation	[20–22,44,65,67,87,91]
14	Clarithromycin	Sorption Photodegradation Phytoremediation	[21,22,26,65,67,91,94,97]
15	Erythromycin	Biodegradation Sorption Microbial degradation	[21,22,56,64,66,67,83,96]
16	Lincomycin	Biodegradation Phytoremediation	[20,21,44,56,67,83,91,94,97]
17	Sulfamethoxazole	Biodegradation Photodegradation	[25,26,56,57,67,83,86,91]
18	Atenolol	Sorption Phytoremediation	[21,56,59,63,65,67,91]
19	Naproxen	Biodegradation Phytoremediation	[4,51,64,67,71,75,83]

Table 3. *Cont.*

Sl. No.	Emerging Contaminants	Dominant Removal Process	References
20	Ofloxacin	Adsorption Biodegradation Phytoremediation	[57,66,67,73,79,83,87]
21	Roxithromycin		
22	Oestrone, 17 beta-oestradiol, and 17 alpha- alpha-ethynylestradiol	Biodegradation Phytoremediation	[15,21,22,52,63,65,84,86,91]
23	Chlorophene	Adsorption Biodegradation Phytoremediation	[21,48,52,56,67,73,77,83,85,89]
24	Ciprofloxacin	Biodegradation Sorption Photodegradation Phytoremediation	[20,44,66,67,84,86,87,92]
25	Amoxicillin		
26	Chlorpyrifos	Adsorption Biodegradation Phytoremediation Photodegradation	[21,26,59,66,67,71,73,78,82,91,95]
27	Malathion		

The removal of ECs in constructed wetlands is governed by multiple, contaminant-specific mechanisms rather than a single universal pathway. As shown in Figure 6 and Table S2, the relative contribution of phytoremediation, biodegradation, sorption, and photodegradation varies substantially among compounds according to their physicochemical properties, biodegradability, hydrophobicity, ionization behaviour, and interactions with wetland plants and substrates. Phytoremediation was particularly important for compounds such as ibuprofen (~50%), clofibric acid (~37%), bisphenol-A (~29%), caffeine (~20%), and codeine (~14%). Plant-associated removal may involve direct uptake, rhizosphere interactions, enzymatic transformation, and modification of the surrounding microbial environment. The relatively greater contribution observed for ibuprofen and bisphenol-A indicates that plant–microbe interactions can complement conventional physicochemical removal processes [21–23]. Biodegradation represented an important pathway for several pharmaceuticals, including bezafibrate (~83%), sulfamethazine (~62%), caffeine (~57%), naproxen (~56%), ofloxacin (~67%), lincomycin (~43%), bisphenol-A (~42%), and triclocarban (~41%). These patterns highlight the importance of microbial communities associated with wetland substrates and plant rhizospheres. Aerobic, anoxic, and anaerobic microsites within CWs can support different microbial transformation pathways [21,23,37,67]. Sorption was particularly prominent for relatively persistent or sorptive compounds such as oxytetracycline (~88%), atenolol (~60%), erythromycin (~32%), clofibric acid-related systems, ofloxacin (~20%), triclosan (~18%), and triclocarban (~16%). Sorption occurs through interactions with organic matter, biofilms, sand, gravel, and other substrate components [21,91,92]. However, sorption should be interpreted primarily as retention or immobilization rather than complete contaminant destruction, and subsequent desorption or transformation may occur [21,22]. Photodegradation contributed substantially to the removal of compounds such as diclofenac (~74%), imipramine (~38%), acetaminophen (~32%), naproxen (~26%), and sulfamethazine (~20%) (Figure 6). This pathway can be enhanced in wetland systems where contaminants are exposed to sunlight, particularly in shallow or free-water-surface zones. Its importance is therefore strongly influenced by

wetland configuration, water depth, light penetration, season, and contaminant photoreactivity [21–23]. Figure 6 demonstrates that EC removal in CWs is compound-specific and results from the interaction of several concurrent processes. For example, diclofenac is strongly associated with photochemical transformation, ibuprofen with phytoremediation and biodegradation, oxytetracycline predominantly with sorption, and bezafibrate primarily with biodegradation. Such differences explain why the removal efficiency of CWs can vary considerably among ECs even under similar operating conditions.

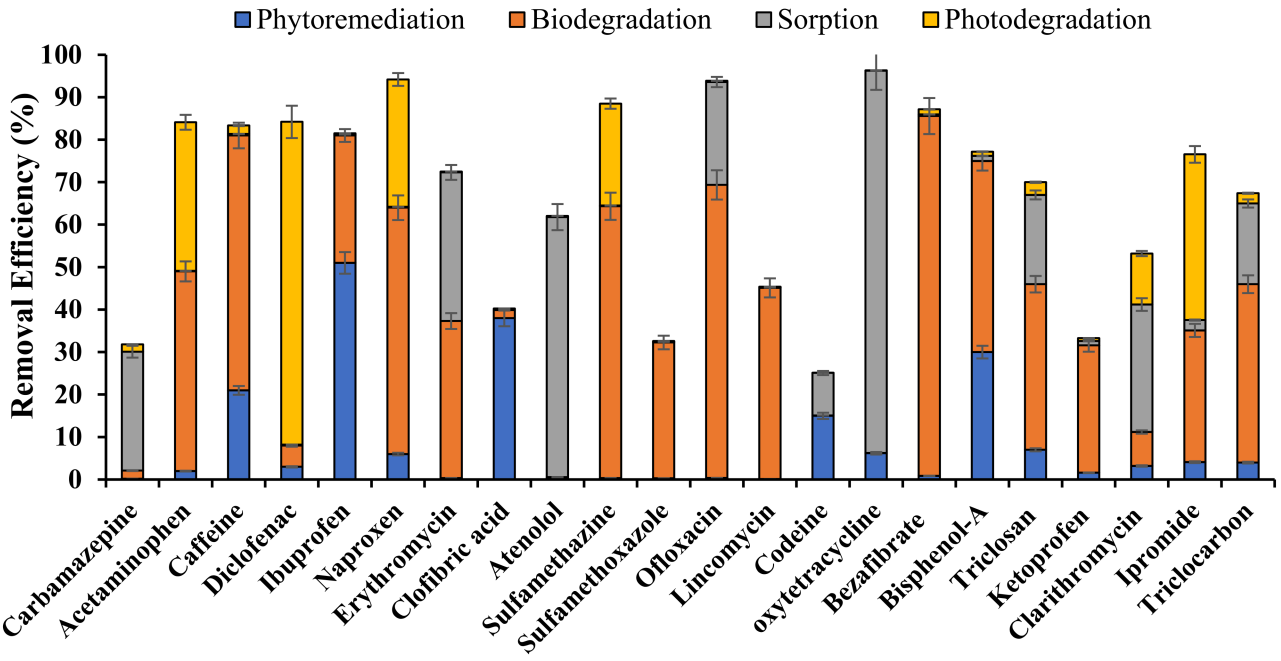


Figure 6. Compound-specific contribution of major EC removal mechanisms, i.e., phytoremediation, biodegradation, sorption, and photodegradation in constructed wetlands.

4.2.1. Sorption

Sorption is an abiotic process that naturally occurs due to either electrostatic interactions between compounds and substrate particles, referred to as adsorption, or through hydrophobic interactions between the lipophilic cell membrane of a microorganism or the lipid fraction of a solid, referred to as absorption [3] (Figure 7). ECs with diverse chemical structures and physicochemical properties exhibit distinct interactions with wetland substrates, resulting in variable sorption affinities and rates. The prediction and modelling of sorption behaviour are further influenced by environmental conditions within CWs, as the functional groups of individual ECs can interact differently with substrate media and undergo complex, pH-dependent speciation [3,41,47]. Sorption processes are commonly evaluated using linear, Langmuir, and Freundlich isotherm models to characterize the equilibrium interactions between ECs and wetland substrates [20,21,93]. The models yield sorption coefficients that help assess sorption capacity and efficiency, enabling comparisons across different compounds. The Langmuir model provides the Langmuir constant (KL) and the substrate's maximum adsorption capacity (QM) [41], while the Freundlich model generates the Freundlich constant (Kf) and the exponent (n) [41,56,68]. These coefficients facilitate the comparison of adsorption performance across various experiments and substrates. Since sorption and desorption influence contaminant mobility in wetlands, understanding the physicochemical factors that govern sorption is essential for modelling long-term water quality [41,68]. Sorption processes are governed by the

physicochemical conditions prevailing within the wetland, the intrinsic physicochemical properties of the target contaminants, and the physicochemical characteristics of the wetland substrate [44,68,69].

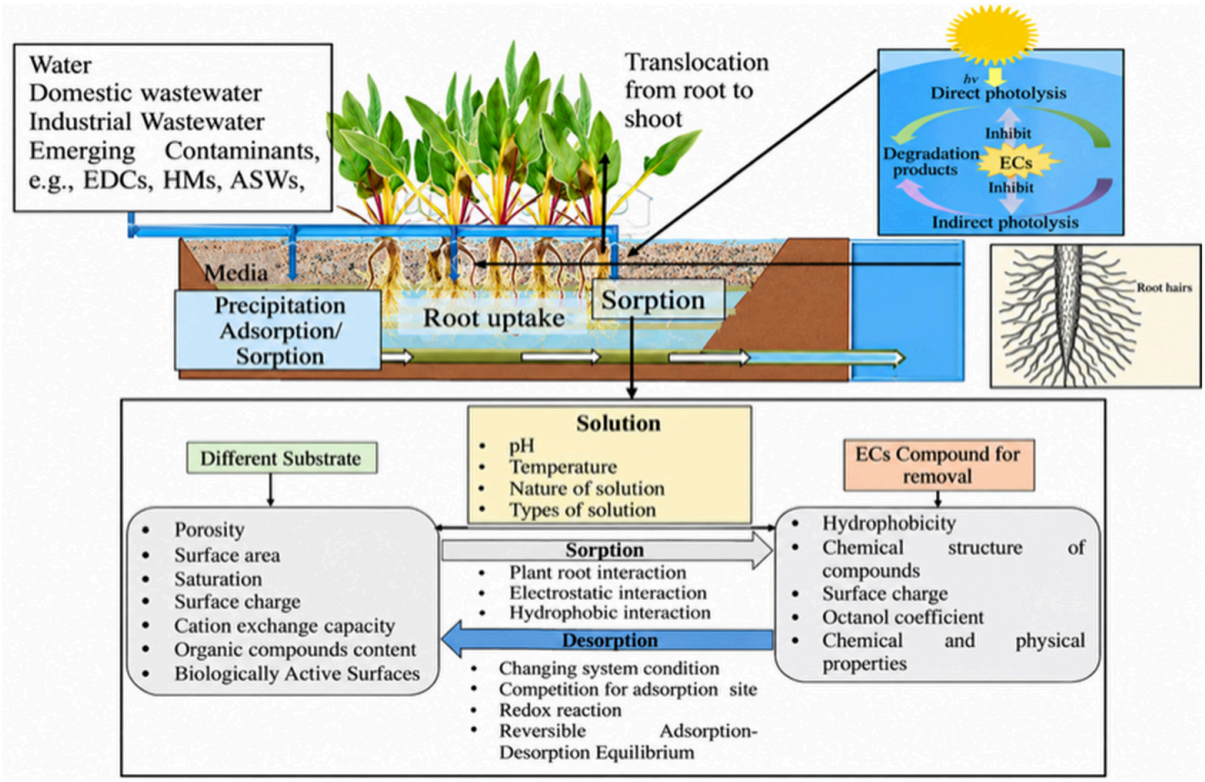


Figure 7. Schematic representation of sorption and photodegradation processes involved in the removal of emerging contaminants in constructed wetlands.

4.2.2. Substrate Sorption

CW substrates support plant root anchorage and offer a large surface area for microbial colonization [67]. These substrates can absorb ECs from wastewater and may include natural materials, agricultural or industrial by-products, and synthetic components [20,21,66,67]. EC removal in constructed wetlands is strongly influenced by contaminant physicochemical properties and wetland conditions. LogKow determines hydrophobicity and therefore the tendency of ECs to undergo sorption onto substrates and organic matter. pKa controls ionization, which affects sorption, mobility, and biodegradation [66]. Solubility influences contaminant transport and contact with microbial communities. Hydrophobic and less soluble compounds generally show greater sorption, whereas polar and highly soluble ECs are more likely to remain in the aqueous phase and undergo biodegradation or transformation [67,94]. Wetland configuration, redox conditions, vegetation, microbial activity, light exposure, and hydraulic retention further regulate biodegradation and photodegradation pathways. Thus, EC removal results from the combined effects of contaminant properties and wetland-specific environmental conditions [21,22,67].

The adsorption of ECs onto CW substrates involves various mechanisms such as hydrophobic interactions, van der Waals forces, electrostatic interactions, and ion exchange [44,67] (Figure 8). While polar or ionic ECs are more likely to adsorb to mineral-based substrates, non-polar ECs preferentially bind to substrates rich in organic matter [56]. The sorption behaviour is influenced not only by the physicochemical properties of the ECs such as their acid dissociation constant (pKa), octanol–water partition coefficient (Log

Kow), and linear sorption coefficient (Log Kd), but also by characteristics of the substrate itself, including composition, natural organic matter (NOM) content, and pH [63].

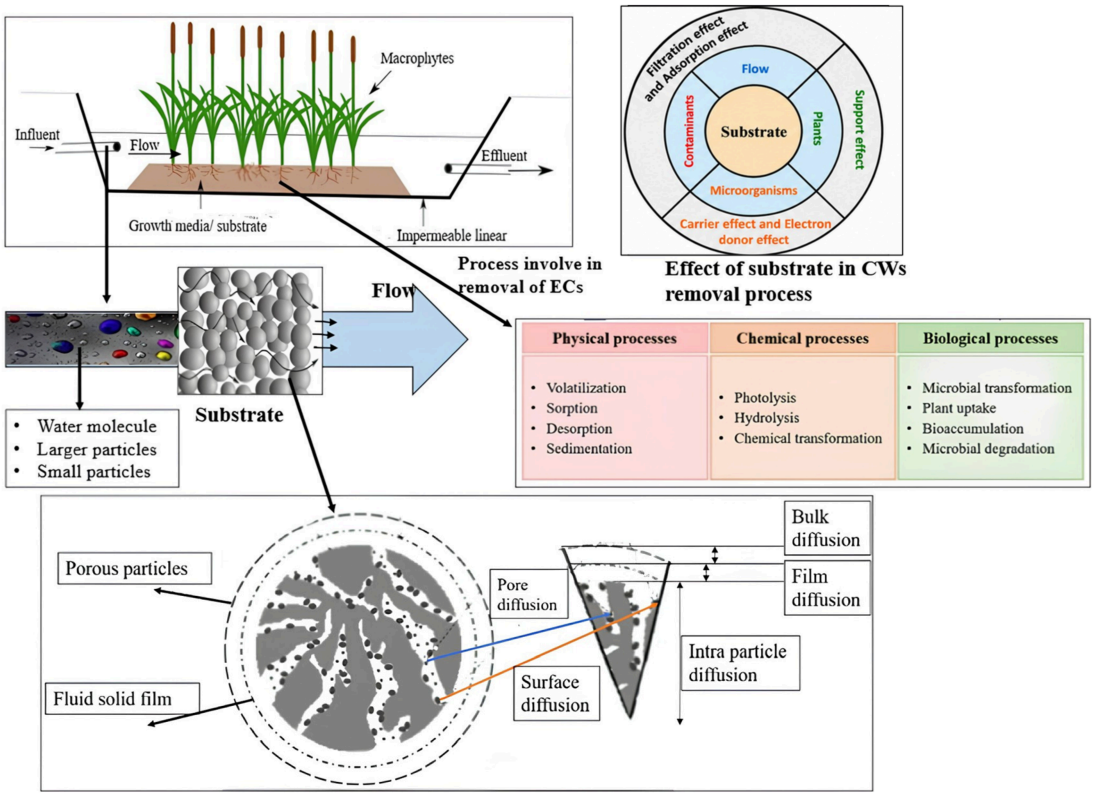


Figure 8. Schematic representation of key physical, chemical, and biological processes involved in emerging contaminant removal in constructed wetlands.

The sorption of ECs onto CW substrates is strongly influenced by their octanol–water partition coefficient (Log Kow) [63,69]. As a general rule, ECs with Log Kow values below 2.5 exhibit low sorption potential, those ranging from 2.5 to 4.0 show moderate sorption, and compounds with Log Kow values above 4.0 tend to exhibit strong sorption behaviour [3,41,69]. A study by [70] observed a notable difference in the removal of ibuprofen and caffeine in CWs: ibuprofen, with a Log Kow of 3.97, demonstrated a higher affinity for sorption onto CW substrates, whereas caffeine, with a Log Kow of just 0.01, showed minimal sorption. In sorption studies, Kd is commonly used to describe how chemicals are distributed between the substrate and water phases [41,68]. According to Ternes and Joss [71], the sorption potential of contaminants can be broadly assessed based on their distribution coefficient (Kd). Compounds with Kd values > 500 L/kg (log Kd > 2.67) are generally considered to exhibit strong sorption, whereas those with values below these thresholds are characterized by relatively weak sorption. The organic carbon–water partition coefficient (Koc) represents the affinity of a compound for organic carbon and is defined as the amount of compound sorbed per unit mass of organic carbon relative to its equilibrium concentration in the aqueous phase [44,68]. Several studies have demonstrated that both Koc and Kd can be used to predict the sorption characteristics of ECs [41,72]. Lv et al. [72] reported that imazalil concentrations in constructed wetland substrates were two to ten times higher than those of tebuconazole across both warm and cold seasons. This discrepancy may be attributed to two factors: (1) imazalil has a higher Koc compared to tebuconazole (4753 vs. 2579 L/kg), and (2) imazalil also exhibits a higher Log Kow (4.10 vs. 3.89) [72]. The sorption behaviour of ECs onto substrates depends not only on their chemical properties (such as Kow, Koc, and Kd), but also on factors like the

organic carbon content of the substrate, the species of plants used in the CWs, and seasonal variations [72]. However, research has shown that substrate sorption plays a relatively minor role in EC removal. For instance, only 7% of imazalil and tebuconazole were removed via substrate sorption in mesocosm CWs, with plant uptake and biodegradation being the dominant removal mechanisms [67,72]. Chen et al. [56] found that sorption had limited impact on removing bromochloroacetonitrile and dibromoacetonitrile in laboratory-scale CWs. Common CW substrates contribute minimally to EC removal, emphasizing the importance of biodegradation and plant uptake. Therefore, future efforts should focus on developing new substrate materials that are cost-effective, durable, and exhibit high sorption capacity for ECs [67–69]. The co-occurrence of multiple emerging contaminants (ECs) may induce competitive sorption, thereby reduce the availability of sorption sites and potentially limit their overall removal in constructed wetlands [44]. For instance, incomplete removal of ibuprofen, iohexol, tebuconazole, and imazalil was reported during a 66-day CW operation, which was attributed to competition among these contaminants for available sorption sites [62,63].

4.2.3. Photodegradation

Photodegradation can contribute to the attenuation of certain emerging contaminants in constructed wetlands through direct and indirect photolysis [7]. In direct photolysis, EC molecules absorb solar radiation directly, resulting in excitation and subsequent chemical bond cleavage. The efficiency of this pathway depends largely on the molecular structure and light-absorbing properties of the contaminant [7,63,73] (Figure 7). Indirect photolysis occurs when sunlight generates reactive species, such as hydroxyl radicals (•OH), singlet oxygen ($^1O_2$), and excited triplet-state dissolved organic matter (^{3}DOM*), which subsequently react with EC molecules. The available evidence indicates that different ECs may undergo different photolytic pathways even within the same treatment system [73]. For example, in an open-water treatment wetland, sulfamethoxazole and propranolol primarily underwent direct photolysis, whereas carbamazepine and atenolol were mainly degraded through hydroxyl-radical-mediated indirect photolysis. Propranolol was also reported to undergo indirect photolysis mediated by ^{3}DOM*. The occurrence and efficiency of photolysis in CWs depend on solar radiation intensity, season, water depth, pH, dissolved organic matter (DOM), and coexisting ions. DOM can act as an important photosensitizer by generating reactive species that facilitate indirect photolysis; however, it may also inhibit photodegradation by competing for light or reactive species. Photolysis is more relevant in free-water-surface CWs, where contaminants are directly exposed to sunlight, whereas its contribution is generally limited in subsurface-flow CWs because of restricted light penetration and shading by emergent or floating vegetation [7,95]. For example, the photodegradation of three antiviral drugs, acyclovir, zidovudine, and lamivudine, under simulated sunlight showed that zidovudine predominantly underwent direct photolysis, while acyclovir and lamivudine were primarily degraded through indirect photolysis. Photodegradation is widely recognized as an important mechanism for the removal of emerging contaminants from aquatic environments; however, research on EC photolysis has predominantly focused on surface water systems, including lakes, rivers, and oceans [66,74]. In SSFCWs, light penetration into the substrate is generally limited [20–22,67], thereby restricting the potential contribution of photodegradation to EC removal. Although photodegradation is an important transformation pathway for emerging contaminants (ECs) in open-water systems, its contribution in CWs is likely limited, particularly in SSFCWs, because of restricted light availability and shading from wetland vegetation. Free-water-surface (FWS) CWs may provide more favourable conditions for photodegradation because the exposed water surface allows for the greater penetration of solar radiation into the water

column. The role and significance of photolytic degradation of ECs in CW systems remain insufficiently understood [66], largely because photodegradation is expected to make only a minor contribution to EC removal in SSFCWs [20–22,67].

Previous studies also indicated that the presence of emergent macrophytes and floating vegetation could block sunlight reaching the water surface of CWs, leading to situations where photodegradation might not be involved in the removal of ECs in CWs [72]. Despite significant progress in the design, configuration, and optimization of constructed wetlands (CWs), a comprehensive understanding of the photolysis mechanisms governing the degradation of emerging contaminants within CWs remains limited. It is worth mentioning that Jasper and Sedlak [75] investigated the sunlight-induced photolysis performance of four kinds of ECs (i.e., atenolol, carbamazepine, propranolol, and sulfamethoxazole, respectively) in a novel open-water unit process treatment wetland and demonstrated the different photolysis mechanisms of these ECs: sulfamethoxazole and propranolol underwent direct photolysis; carbamazepine and atenolol were mainly degraded through hydroxyl radical-induced indirect photolysis [20,21,67]. The photodegradation of propranolol was primarily attributed to indirect photolysis mediated by the excited triplet state of dissolved organic matter (^{3}DOM*). Furthermore, the study developed and applied a photochemical model to predict the photodegradation behaviour of ECs, providing insights into their transformation kinetics and highlighting the potential of CW systems for the effective attenuation of emerging contaminants [75]. Photodegradation of ECs in CWs is influenced by various environmental factors, including sunlight intensity, seasonal fluctuations, water depth, pH, and the presence of coexisting ions (Figure 7) [7,28,63,75]. Studies have shown that solar radiation and seasonal changes significantly impact the efficiency of EC removal in CWs. The removal rates of compounds such as diclofenac, triclosan, ketoprofen, and naproxen differ markedly between summer and winter [23]. These seasonal variations in EC removal efficiency are primarily associated with differences in solar irradiance and sunlight exposure, consistent with findings reported in previous studies [20,21,67]. Seasonal patterns have also been documented for other classes of emerging contaminants, including antiseptics, flame retardants, pesticides, and plasticizers [23]. Dissolved organic matter (DOM) serves as an important photosensitizer and plays a significant role in regulating the photodegradation of ECs in CW systems [7,66]. In aquatic environments, DOM is widely recognized as a major precursor of reactive oxygen species (ROS) and excited triplet-state DOM (^{3}DOM*), which can enhance the photolytic transformation of ECs [7,66]. For example, the photodegradation of propranolol and sulfamethoxazole in municipal wastewater effluent has been reported to occur predominantly through ^{3}DOM*-mediated photosensitized reactions [66]. DOM generally promotes the photodegradation of ECs by generating reactive intermediates, it may also inhibit photolytic transformation under certain environmental conditions through light attenuation and the scavenging of reactive species [7]. A recent study demonstrated that increasing concentrations of humic acid, a major component of dissolved organic matter (DOM), inhibited the photolytic degradation of the antiviral drug acyclovir, resulting in a decrease in the rate constant from 0.01023 to 0.00816 h^{-1} [20,21,66,67].

4.2.4. Microbial Biodegradation

Microbial biodegradation is a key mechanism governing the removal of ECs in CWs, involving their biotransformation, and in some cases, subsequent transformation by diverse microbial communities, including heterotrophic and autotrophic bacteria, fungi, and other microorganisms [98–100]. These microbial transformation processes may occur under both aerobic and anaerobic conditions, depending on the prevailing physicochemical and redox conditions within the CW matrix (Figure 9) [1,21,22,63]. The notation d (shown as d in Figure 9) represents the radial distance from the root centre towards the surrounding

soil and indicates the spatial gradient along which oxygen concentration and microbial distribution vary. Radial oxygen loss (ROL) from plant roots plays a crucial role in regulating the physicochemical and microbial characteristics of constructed wetland substrates. Oxygen released through the root surface creates localized aerobic microzones within the predominantly anaerobic substrate, with oxygen concentration generally decreasing as the radial distance from the root centre increases. This oxygen gradient influences microbial abundance, diversity and community composition across the root–soil interface. The rhizosphere, which is directly influenced by root activity, generally supports enhanced microbial abundance and diversity due to the combined availability of root-derived oxygen and organic exudates. Bulk soil, located farther from the root system, generally experiences relatively limited root-derived oxygen and organic substrates, resulting in different microbial community structures and metabolic activities. These spatially differentiated aerobic and anaerobic microenvironments facilitate complementary microbial processes, including aerobic oxidation and anaerobic degradation pathways, thereby promoting the transformation and biodegradation of emerging contaminants in constructed wetlands. For example, aerobic bacteria can utilize primary substrates such as ammonia, propane, and butane to degrade ECs into carbon dioxide and water [48]. Anaerobic heterotrophic bacteria in CWs can break down chloroform via a two-step mechanism involving fermentation and methanogenesis [4,22]. Microbial groups in these anaerobic pathways include methanogens, sulfate-reducing bacteria, and fermenters with terminal electron acceptors like nitrate and sulfate, facilitating the degradation processes [4,20,35,96–100]. Previous studies have shown that the biodegradation of ECs is a complex process influenced not only by their physicochemical properties [44,66], but also by the design and environmental conditions of CWs. Factors such as the type of CW, substrate material, feeding mode, pH, dissolved oxygen levels, and temperature play a significant role in this process [72,73]. The molecular structure of ECs, including their mono- or polycyclic configurations and the presence of specific functional groups, such as carbonyl and ester moieties, plays a critical role in determining their degradation pathways and overall removal efficiency in constructed wetland systems [73]. Biodegradable ECs generally possess structural characteristics such as short side chains, unsaturated aliphatic moieties, and electron-donating functional groups, which facilitate microbial transformation. Persistent ECs commonly contain long or highly branched alkyl chains, saturated or polycyclic ring structures, and electron-withdrawing substituents, such as halogens and sulfate groups, which may increase their resistance to biodegradation [73]. Diclofenac, a chlorinated pharmaceutical containing two chlorine atoms, generally exhibits relatively low and variable removal efficiencies across different CW configurations, ranging from 20–50% in SF-CWs, 24–93% in HSSF-CWs, and 53–73% in VSSF-CWs [21,28,67]. The biodegradation of ECs in CW systems is strongly influenced by environmental conditions, particularly pH, redox potential, and temperature, which can substantially affect microbial activity and contaminant transformation processes.

The biodegradation of ECs generally follows two primary pathways: metabolic and co-metabolic reactions. In metabolic reactions, various bacterial strains isolated from CW substrates can utilize specific ECs as a carbon source for growth, thereby transforming them into biomass, carbon dioxide, and other by-products [52]. In co-metabolic processes, ECs do not serve as primary growth substrates or energy sources for functional microorganisms [56]. Their transformation occurs incidentally through enzymatic activities associated with microorganisms involved in the degradation of other growth-supporting substrates, including sulfate-reducing bacteria and ammonia- and methane-oxidizing bacteria [28,52].

Among these, ammonia-oxidising bacteria may significantly contribute to EC removal in CWs. For example, Zhang et al. [28] observed a strong positive correlation between the removal of ibuprofen and NH_4-N, suggesting that ammonia oxidizers could be key agents in

ibuprofen degradation. Despite substantial advances in identifying functional microorganisms involved in EC removal within constructed wetlands, the intricate interactions between microbial communities and contaminant transformation processes remain inadequately understood. Advances in molecular biology and omics-based approaches, including high-throughput pyrosequencing, quantitative real-time PCR (qPCR), and metagenomic analysis, are expected to provide deeper insights into the composition, functional roles, and dynamics of specific microorganisms and microbial communities involved in the removal and transformation of pharmaceuticals in CW systems [7,32,63]. High-throughput sequencing has become a widely adopted tool for exploring how system design, hydraulic conditions, and environmental factors influence the structure and composition of microbial communities in CWs [21,52]. High-throughput sequencing approaches are primarily qualitative and may not provide a direct quantitative assessment of EC removal or the abundance of specific functional microorganisms and genes involved in contaminant degradation. Quantitative real-time PCR (qPCR) can serve as a valuable complementary approach for establishing quantitative relationships between EC removal performance and functional microbial populations or genes. Chen et al. [56] reported significant correlations between chloroform biodegradation rates in plant-litter-amended CWs and the copy numbers of methyl-CoM reductase A (MCRA) and dissimilatory sulfite reductase A (DSRA) genes, with coefficients of determination (R^2) of 0.81 and 0.97, respectively. These strong correlations suggest that microorganisms harboring DSRA and MCRA genes may play important functional roles in chloroform biodegradation within CW systems.

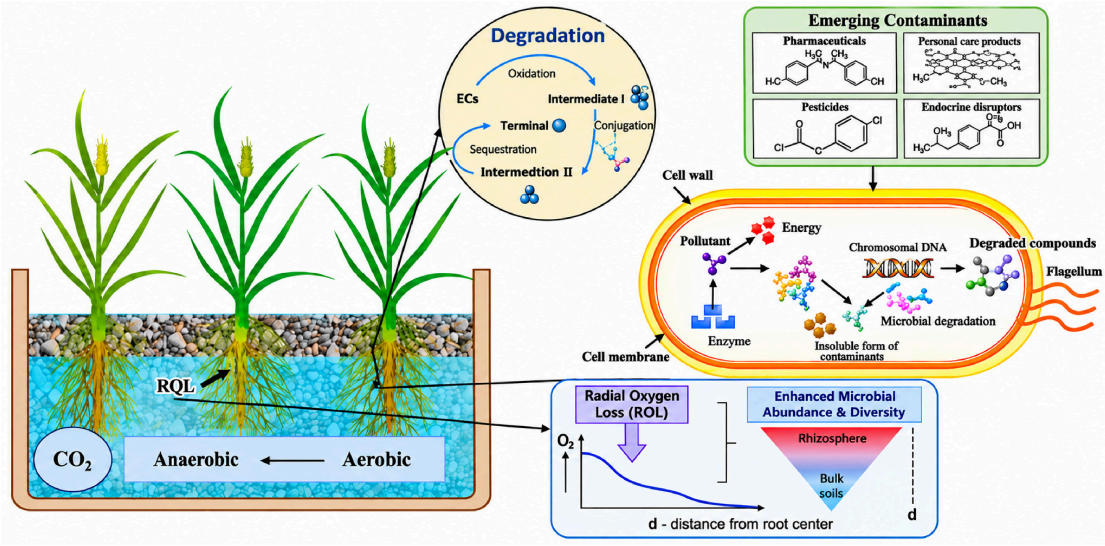


Figure 9. Schematic representation of EC removal in CWs via plant–microbe interactions, rhizosphere processes, and microbial degradation.

5. Influence of CW Design and Operational Parameters on EC Removal

Based on the available literature, a comprehensive synthesis of the key removal processes involved in the elimination of ECs in CWs was conducted. Experimental investigations indicated that a variety of possible removal processes were involved in EC removal, such as photodegradation, biodegradation, sorption, and phytoremediation. The removal processes involved in EC attenuation do not operate independently; rather, EC elimination in CWs generally results from the synergistic interaction of multiple mechanisms (Table 4). As illustrated in Figure 5, the relative contribution of individual removal pathways cannot be universally quantified or generalized, as it varies substantially with CW configuration, contaminant characteristics, operational parameters, and prevailing environmental condi-

tions [7]. It is widely accepted that VFCWs favour aerobic microbial processes and HFCWs support anaerobic biodegradation. Meanwhile, substrate type, vegetation species and physicochemical properties of target ECs could all affect each removal process, therefore leading to the variation of relative contribution of each removal mechanism [20,66]. Photodegradation might be the predominant removal process responsible for the removal of readily photodegradable ECs comprising naproxen and diclofenac, while biodegradation, sorption and phytoremediation might play a major role in the removal of ECs in subsurface flow CWs (SSF CWs) [27]. The factors influencing EC removal performance in CWs can generally be categorized into three major groups: system design, operational conditions, and environmental factors. These parameters exert significant and often variable effects on the removal efficiency of ECs, depending on the CW configuration and treatment conditions. Therefore, optimizing system design and operational parameters offers a feasible approach to improving the overall performance of CWs for EC removal [80].

5.1. Design of CWs

A CW system is an engineered system that uses natural processes involving vegetation, soil, and associated microbial communities to treat wastewater. Below is a detailed design framework for a CW system.

5.1.1. Wetland Types

A comparative summary of the removal efficiencies of selected emerging contaminants in horizontal flow, vertical flow, and hybrid constructed wetlands is provided in Table S3, highlighting variations in removal performance among different CW configurations. As shown in Figure 10, caffeine, ibuprofen, and naproxen exhibited higher removal efficiencies in VFCWs than in HFCWs and hybrid CW systems. In addition to differences in redox conditions and oxygen availability, CW configuration can substantially influence the composition, diversity, and functional characteristics of microbial communities, thereby affecting EC transformation across HFCWs, VFCWs, and hybrid systems [28,72]. The rhizosphere micro-environment present in SSFCWs can create spatial and temporal variations in redox potential, which often enhances the degradation of ECs. Previous studies have shown that conventional VFCWs and HFCWs can achieve high ibuprofen removal efficiencies, ranging from 15% to 99% [20,21,28,52,72]. As shown in Figures 2 and 11, variations in water flow patterns among HFCWs, VFCWs, and hybrid systems lead to distinct redox potentials and oxygen conditions. The configuration of CWs is an important determinant of EC degradation and overall removal performance [28]. VFCWs have generally been reported to exhibit greater biodegradation potential for ECs than HFCWs, which is largely attributed to their enhanced oxygen transfer and predominantly aerobic conditions [23].

HFCWs are generally considered anoxic systems; the rhizosphere can create aerobic microenvironments that support the growth of aerobic microorganisms. These oxidized zones contribute to the removal of ECs. However, the limited oxygen availability in HFCWs often results in moderate removal efficiencies for ECs [63]. Therefore, strategies aimed at enhancing oxygen transfer and maintaining favourable aerobic conditions may improve their treatment performance. VFCWs generally provide more aerobic conditions and higher redox potentials, which can promote microbial activity and the biodegradation of ECs. VFCWs have been reported to achieve comparatively higher removal efficiencies for ECs with greater biodegradability than HFCWs [47,48]. For example, compounds such as diclofenac, caffeine, ibuprofen, naproxen, etc. are more effectively removed in VFCWs than in HSSFCWs [20,27,52]. HFCWs have been recognized as a viable treatment technology for the removal of photodegradable ECs, as photodegradation processes can occur in HFCWs or in open water bodies where solar radiation directly reaches the water surface [20,52]. For

instance, previous studies have indicated that the removal of ECs such as ibuprofen and diclofenac in HFCWs was predominantly facilitated by solar-induced photodegradation [23]. However, in CW systems characterized by high macrophyte density, the effectiveness of photodegradation may be significantly reduced due to the attenuation of sunlight by plant canopies, which limits light penetration to the water surface [4,22,52]. The integration of multiple CW configurations has been increasingly adopted as an effective strategy to enhance the removal of organic pollutants, including organic matter and nutrients [81]. The hybridization of CW systems has shown promise in enhancing the removal efficiencies of ECs, particularly those that are refractory in nature [24,81,82]. With the increasing complexity of wastewater composition in recent years, single-stage CWs may be inadequate for the effective removal of a wide range of ECs [72,73]. Diclofenac removal has been reported to be more effective under aerobic conditions, highlighting the strong influence of the prevailing redox environment on its transformation, whereas sulfamethoxazole generally exhibits greater removal under anoxic conditions [76,82]. The diverse and often contrasting physicochemical characteristics and biodegradation requirements of ECs suggest that hybrid CW systems combining VFCWs and HFCWs could provide complementary redox conditions, thereby enhancing the simultaneous removal of multiple contaminants, including diclofenac and sulfamethoxazole. A recent mesocosm-scale study developed a two-stage hybrid CW system, comprising VFCWs in the first stage followed by HFCWs in the second. The system exhibited high removal efficiencies for the target ECs, achieving 95% for diclofenac and 75% for sulfamethoxazole [82]. The hybrid CW configuration offers a promising strategy for the co-removal of various ECs, which commonly co-occur in wastewater matrices. For example, one study investigated the treatment efficacy of a hybrid CW system comprising two alternating VFCWs followed by a single HFCW unit for the removal of twenty-two ECs [48]. The results demonstrated that VFCW beds were predominantly responsible for the removal of acetaminophen, carbamazepine, caffeine, diclofenac, and naproxen, while HFCWs contributed significantly to the attenuation of caffeine and triclocarban. The integration of VFCWs and HFCWs into hybrid CW systems represents an effective intensification strategy for improving the overall removal of ECs [81,82]. As shown in Figure 10, the removal efficiencies of individual ECs varied considerably among different CW configurations. Hybrid CWs and VFCWs exhibited superior removal performance compared with standalone HFCWs and VFCWs [20,21].

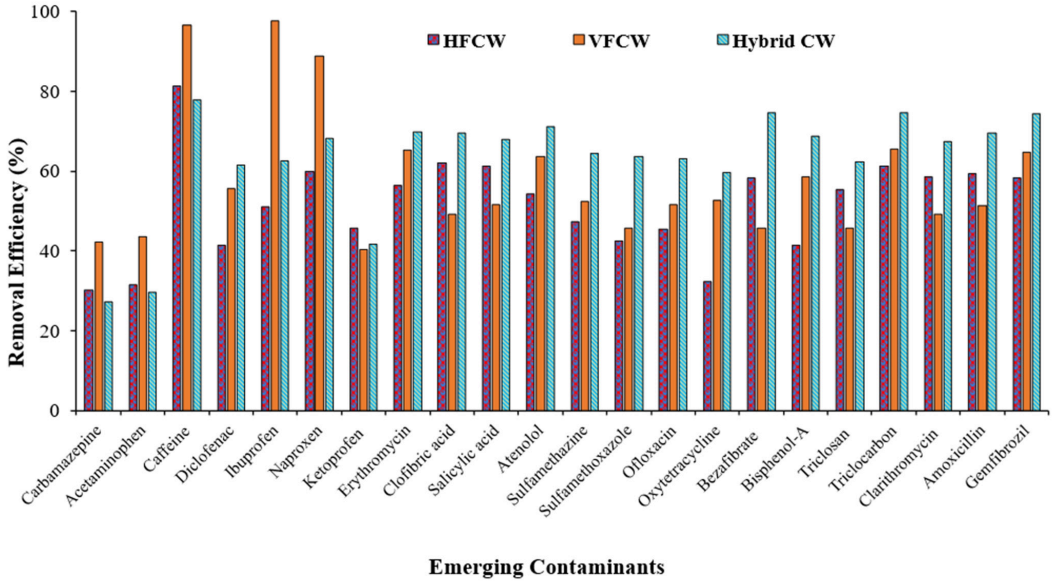


Figure 10. Removal of selected ECs in CWs with different setups: HFCWs, VFCWs, and hybrid CWs.

Table 4. Design and experimental parameters of constructed wetlands for emerging contaminant removal.

Emerging Contaminants	Influent Water	Plants	Types of CWs	Configuration and Hydraulic Conditions	References
Acetaminophen, Caffeine Carbamazepine, Bisphenol-A, Ketoprofen, Ibuprofen, Diclofenac, Clofibric acid	Stormwater, Wastewater	*Phragmites australis*	Hybrid VFCWs HFCWS	VF (3 m^2); HF (6 m^2) The porous media thickness 45.0 cm	[4,23,32,76]
Caffeine, Di-isobutyl Phthalate, Tris(1-chloro-2-propyl) phosphate, Diethyl phthalate, Di-n-octyl phthalate Bis (2-ehtylxexyl) phthalate	University campus wastewater, Greece	*Typha latifolia*	Hybrid VFCWs HFCWS	Diameter of 0.82 m × 1.5 m. 3 m × 0.75 m × 1 m. The porous media thickness was 50.0 cm	[27,43,67,72]
Ibuprofen, Caffeine	Wastewater Brazil	*Heliconia rostrata* and *Eichornia crassipes*	$VFFFCW_S$	VFCW (80 × 60 cm) HRT 15 days	[21,43,67,77,83]
Carbamazepine, Gemfibrozil, Sulfapyridine Sulfamethoxazole,	Rural and municipal wastewater Manitoba, Canada	*Typha* sp.	Hybrid Lagoon and SF-CW in series	Two lagoons in series: 134 m × 134 m × 2.3 m each; water depth; 1.5 m, storage volume: 29,400 m^3 each. SF-CW in between lagoons	[23,27,43,78]
Enrofloxacin, Tetracycline	Swine wastewater Portugal	*Phragmites australis*	VSSF-CW	Sixteen microcosms: Seven were planted and nine were left unplanted (0.4 m × 0.3 m × 0.3 m) Water depth: Just below the substrate surface (100% flooding)	[21,23,31,43,51,61]
Acetaminophen, Diclofenac, Ibuprofen	Urban wastewater from a municipal sewer in Spain	*Phragmites australis*	Hybrid HSSF-CWs	Three treatment lines: (1) Control line: settler-wetland permanently saturated. (2) batch line: settler-wetland operated with saturated/unsaturated phases; (3) anaerobic line: HUSB wetland permanently saturated. Layout of each treatment line: Two wetlands in parallel (0.95 m × 0.7 m × 0.45 m each). Water depth: 25 cm. HRT: 2.5 d per treatment	[20,22,23,43,57,63,75]

Table 4. *Cont.*

Emerging Contaminants	Influent Water	Plants	Types of CWs	Configuration and Hydraulic Conditions	References
Caffeine, Carbamazepine, Clofibric acid, Diclofenac, Ibuprofen, Ketoprofen, Salicylic acid, Naproxen	Synthetic urban wastewater in Singapore	*Typha angustifolia*	HSSF-CW	Container: 1.2 m × 0.6 × 0.6 m. Gravel depth: 30 cm. Water depth: 25 cm. HRT: 2 d or 4 d. Plants' initial density: 14 plants m^{-2}.	[23,28,65]
Carbamazepine Diclofenac Ibuprofen, Naproxen		*Typha angustifolia*	HSSF-CW		[20,28,65]
Amoxicillin, Ampicillin, Clarithromycin Doxycycline Erythromycin Lincomycin Sulfamethoxazole Sulfadimethoxine Trimethoprim	Urban wastewater from the primary clarifier of Leon WWTP in Spain	*Typha angustifolia*	SF-CW (with gravel)	Container: 130 cm long, 80 cm wide and 50 cm deep; 25 cm of free water was over 25 cm layer of gravel (D10 b = 4 mm). Effluent pipe: At the top. HRT: 3.3 d. Plant initial density: 50 plants m^{-2}.	[26,39,48,59]
		-	SF-CW (with gravel, unplanted)	Container: 130 cm long, 80 cm wide and 50 cm deep; 25 cm of free water was over 25 cm layer of gravel (D10 = 4 mm). Effluent pipe: At the bottom. HRT: 6.1 d.	[20,22,26,48]
		Phragmites australis	HSSF-CW	Container: 130 cm long, 80 cm wide and 50 cm deep. Substrate (gravel): 45 cm deep; D10 = 4 mm. Water depth: 40 cm. Effluent pipe: At the bottom. HRT: 2.5 d. Plant initial density: 50 plants m^{-2}.	[22,23,48,52,57]
		-	HSSF-CW (unplanted)	Container: 130 cm long, 80 cm wide and 50 cm deep. Substrate (gravel): 45 cm deep; D10 = 4 mm. Water depth: 40 cm. Effluent pipe: At the bottom. HRT: 2.6 d.	[20,48,52,55]
		Phragmites australis	Hybrid HSSF-CWs	Five treatment lines: Two CW. First CW tank: each 93 cm × 55 cm × 45 cm. Second CW tank: 134 cm × 100 cm × 80 cm. Gravel depth: 30 cm. Water depth: 25 cm HRT: 3.5 d.	

Table 4. *Cont.*

Emerging Contaminants	Influent Water	Plants	Types of CWs	Configuration and Hydraulic Conditions	References
Caffeine, Carbamazepine, Diclofenac, Ibuprofen, Ketoprofen, Naproxen, Salicylic acid	Urban wastewater from the primary clarifier of Leon WWTP in Spain	*Typha angustifolia*	SF-CW (with gravel)	Container: 130 cm × 80 cm × 50 cm. Water depth: 25 cm. HRT: 3.3 d.	[26,48,59,64]
		-	SF-CW (with gravel, unplanted)	Container: 130 cm × 80 cm × 50 cm. HRT: 6.1 d.	[26,59,64]
		Phragmites australis	HSSF-CW	Container: 130 cm × 80 cm × 50 cm. Water depth: 40 cm. HRT: 2.5 d	[4,28]
		-	HSSF-CW (unplanted)	Container: 130 cm × 80 cm × 50 cm. Water depth: 40 cm. HRT: 2.6 d.	[12,59]
		Phragmites australis	Hybrid HSSF-CWs	Five treatment lines: Two CW tanks in series in each line. First CW tank: Two parallel compartments, each 93 cm long, 55 cm wide and 45 cm deep. Second CW tank: 134 cm long, 100 cm wide and 80 cm deep. Gravel depth: 30 cm. Water depth: 25 cm. HRT: 3.5 d. Plant initial density: 50 plants m^{-2}.	[20,21,52,79]
Monensin, Salinomycin, Narasin	Synthetic agricultural wastewater in Canada	*Phalaris arundinacea*, *Typha latifolia*	SF-CW HSSF-CW	Container: A high-density polyethene half-pipe, 6.1 m long, 1.5 m wide and 0.75 m deep; 9.29 m^2; 4.17 m^3. HRT: (1) SF-CW: 4.1 d; (2) HSSF-CW: 2.2 d. Water depth: (1) SF-CW: 0.15 m of water was over 0.6 m of substrate; (2) HSSF-CW: 0.2 m of water was below the substrate surface (0.6 m) at the outlet. Inlet zone: An inflow distribution manifold and a transverse deep zone without plants in 2 m length. Treatment zone: A shallow zone (4.1 m long) vegetated with alternate bands of plants. Outlet zone: A tipping bucket and a sump. Substrate: (1) In the six SF-CWs: three were filled with sandy clay loam soil; the other three were filled with sandy soil; (2) in the three HSSF-CWs: sandy soil.	[20,21,79]

Table 4. *Cont.*

Emerging Contaminants	Influent Water	Plants	Types of CWs	Configuration and Hydraulic Conditions	References
Carbamazepine, Clofibric acid, Ibuprofen	Synthetic urban wastewater (two organic carbon sources: one was starch; the other was glucose)	*Phragmites australis*	HSSF-CW	Container: 0.93 m long, 0.59 m wide and 0.52 m deep. Substrate (gravel): 35 cm deep; initial porosity was 40%; D60 = 3.5 mm; Cu = 1.7. Water depth: 30 cm. Operating mode: In batch.	[23,27,32]
Paracetamol	Wastewater near Lecce, Italy	*Phragmites australis*, *Typha latifolia*	HSSF-CW	Three HSSF-CWs: *Phragmites* bed, *Typha* bed, and unplanted bed (3 m wide, 5 m long, 9.4 m^3 each). Water depth: 0.6 m–0.65 m. Substrate (upper to lower): 0.1 m of soil layer; 0.2 m of stone layer; 0.30–0.35 m of gravel layer. Side banks: 45° inclination.	[76,78]
Caffeine, Carbamazepine, Clofibric acid, Diclofenac, Ibuprofen, Ketoprofen, Naproxen, Salicylic acid	Synthetic urban wastewater in Singapore	*Typha angustifolia*	HSSF-CW	Container: 1.2 m long, 0.6 m wide and 0.6 m deep. Gravel depth: 30 cm. Water depth: 25 cm. HRT: 2 d or 4 d. Plants initial density: 14 plants m^{-2}.	[24]
Ciprofloxacin HCl, Oxytetracycline HCl, Sulfamethazine	Swine wastewater from a pig farm in Xiamen, China	*Hybrid pennisetum*	VSSF-CW	Two VSSF-CWs: 1 m^2 each. Substrate (from upper to lower):20 cm layer of red soil; 30 cm layer of volcanic rock or zeolite; 10 cm layer of gravel. Plant initial density: 15 plants m^{-2}. HRT: 30 h	[45,47]
Ibuprofen Caffeine Bisphenol-A	Wastewater	*Saccharum* spp., *Panicumcropo.*, *Heliconia psittacorum.*, *Vetiveria zizanoides y Clorophytum conmutatum*	HSSFW	Container: 1.2 m long, 0.6 m wide and 0.6 m deep. Gravel depth: 30 cm. Water depth: 25 cm. HRT: 2 d or 4 d. Plants initial density: 14 plants m^{-2}.	[66]

Table 4. *Cont.*

Emerging Contaminants	Influent Water	Plants	Types of CWs	Configuration and Hydraulic Conditions	References
Sulfadiazine, Sulfamethazine, Sulfamethoxazole	Swine wastewater from a concentrated swine feeding plant located in Nanjing, China	*Ryegrass (Lolium): Dryan*, *Waseyutaka*, *Tachimasari*	SF-CW	Container: 0.5 m long, 0.4 m wide and 0.4 m deep. Working depth: 0.3 m. Capacity volume: 60 L. Foam plate for macrophytes floating: 0.5 m long, 0.4 m wide and 0.015 m thick, with 16 rounded holes (4 × 4, 2 cm diameter). Eight treatment systems (with replications): Two Dryan beds, two Waseyutaka beds, two Tachimasari beds, and two unplanted beds.	[76]

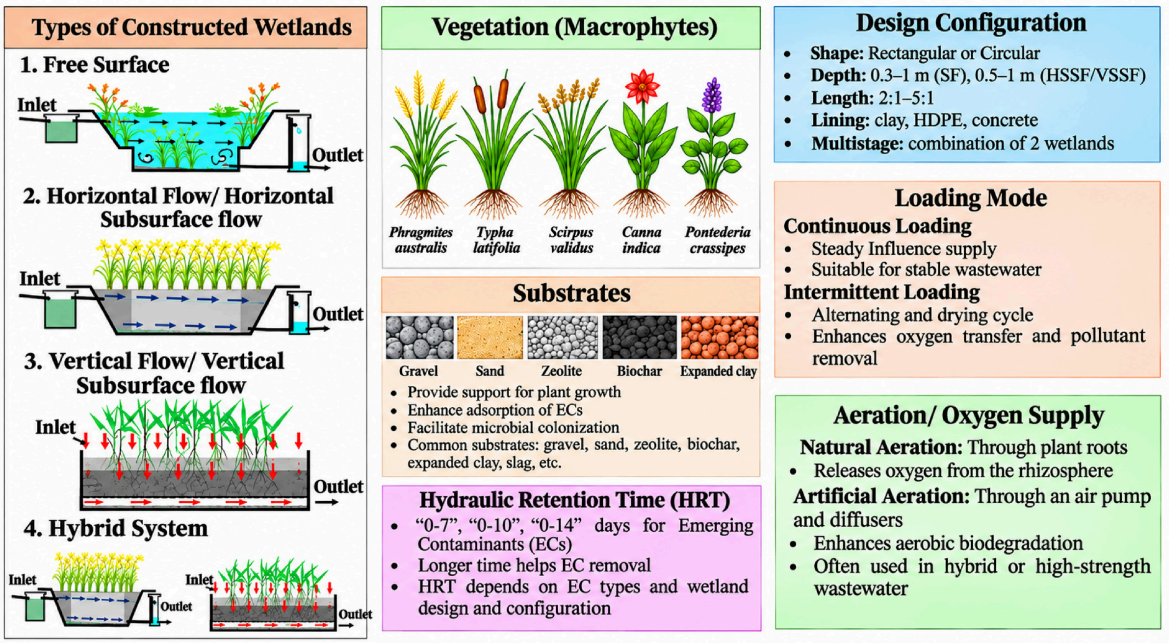


Figure 11. Design framework of constructed wetlands for emerging contaminant removal.

The removal efficiencies of individual ECs differed substantially across the investigated CW configurations, highlighting the compound-specific nature of EC attenuation (Figure 10). Hybrid CWs demonstrated the highest and most consistent removal performance, followed by VFCWs, while HFCWs exhibited comparatively variable removal efficiencies [4,21,22] (Figure 10). The enhanced performance of hybrid systems can be attributed to the integration of aerobic, anoxic, and anaerobic microenvironments, greater microbial diversity, improved contaminant–media interactions, and the synergistic contribution of biodegradation, sorption, plant uptake, and other physicochemical processes [21,22]. By combining the complementary treatment characteristics of VFCWs and HFCWs, hybrid systems provide a broader range of environmental conditions capable of facilitating the removal of ECs with diverse physicochemical characteristics and degradation pathways. The higher removal efficiency of particular CW configurations for specific ECs highlights the significant influence of contaminant-specific properties and redox-dependent transformation pathways on overall EC removal [21,22,56].

5.1.2. Wetland Plants

The literature-based comparison of emerging contaminant removal efficiencies in planted and unplanted constructed wetlands is provided in Table S4, highlighting the influence of vegetation on contaminant removal performance. Wetland vegetation has been shown to play a crucial role in enhancing the removal of ECs from CWs. Numerous studies have demonstrated that wetland plants facilitate the transfer of oxygen from their aerial tissues to the rhizosphere, thereby increasing dissolved oxygen (DO) availability within the wetland matrix and promoting aerobic microbial processes [20,52] (Figure 11). This oxygen enrichment creates a more favorable microenvironment for microbial proliferation compared to unplanted systems [35]. Enhanced oxygen availability further supports greater microbial diversity and abundance, ultimately augmenting the biodegradation potential of ECs [20]. Recent studies have demonstrated that plant species selection can influence the abundance and community dynamics of nitrite-oxidizing bacteria and complete ammonia-oxidizing (comammox) microorganisms within CW systems [20,47]. In addition to oxygen transfer, plant-derived root exudates supply readily available carbon and energy sources, which stimulate microbial activity and facilitate the biodegradation of ECs (Figure 11). These

exudates may directly contribute to the removal of ECs through hydrophilic interaction-mediated sorption processes [72]. Previous studies have compared the removal efficiencies of emerging ECs in planted versus unplanted CWs. Due to the limited availability of data for many ECs, statistical comparisons were performed only for a select few compounds, including carbamazepine, ibuprofen, caffeine, naproxen, diclofenac, sulfamethoxazole, and ketoprofen (Figure 10). Planted constructed wetlands generally exhibited higher removal efficiencies than unplanted systems for most emerging contaminants [21], highlighting the contribution of vegetation to contaminant removal. Among the investigated compounds, caffeine showed the highest removal (~82%), followed by sulfamethazine (~74%), naproxen (~64%), ibuprofen (~63%), and sulfamethoxazole (~62%), whereas gemfibrozil (~45%) and carbamazepine (~41%) showed comparatively lower removal in the planted CWs [20,21] (Figure 12). In unplanted systems, removal efficiencies were generally lower, ranging from approximately 22–77%, with caffeine again showing the highest removal. The greatest differences between planted and unplanted systems were observed for oxytetracycline, lincomycin, bisphenol A, triclocarban, and clarithromycin, whereas compounds such as fluconazole showed relatively smaller differences. The results indicate that vegetation can enhance EC removal through plant uptake, rhizosphere-mediated microbial processes, sorption, and associated physicochemical interactions, although removal performance remains compound-specific. The presence of vegetation may not result in statistically significant improvements in treatment performance. Despite extensive research into the role of wetland vegetation in EC removal, the current understanding of species-specific contributions to the elimination of individual ECs remains limited [22]. As such, there is no consensus regarding the comparative removal efficiencies of ECs across different planted CW configurations.

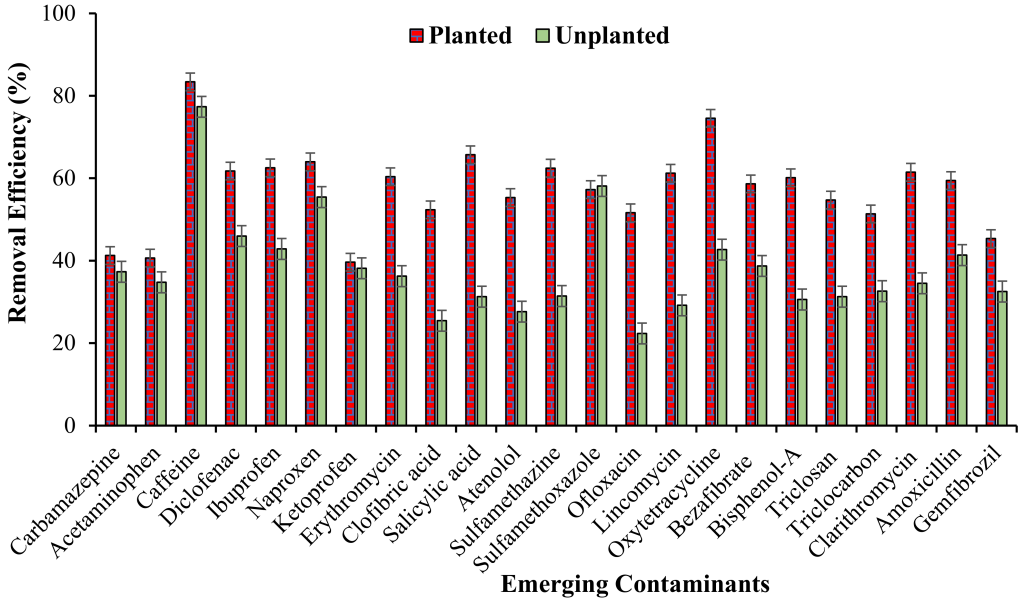


Figure 12. Comparative removal efficiency of selected emerging contaminants in planted and unplanted constructed wetlands.

Numerous studies have established that wetland vegetation plays a pivotal role in the removal of emerging contaminants through processes such as plant uptake, phytodegradation, and phytovolatilization. The incorporation of plant litter (i.e., senescent plant biomass) into constructed wetlands has emerged as a promising approach for enhancing contaminant removal efficiency and overall treatment performance [21,23]. Plant litter can function as a natural bioadsorbent, a source of primary carbon substrate, and a redox mediator,

thereby facilitating the degradation and removal of ECs [4,21,56,73]. Previous research has indicated that plant biomass may exhibit a higher sorption capacity for certain contaminants, such as chloroform, compared to conventional CW substrates, due to its elevated organic matter content [21,56]. Chen et al. [4] reported that CWs amended with cattail litter achieved the highest chloroform sorption rates, ranging from 810 to 950 $\mu g\ m^{-2}\ d^{-1}$, among all tested mesocosms. Plant litter can serve as a carbon source to support functional microbial populations involved in co-metabolic transformation pathways, thereby enhancing EC removal efficiency in CWs [20,56,74] (Figure 12). Although research on the contribution of plant litter to EC removal in CWs remains relatively scarce, available evidence indicates that plant litter can promote the diversity and abundance of bacterial communities within CW systems, thereby altering microbial community composition at both the class and genus levels [24,81,83].

Recent studies have shown that the addition of plant litter can stimulate the growth of key functional microbial groups, including organic matter-degrading microorganisms, nitrifying bacteria, and denitrifying bacteria [20,52,73]. These microbial communities play an important role in the biodegradation and transformation of ECs within CWs. Therefore, incorporating plant litter may enhance EC removal by promoting the development of functionally diverse microbial communities that contribute to contaminant degradation processes [73].

5.1.3. Wetland Substrates/Compositions

The substrate or media matrix is an important component of constructed wetlands, providing structural support for vegetation and microbial colonization while also contributing directly to the retention and attenuation of emerging contaminants through sorption and associated physicochemical interactions [4]. Common CW media include gravel, sand, zeolite, ceramic materials, oyster shells, and other porous substrates, while modified media such as biochar and manganese-oxide-coated sand can provide additional sorption capacity [21,23]. The selection and particle size of the media influence EC removal by controlling surface area, hydraulic conductivity, oxygen availability, and opportunities for microbial colonization. For example, studies summarized in the reviewed literature reported a lower removal of ibuprofen, acetaminophen, diclofenac, and bisphenol A in vertical-flow CWs containing fine gravel compared with systems using coarse sand, which was associated with differences in surface area and oxygen availability [4,52]. In horizontal-flow CWs, the reported substrate performance followed the order zeolite > medical stone > ceramic > sand > oyster shells > gravel, indicating that media characteristics strongly influence contaminant retention [84]. Zeolite can provide favorable sorptive interactions because of its microporous structure and surface functional groups, whereas biochar and manganese-oxide-coated sand have been investigated as enhanced media because of their comparatively high sorption capacity [52]. The role of media is also compound-specific because ECs differ substantially in hydrophobicity, ionization, molecular structure, and solubility [52]. Hydrophobic compounds such as triclosan, triclocarban, and bisphenol A may interact strongly with organic-rich or high-surface-area media, whereas compounds such as carbamazepine, atenolol, and several antibiotics can exhibit different sorption and transformation behaviour depending on the substrate and prevailing pH and redox conditions [27,85]. For example, the reviewed evidence identifies sorption as an important pathway for carbamazepine, bisphenol A, clarithromycin, atenolol, and ofloxacin, while compounds such as ibuprofen, diclofenac, naproxen, and acetaminophen may undergo a combination of biodegradation, sorption, photodegradation, and phytoremediation [4,52,84]. To improve the removal of ECs, novel substrate materials, including biochar and manganese oxide-coated sand, have been explored because of their enhanced sorption capacities [84]. CWs amended with

Fe-impregnated biochar derived from *Cyperus alternifolius* achieved up to 99% pesticide removal, substantially exceeding the 45–64% removal observed in CWs without biochar amendment [85].

Fe-impregnated biochar-amended CWs have shown significantly improved pesticide removal efficiency compared to those without biochar. This enhancement is likely attributed to the increased sorption capacity and microbial degradation promoted by Fe-biochar, which improves the removal of ECs [23,47]. In addition to high-sorption substrates, researchers have also developed materials with strong catalytic and oxidative properties to further boost the performance of CWs. Ji et al. [83] emphasised the advantages of these modifications. Xie et al. [86] developed an innovative CW using manganese oxide (birnessite)-coated sand, which achieved triclosan removal efficiencies of over 90.5%, significantly higher than the 83.8% observed in unmodified CWs (Figure 12). The authors attributed this enhanced removal performance to the strong adsorptive and oxidative properties of the Mn oxide-coated substrate. Engineered substrates with enhanced sorption capacity and catalytic oxidation potential offer considerable promise for the effective degradation of persistent and toxic emerging contaminants (ECs). Thus, optimizing the substrate composition emerges as a promising approach to enhance EC removal in CWs. Substrates can indirectly impact the removal efficiency of ECs by shaping the structure and activity of microbial communities [86]. Kizito et al. [87] investigated the long-term performance of vertical subsurface flow (VSSF) CWs using three different substrates: corn cob biochar, wood biochar, and conventional gravel. Their results showed that systems packed with biochar (corn cob and wood) achieved significantly higher removal efficiencies for organic matter (up to 59%) compared to gravel-based systems (22–49%). Moreover, biochar-supported systems exhibited a richer and more diverse microbial community, particularly in the case of wood biochar. Substrate selection plays a crucial role in the effectiveness of CWs for wastewater treatment. The selection of an appropriate substrate can provide greater surface area and improved oxygen availability while fostering microbial diversity and activity, thereby enhancing the removal of ECs [20,72]. Moreover, substrates with high sorption capacity can further improve treatment performance by promoting the direct adsorption and retention of ECs within the wetland matrix [20,38].

5.2. Operational Parameters

5.2.1. Hydraulic Retention Time (HRT)

Hydraulic retention time (HRT) is a critical operational parameter governing the removal of emerging contaminants in constructed wetlands, as it determines the duration of interaction among wastewater, wetland media, microorganisms, and vegetation [21]. In general, an adequate HRT provides sufficient time for removal processes such as biodegradation, sorption, plant uptake, and photodegradation. However, the relationship between HRT and EC removal is compound- and system-specific, depending on the physicochemical properties of individual contaminants, CW configuration, substrate characteristics, microbial activity, and prevailing redox conditions [27]. Positive associations between HRT and removal have been reported for several pharmaceuticals, including diclofenac, ketoprofen, clarithromycin, ofloxacin, sulfamethazine, and gemfibrozil. For example, increasing HRT from 3 to 9 days improved oxytetracycline removal from 89% to 99%, suggesting that adequate contact time can enhance adsorption and biodegradation [21–23]. Similarly, the removal of pesticides such as imazalil and tebuconazole increased with increasing HRT in planted CW mesocosms containing *Typha latifolia*, *Phragmites australis*, *Iris pseudacorus*, and *Juncus effusus* [72]. Other studies have reported increasing EC removal at HRTs of approximately 27.5, 30.5, 45.9, and 137.5 h [39]. Longer HRT does not necessarily result in proportionally greater EC removal. For example, diclofenac removal has been reported

to increase from 39% to 94% when HRT increased from 10 to 13 days, whereas caffeine removal reached approximately 96% at 5 days compared with >50% at 3 days. Conversely, ibuprofen and naproxen achieved approximately 80% and 91% removal, respectively, at an HRT of 4 days in planted HFCWs, indicating the importance of biodegradation and rhizosphere-associated processes. In contrast, carbamazepine showed comparatively low removal (<25%) even at approximately 5 days HRT, reflecting its greater persistence and recalcitrance [53]. Campos et al. [88] reported no significant correlation between HRT and EC removal under certain concentration conditions, demonstrating that the effect of HRT cannot be generalized across contaminants or operating conditions. Nominal HRT should be distinguished from effective or tracer-derived HRT. Nominal HRT is theoretically calculated as $HRT = V/Q$, whereas tracer-derived HRT represents the actual hydraulic residence behaviour of wastewater within the CW. Hydraulic short-circuiting, preferential flow, dead zones, and incomplete mixing can cause substantial deviations between nominal and effective HRT, thereby reducing the actual contact time available for EC attenuation [21,22,32,37,42]. Hydraulic efficiency and tracer-based residence-time characteristics should be considered alongside nominal HRT when evaluating EC removal, particularly in hybrid and multi-stage CWs [4,6,23,32]. Thus, HRT should be optimized rather than simply maximized. Excessively short HRT may restrict contaminant–media interactions and microbial transformation, whereas unnecessarily long HRT can increase land requirements, construction costs, and operational constraints without ensuring proportional improvement in EC removal [55,72]. Hybrid CWs may partly overcome these limitations by providing sequential treatment compartments with complementary aerobic, anoxic, and anaerobic conditions, thereby increasing the range of transformation pathways available for compounds such as diclofenac, acetaminophen, sulfamethoxazole, sulfapyridine, trimethoprim, and atenolol [53,55,57]. Effective HRT should be interpreted together with contaminant-specific properties, dominant removal mechanisms, hydraulic efficiency, substrate characteristics, redox conditions, and plant–microbe interactions rather than being considered an independent predictor of EC removal.

5.2.2. Loading Mode

Feeding modes in CWs are generally classified into three categories: continuous, batch, and intermittent flow. Increasing evidence suggests that batch and intermittent feeding can enhance EC removal compared with continuous-flow operation [27]. This improvement may be attributed to periodic transitions between saturated and unsaturated conditions, which facilitate oxygen replenishment and promote favourable redox conditions within the wetland bed [27]. Enhanced oxygen availability can subsequently stimulate microbial activity and biodegradation processes involved in EC removal. Recent studies have reported higher EC removal efficiencies in batch-operated CWs [20,48]. For example, batch-fed HFCWs demonstrated improved removal of ibuprofen, acetaminophen, and bisphenol A compared with continuous-flow operation [48]. Maillard et al. [89] investigated the influence of feeding mode on EC removal in SSFCWs planted with *Phragmites australis* and reported higher degradation rates under batch operation than under continuous feeding. The enhanced performance was associated with increased redox potential conditions induced by periodic feeding. Therefore, batch or intermittent feeding represents a promising operational strategy for improving EC removal in CWs by enhancing oxygen availability, redox dynamics, and microbial degradation processes [20,21,47].

5.2.3. Aeration

Aeration can overcome oxygen-transfer limitations within CW beds and improve redox conditions, thereby promoting microbial-mediated removal processes. This intensification

strategy is particularly relevant to HFCWs [47]. Oxygen availability within CWs is strongly influenced by system configuration, with saturated HFCWs generally exhibiting limited oxygen transfer, which can constrain the removal of ECs [90]. Accordingly, active aeration has been proposed as an effective approach for enhancing EC treatment performance by increasing oxygen availability and modifying redox conditions. Previous studies have reported improved removal of several ECs, including caffeine, ibuprofen, naproxen, diclofenac, acesulfame, and carbamazepine, under actively aerated conditions [90]. These findings suggest that artificial aeration can enhance microbial activity and associated biodegradation processes, thereby improving the overall EC removal efficiency of CWs. Multiple studies have investigated the impact of artificial aeration on EC removal to assess its potential for improving treatment outcomes [91]. Auvinen et al. [92] investigated the impact of active aeration on the removal of ECs, specifically metformin and valsartan, in a pilot-scale SSFCW. Continuous aeration has been reported to substantially enhance EC removal compared with non-aerated conditions, increasing removal from 68 ± 32% to 99 ± 1% for metformin and from 17 ± 19% to 99 ± 1% for valsartan. Wolff et al. [91] demonstrated that active aeration improved the treatment performance of CWs for antibiotic removal. Avila et al. [48] further reported the enhanced removal of several ECs, including ibuprofen, acetaminophen, diclofenac, triclosan, ethinylestradiol, and bisphenol A, in HFCWs treating municipal wastewater (Table 5). Despite increasing evidence of the effectiveness of active aeration in improving EC removal, its influence on the subsequent transformation and mineralization of EC-derived transformation products remains insufficiently understood. Enhanced parent-compound removal does not necessarily indicate complete degradation or detoxification. Further research is therefore required to elucidate how active aeration influences EC transformation pathways, the formation and persistence of intermediate products, and their potential residual toxicity in CWs [92,93].

5.2.4. Influence of Vegetation

Wetland vegetation can influence the removal of emerging contaminants in constructed wetlands by modifying the rhizosphere environment and supporting microbial and plant-mediated processes. Plant roots facilitate the transfer of oxygen from the aerial parts of the plant to the rhizosphere, which can increase dissolved oxygen concentrations and create favourable conditions for microbial colonization and biodegradation [3,4,94]. Root exudates provide readily available carbon and energy sources that can stimulate microbial metabolic activity and potentially contribute to contaminant sorption and transformation. Plants can contribute to the removal of emerging contaminants in constructed wetlands by transporting oxygen from their aerial tissues to the rhizosphere and releasing root exudates that stimulate microbial activity [23,95]. Oxygen transfer increases dissolved oxygen availability around the roots, creating more favourable aerobic microsites for microbial colonization and the biodegradation of ECs. Meanwhile, root exudates provide readily available carbon and energy sources that can enhance microbial metabolic activity, thereby supporting the transformation and attenuation of contaminants [27,96]. The contribution of vegetation, however, is compound- and system-dependent and should not be considered a universally dominant removal mechanism. The reviewed studies indicate that plants can enhance the attenuation of pharmaceuticals such as naproxen, ibuprofen, diclofenac, and caffeine, with *Phragmites australis* showing greater removal performance than *Typha* spp. under some conditions, particularly during summer. Differences between *P. australis*- and *Typha*-planted systems for acetaminophen removal were not statistically significant, indicating that plant species alone may not determine EC removal performance [21].

Table 5. Quantitative assessment of EC removal performance in constructed wetlands.

ECs	Types of CWs	Plants	Substrate	Loading Rate (cm/d, m/d)	HRT (d, h)	Operational Mode	Scale	Influent Water	References
Ibuprofen Ketoprofen Naproxen Diclofenac Caffeine Salicylic acid Galaxolide	HFCWs FCWs	*Typha angustifolia*, *Phragmites australis*	Gravel	5cm/d	29.6, 37.4, 37.8, 54 37.4, 41.8, 71.1	Continuous	Mesoscale	Urban wastewater in Spain	[47,48]
Caffeine Carbamazepine N, N-diethyl-meta-toluamide Fluoxetine	Hybrid CWs	*Phragmites australis*	Fine and coarse gravel	-	-	Intermittently	Mesocosm	Urban wastewater from municipal sewer in Spain	[4,48]
Ibuprofen	HFCWs VFCWs	*Typha latifolia* *Phragmites australis* *Juncus effuses*	Coarse gravel, Sand	4 HLRs (1.8, 3.4, 6.9, 13.8 cm/d)	-	Continuous	Microcosm	Synthetic wastewater, Aarhus, Denmark	[7,28,80]
Triclocarban Sulfamethoxazole	VFCWs	*Typha latifolia* *Phragmites australis* *Juncus effuses*	Mixed gravel	2.2 cm/d	-	Batch	Microcosm	Simulated wastewater, Canada	[65]
Diclofenac Sulfamethoxazole	Hybrid CWs SSFCWs and FWCWs in serial	CW1: *Miscanthus giganteus* CW2: *Eichhornia crassipes* *Pistia stratiotes*	Coarse gravel, Sand	-	CW1: 7 h CW2: 161 h	-	Mesocosm	Synthetic wastewater	[81,82]
Tonalide Oxybenzone	Hybrid CWs, VFCWs, HFCWs, FWSCWs in series	*Phragmites australis* *Typha* spp., *Scirpus* spp., *Iris pseudacorus*, *Cyperus rutundus*, *Juncus* spp.,	Sand Siliceous gravel	-	>7.4 d	Intermittently	Full scale	Wastewater, municipality with urban runoff	[57]

Table 5. *Cont.*

ECs	Types of CWs	Plants	Substrate	Loading Rate (cm/d, m/d)	HRT (d, h)	Operational Mode	Scale	Influent Water	References
Tramadol Naproxen Bisphenol-A Triclocarbon Diclofenac Caffeine Ibuprofen Acetaminophen Atenol Ofloxacin Erythromycin Gemfibrozil	FWSCWs	*Salix alba*, *Iris pseudacorus*, *Juncus*, *Carex caryophylla*	-	0.2	3.5 day	Continuous	Full scale	Wastewater in France	[84]
Diclofenac, Acetaminophen, Trimethoprim Sulfamethoxazole Caffeine Diclofenac	HF + HF + FWS	*Cyperus alternifolius*, *Thalia dealbata*, *Canna lily*, *Iris pseudacorus*, *Oenanthe javanica*	-	0.8	1.6 day	Continuous	Full scale	Wastewater in China	[79]
Ofloxacin Sulfamethoxazole Carbamazepine	VFCWs	*Phragmites australis*	Sand	8.0	0.5 day	Continuous	Lab scale	Wastewater in China	[69,79]
Gemfibrozil Caffeine Ibuprofen Caffeine	HFCWs	*Typha angustifolia*, *Chrysopogon zizanioide*, *Cyperus papyrus*	-	-	1 day	Continuous	Full scale		[67,93]
Ketoprofen Diclofenac Naproxen Erythromycin Lincomycin Metoprolol Clofibric acid Sulfamethoxazole Carbamazepine	FWSCWs, FWSCWs, VFCWs	FWSCWs; *Phragmites australis* FWSCWs; *Phragmites australis*, *tree* VSSFCWs; *Phragmites australis*	-	FWSCWs; 43.3 cm/d, FWSCWs; 102.7 cm/d, VSSFCWs; 17.6 cm/d	FWSCWs; 4 d, FWSCWs; 0.82 d, VSSFCWs; 1.7 d	-	Full scale	Effluent in the Netherlands	[93]

Vegetation may therefore contribute to EC removal through several interacting pathways, including rhizosphere oxygenation, stimulation of microbial biodegradation, root-associated sorption, plant uptake, and phytotransformation [23]. These processes have been reported for ECs including carbamazepine, acetaminophen, caffeine, diclofenac, ibuprofen, naproxen, ofloxacin, ciprofloxacin, and steroid hormones, although the relative contribution of individual pathways differs among compounds [67]. From a design and operational perspective, plant selection should consider species characteristics, root-system development, compatibility with the CW configuration, and the physicochemical characteristics of the target ECs. *P. australis* and *Typha* spp. are among the commonly investigated species in VFCWs, HFCWs, and hybrid systems treating pharmaceuticals, personal-care-related contaminants, endocrine-disrupting compounds, and antibiotics [21,23]. For example, *P. australis* has been used for systems targeting acetaminophen, caffeine, carbamazepine, bisphenol A, ketoprofen, ibuprofen, diclofenac, and clofibric acid, whereas *Typha* spp. have been investigated for compounds including caffeine, carbamazepine, gemfibrozil, sulfamethoxazole, and phthalates [23]. Vegetation should be regarded as an important supporting component of CW design rather than as a standalone or universally dominant mechanism for EC removal. Its effectiveness depends on plant species, root development, microbial communities, contaminant properties, wastewater characteristics, and hydraulic and environmental conditions. Therefore, vegetation management should be integrated with appropriate substrate selection, hydraulic configuration, and oxygen/redox management to achieve consistent EC attenuation [4,21,23].

Comparative studies suggest that the effectiveness of vegetation is compound- and species-dependent [21]. *Phragmites australis* has been reported to enhance the degradation of naproxen, ibuprofen, diclofenac, and caffeine, and in some cases, to perform better than *Typha* spp., particularly during warmer summer conditions. However, differences between plant species are not always statistically significant. For example, although acetaminophen removal showed a slight difference between *P. australis*- and *Typha*-planted HFCWs, the difference was not significant at the 95% confidence level [48,91].

The available experimental designs further demonstrate that vegetation effects should be interpreted alongside hydraulic and operational conditions. For example, studies in the reviewed literature have evaluated *P. australis*, *Typha latifolia*, and *Typha angustifolia* under HSSF/HFCW systems, with reported HRTs ranging from approximately 1–4 days in several experiments and plant densities of about 14–50 plants m^{-2} [22,23]. These differences in HRT, plant density, substrate characteristics, and system configuration make direct comparison of species-specific EC removal efficiencies difficult. The reviewed evidence does not consistently demonstrate a statistically significant advantage of planted over unplanted systems. Although EC removal efficiencies were generally higher in planted CWs, several studies reported no significant differences between planted and unplanted configurations. Moreover, young vegetation may contribute more substantially to EC removal than mature plants, suggesting that plant age, biomass development, and seasonal growth should be considered when assessing vegetation effects. Beyond living plants, plant litter may also contribute to EC removal. Senescent biomass can act as a natural bioadsorbent, provide an additional carbon source, and function as a redox mediator [77]. For example, cattail-litter-amended CWs exhibited chloroform sorption rates of approximately 810–950 $\mu g\ m^{-2}\ d^{-1}$ in the reported mesocosm experiments. Plant litter may also stimulate microbial communities, including organic-matter degraders and nitrifying and denitrifying bacteria, thereby indirectly supporting EC biodegradation. Vegetation should therefore not be considered a universally dominant removal mechanism. Its contribution is likely determined by the interaction among plant species, root architecture, plant age, seasonal growth, rhizosphere oxygenation, microbial communities, contaminant properties, and hydraulic/operational

conditions. Future research should quantify the relative contributions of plant uptake, phytotransformation, rhizospheric biodegradation, and sorption using controlled planted–unplanted comparisons and compound-specific mass balances [4,21,67].

5.2.5. Influence of Specific Surface Area (m^2/PE)

Specific surface area (m^2/PE) is a potentially useful design and performance indicator for comparing CW systems with different treatment capacities. It is generally expressed as the horizontal treatment area allocated per population equivalent (PE), where PE represents the equivalent organic/hydraulic loading generated by a defined number of inhabitants. In the reviewed studies, PE is estimated based on the applied hydraulic load and an assumed standard per-capita wastewater generation rate [91]. Thus, m^2/PE provides an indication of the treatment area available relative to the wastewater load imposed on the CW and can be used to facilitate comparisons among systems of different sizes. However, the relationship between specific surface area and EC removal should be interpreted cautiously. A larger m^2/PE value may theoretically provide greater contact area, longer effective residence time, and increased opportunities for sorption, biodegradation, plant uptake, and other removal processes. The available evidence does not establish m^2/PE as an independent or consistent predictor of EC removal. EC removal is simultaneously controlled by contaminant physicochemical properties, hydraulic loading rate, HRT, substrate characteristics, redox conditions, vegetation, microbial activity, temperature, and CW configuration [91,92]. Increasing the treatment area per PE does not necessarily result in a proportional increase in EC removal. A limitation is that m^2/PE is often reported for the entire CW treatment train rather than for the individual treatment stage responsible for EC removal, making comparisons among specific configurations difficult [92]. This issue is particularly important for hybrid systems, in which different units may provide distinct aerobic, anoxic, anaerobic, or sorptive environments. The reviewed literature also contains relatively few studies that systematically evaluate EC removal as a function of m^2/PE, preventing robust conclusions regarding its predictive value. Therefore, m^2/PE should currently be considered a comparative design indicator rather than a validated standalone predictor of EC removal efficiency [92]. Future studies should systematically evaluate the relationship between specific surface area and EC removal using standardized PE calculations while controlling for HLR, HRT, influent EC concentration, CW configuration, substrate, vegetation, and climatic conditions. Reporting EC mass removal or removal efficiency per unit treatment area (e.g., mg m^{-2} d^{-1}) alongside m^2/PE would provide a more mechanistic basis for determining whether increasing the treatment area actually improves EC removal. Such analyses would help establish whether an optimum surface-area-to-loading ratio exists for the removal of different EC classes.

5.3. Redox Potential

Redox potential (RP) is an important operational parameter governing the prevailing electron-acceptor conditions, and consequently, the microbial transformation pathways of emerging contaminants (ECs) in constructed wetlands (CWs) [48]. However, RP should not be interpreted using rigid numerical thresholds to classify an entire CW bed as either aerobic or anaerobic. Unlike homogeneous engineered reactors, CWs contain spatially and temporally heterogeneous microenvironments generated by differences in oxygen diffusion, plant-mediated oxygen transport, organic-matter availability, microbial activity, water distribution, and substrate characteristics [48]. Aerobic, anoxic, and anaerobic zones can coexist within the same wetland bed, and their relative extent can vary with depth, distance from the inlet, hydraulic loading, season, and operational conditions [4,23]. Measured RP should be regarded primarily as an operational indicator of the prevailing redox state at a specific location and time, rather than as a universal threshold defining a particular removal mechanism.

Relatively oxidizing conditions may favour oxygen-dependent biodegradation, whereas more reducing environments can support alternative electron-acceptor processes involving nitrate, iron or manganese oxides, sulfate, and, under strongly reducing conditions, carbon dioxide [23]. Thus, the effect of RP on EC removal depends not only on the measured potential, but also on the availability and competition among electron acceptors and the metabolic capabilities of the microbial community. Both oxidising and reducing microsites may therefore contribute to the overall transformation of ECs within CWs [91]. ECs illustrate this compound-specific behaviour. Avila et al. [47] reported higher ibuprofen removal under aerobic conditions (85%) than under control (63%) and anaerobic conditions (52%), suggesting that oxygen-associated processes may contribute substantially to its transformation. In contrast, diclofenac has been reported to exhibit more favorable removal under anaerobic conditions [27], demonstrating that increasing oxygen availability does not necessarily enhance the removal of all ECs. Other compounds appear less sensitive to the prevailing redox environment. For example, acetaminophen and oxybenzone exhibited high average removal efficiencies, generally above 90%, with no significant differences between aerobic and anaerobic conditions [57]. 17β-estradiol and estrone were efficiently removed under both aerobic and anaerobic conditions, with no pronounced difference between the two conditions [95]. These contrasting responses indicate that EC removal cannot be predicted solely from RP measurements. The molecular structure and electron-accepting/donating characteristics of individual compounds, together with microbial community composition and the availability of alternative electron acceptors, can determine whether a particular redox environment favours transformation [95]. Moreover, a measured RP value represents the integrated electrochemical conditions surrounding the electrode and may not directly reflect the microscale conditions experienced by microorganisms or EC molecules within biofilms, plant roots, and porous media. Some studies should characterize RP together with dissolved oxygen, nitrate/nitrite, sulfate, Fe(II)/Fe(III), dissolved organic carbon, and other relevant electron acceptors/donors, while considering spatial gradients within CW beds. Combining these measurements with microbial-community analyses and transformation-product identification would provide stronger evidence for linking redox conditions to specific EC transformation pathways. Although several studies have demonstrated EC removal under different redox conditions, the precise contribution of individual redox-dependent mechanisms remains insufficiently resolved because of the complex and heterogeneous nature of CW environments [57,95]. Thus, RP should be used as a context-dependent operational parameter for interpreting redox conditions rather than as a universal mechanistic threshold.

5.4. Effect of Temperature and pH

Temperature is another crucial environmental factor that significantly influences the removal of ECs in CWs [15,57]. It is well-established that warmer seasons, compared to colder ones, generally enhance the removal efficiency of ECs in CWs. This improvement is primarily attributed to increased microbial activity, higher growth rates, and elevated photo transformation processes at higher temperatures [95]. Cold conditions tend to suppress biological activity and photodegradation, resulting in the accumulation of ECs within the system. Due to limited data on the removal of certain ECs in CWs, the analysis was restricted to a selected group of frequently studied compounds, including carbamazepine, ibuprofen, caffeine, naproxen, diclofenac, salicylic acid, ketoprofen, gemfibrozil, sulfadiazine, and sulfamethoxazole. As illustrated in Figure 12 and Table S5, it indicated significant seasonal variations in the removal efficiencies of caffeine, clofibric acid, salicylic acid.

Seasonal variations in EC removal performance were primarily governed by changes in solar radiation and temperature [85]. Higher solar radiation can promote the photodegra-

dation of light-sensitive ECs, while elevated temperatures enhance microbial metabolic activity and biodegradation kinetics. The combined effects of these factors can therefore contribute to improved EC removal during warmer and more favorable climatic conditions. Matamoros et al. [23] examined seasonal impacts on the removal of 20 ECs in CWs planted with *Daphnia* sp. and found significantly higher removal rates of diclofenac, triclosan, ketoprofen, and naproxen in summer compared to winter, indicating a strong seasonal influence. Matamoros et al. [23] also reported notable seasonal differences in the removal of biodegradable ECs like naproxen and ibuprofen, attributed to temperature-driven changes in biodegradation rates. The removal of diclofenac, ibuprofen, ketoprofen, carbamazepine, and gemfibrozil did not exhibit statistically significant seasonal variation, although removal efficiencies tended to be higher in warmer months (Figure 13). This phenomenon may be attributed to the higher concentrations of ECs observed during colder periods, which could potentially enhance their degradation under first-order kinetic conditions. Plant transpiration, an important process governing the uptake of ECs, generally increases with rising temperatures. Greater evapotranspiration during warmer seasons may enhance plant-mediated EC uptake, suggesting that elevated transpiration rates during summer could further contribute to improved EC removal in CWs [47]. pH has a considerable impact on the environmental behaviour of ECs in CWs. It can influence the efficiency of microbiologically mediated removal processes and also plays a crucial role in the sorption of ECs onto CW substrates [89]. Thus, pH is a key factor affecting EC removal in CWs. Previous research indicates that aquatic microorganisms are highly sensitive to pH variations [8]. Bacterial community composition is strongly influenced by the prevailing physicochemical conditions of aquatic environments, particularly pH [56]. Methanogenic and nitrifying microorganisms exhibit distinct pH optima, with favorable ranges of approximately 6.5–7.5 and 7.0–8.0, respectively. Recent research has demonstrated that EC removal is closely associated with the activities of nitrifying and denitrifying bacteria [56]. Variations in pH can indirectly affect EC removal in CWs by altering microbial metabolism and the activity of key functional microorganisms. The adaptability of different microbial communities to fluctuating pH conditions within CWs remains poorly understood. Furthermore, pH is an important factor governing the fate, transformation, and distribution of ECs within CW systems [95]. pH is a critical factor influencing the speciation of ECs, particularly antibiotics, by determining whether they exist in neutral or ionized forms. This, in turn, affects their sorption onto CW substrates through electrostatic interactions [95]. Bois et al. [96] demonstrated that pH significantly influences glyphosate sorption, with sorption increasing as the pH decreased from 8 to 7. The removal of ECs in CWs is governed not by a single operational or environmental parameter, but by the complex interactions among multiple physicochemical, biological, and hydraulic factors, which collectively influence overall treatment performance [56,95,97].

5.5. Effects of Media Characteristics and Contaminant Properties

The efficiency of CWs in removing contaminants is largely influenced by the type of media used, as it affects the system's physical, chemical, and biological processes. Numerous studies have shown that selecting and optimizing the media can significantly improve contaminant removal by enhancing mechanisms such as adsorption, microbial degradation, and plant uptake [3,52]. The substrate media in CWs serve multiple functions, including providing surfaces for microbial colonization, supporting plant root development, and facilitating contaminant adsorption. Although conventional materials such as sand, gravel, and soil are widely employed, increasing attention has been directed toward advanced substrates, including biochar, activated carbon, zeolite, and iron- or aluminum-coated sands, owing to their enhanced adsorption capacities. Among these, biochar has demonstrated

considerable potential for the removal of hydrophobic ECs and perfluorinated compounds due to its high specific surface area and well-developed porous structure. Activated carbon, in contrast, exhibits a strong affinity for both polar and non-polar ECs, primarily owing to its distinctive surface chemistry and physicochemical properties [33].

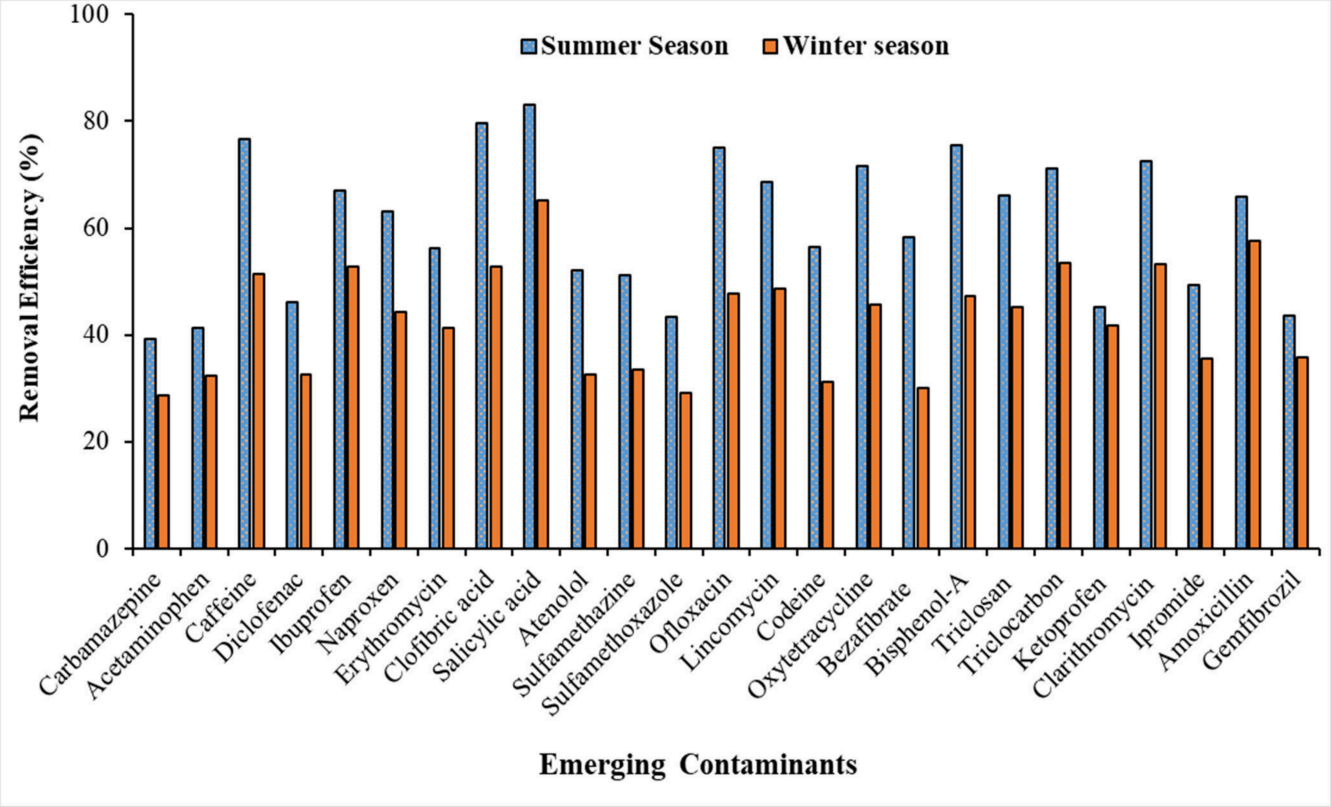


Figure 13. Seasonal variation in the removal efficiency of selected emerging contaminants in constructed wetlands during the summer and winter seasons.

The particle size and composition of the media significantly impact hydraulic conductivity and residence time, two key factors that influence contaminant removal efficiency. Fine-grained media increase the contact time between pollutants and reactive surfaces, thereby enhancing adsorption and degradation processes [33,38]. However, excessive incorporation of fine-grained particles may increase the risk of clogging and consequently compromise CW performance. The addition of materials such as zeolite can enhance the removal of ammonium and other nitrogenous compounds through ion-exchange processes, whereas iron-coated sands can promote phosphorus removal through chemical precipitation and related surface interactions [48]. The composition of the media also affects biological processes, especially microbial degradation. Media rich in organic matter support diverse microbial populations capable of breaking down various ECs, including hormones and antibiotics [3,52]. Plant roots interacting with the media create a rhizosphere enriched in oxygen and organic exudates, promoting microbial activity and contaminant degradation. Studies emphasize the synergistic relationship between the media, microbial communities, and vegetation in achieving high removal efficiency for complex EC mixtures. Environmental factors such as temperature and pH, both influenced by the media, also affect EC behaviour [47,48,92]. pH-modifying substrates, such as dolomite and limestone, can enhance the removal of acidic ECs by modifying their solubility, speciation, and chemical behavior. Reactive media capable of supplying oxygen or establishing distinct redox gradients can further facilitate the transformation and degradation of specific pollutants

under aerobic and anaerobic conditions. Recent advances in CW design have also explored multilayered media configurations to provide diverse physicochemical and biological conditions, thereby expanding the range of ECs that can be effectively removed [20,52]. The removal and fate of ECs in CWs are strongly influenced by their physicochemical and molecular characteristics, which determine their interactions with water, wetland media, vegetation, and microbial communities. Although molecular size can affect transport and retention, it should not be considered an independent predictor of EC removal. A more comprehensive assessment should consider ionisation behaviour, hydrophobicity, molecular structure, functional groups, hydrodynamic transport, and specific interactions with substrates and microorganisms [20,21,47,48,52]. The ionization state of an EC is particularly important because it determines its speciation at the pH conditions prevailing within CWs. Ionizable compounds may occur in neutral or charged forms, and these species can exhibit substantially different sorption affinities and mobility [47]. Log Kow and log D provide complementary information on hydrophobicity and partitioning behaviour. While log Kow is generally more applicable to neutral compounds, log D accounts for the influence of ionization at a specified pH and can therefore provide a more realistic indication of contaminant partitioning under environmentally relevant conditions [48,52]. Hydrophobic neutral compounds are generally more likely to associate with organic matter and hydrophobic domains of substrates or plant tissues, whereas highly polar or ionized compounds may remain predominantly in the aqueous phase and consequently depend more strongly on biodegradation or other transformation processes [92]. The molecular structure and functional groups of ECs also determine their potential interactions with wetland components and their susceptibility to microbial transformation. Compounds containing specific electron-donating or electron-withdrawing groups, aromatic structures, hydrogen-bonding sites, or other reactive functional groups may exhibit different sorption and biodegradation behaviors [21,47,92,98]. For example, triclosan and bisphenol A, which have relatively low water solubility and high octanol–water partitioning, can be substantially removed through association with soil particles and plant roots [48]. Highly polar pharmaceuticals and pesticides may exhibit weaker sorption and greater mobility, although their removal can still be substantial when favorable microbial transformation pathways are present.

Molecular size should therefore be interpreted in conjunction with hydrodynamic transport and molecular structure. Larger molecular dimensions may influence diffusion through pores and biofilms and may affect accessibility to sorption sites; however, molecular size alone does not necessarily imply greater sorption or biodegradation [48]. Transport through a CW is also governed by hydraulic loading, residence time, dispersion, preferential flow, pore structure, and interactions with the substrate and biofilm. A relatively small compound may be strongly retained because of favorable chemical interactions, whereas a larger compound may remain mobile if its interactions with the wetland matrix are weak. The interaction between contaminant properties and microbial community characteristics is equally important [51]. EC transformation depends on whether microorganisms possessing the required metabolic capabilities are present and whether appropriate electron donors and acceptors are available [65]. Some ECs may undergo aerobic transformation within oxygenated rhizosphere zones, whereas others may be preferentially transformed under anoxic or anaerobic conditions [48,65]. Plant roots can further influence these processes by providing attachment surfaces, releasing root exudates, and creating localized physicochemical conditions that support diverse microbial communities. Li et al. [98] demonstrated that plant roots can contribute to the microbial degradation of organic ECs by providing microhabitats that support microbial populations.

6. Sustainability of Constructed Wetlands for Emerging Contaminant Removal

Constructed wetlands are increasingly considered as a potentially sustainable approach for wastewater treatment because they can integrate physical, chemical, and biological processes with relatively simple infrastructure. However, their sustainability should not be regarded as an inherent or universally demonstrated advantage. Rather, the environmental, economic, and social sustainability of CWs depends on wetland configuration, land availability, substrate selection, vegetation, climate, hydraulic loading, contaminant characteristics, maintenance requirements, and the intended end use of the treated water and biomass. Therefore, the sustainability benefits discussed below should be interpreted as potential benefits that require site-specific evaluation and comparison with alternative treatment technologies.

6.1. Potentially Lower Energy Requirements

CWs may require less external energy than energy-intensive conventional treatment systems because contaminant attenuation can occur through naturally driven processes, including sedimentation, sorption, microbial transformation, plant uptake, and photochemical reactions. Systems designed without continuous mechanical aeration may have particularly low direct energy requirements. However, this benefit can be reduced when pumps, recirculation, active aeration, ultraviolet treatment, or other energy-intensive components are incorporated [3]. The energy embodied in construction materials, substrate production, transportation, and periodic replacement should be considered when evaluating the overall energy footprint (Table 6).

Table 6. Evidence-based alignment of constructed wetlands for emerging contaminant removal with relevant Sustainable Development Goals (SDGs).

SDG	SDG Target	Relevant Indicator/Parameter	Relevance to Constructed Wetlands
SDG 3: Good Health and Well-being	Target 3.9: Reduce illnesses and deaths from hazardous chemicals and pollution	Removal of emerging contaminants (ECs), pharmaceuticals, personal care products, endocrine-disrupting compounds and other pollutants	CWs can reduce concentrations of potentially toxic contaminants in wastewater, thereby decreasing their potential environmental and human-health risks
SDG 6: Clean Water and Sanitation	Target 6.3: Improve water quality by reducing pollution and increasing wastewater treatment and safe reuse	EC removal efficiency (%), COD, BOD, nutrients, heavy metals and other water-quality parameters	CWs provide decentralised and nature-based wastewater treatment and can substantially reduce the range of conventional and emerging contaminants
SDG 6: Clean Water and Sanitation	Target 6.6: Protect and restore water-related ecosystems	Reduction in pollutant loading to receiving water bodies; improvement in treated effluent quality	Effective treatment through CWs can reduce contaminant discharge into rivers, wetlands and other aquatic ecosystems
SDG 9: Industry, Innovation and Infrastructure	Target 9.4: Upgrade infrastructure to make it sustainable and resource-efficient	Energy consumption, treatment efficiency, use of low-cost substrates and decentralised treatment systems	CWs represent low-energy, nature-based treatment infrastructure that can be integrated with conventional or advanced treatment systems

Table 6. *Cont.*

SDG	SDG Target	Relevant Indicator/Parameter	Relevance to Constructed Wetlands
SDG 11: Sustainable Cities and Communities	Target 11.6: Reduce the environmental impact of cities, particularly water and waste pollution	Wastewater treatment efficiency, pollutant load reduction and decentralised treatment capacity	CWs can support decentralised wastewater treatment and improve urban wastewater management, particularly where centralised infrastructure is limited
SDG 12: Responsible Consumption and Production	Target 12.4: Environmentally sound management of chemicals and wastes	Reduction in pharmaceutical residues, personal care products and other chemical contaminants in wastewater	CWs can contribute to improved management of chemical contaminants by reducing their release into the environment
SDG 13: Climate Action	Target 13.3: Improve capacity for climate-change mitigation and adaptation	Energy demand, greenhouse-gas emissions, biomass production and carbon-storage potential	CWs generally require relatively low external energy inputs and may provide potential carbon-sequestration and climate-mitigation benefits through vegetation and sediment carbon storage
SDG 14: Life Below Water	Target 14.1: Reduce marine pollution, including pollution from land-based activities	Reduction in contaminant discharge to rivers and downstream aquatic environments	Removal of ECs before discharge can potentially reduce the transport of contaminants from terrestrial wastewater sources to downstream aquatic and marine ecosystems
SDG 15: Life on Land	Target 15.1: Conserve and restore terrestrial and freshwater ecosystems	Pollutant load reduction, vegetation establishment and ecosystem-supporting functions	CWs can create vegetated aquatic habitats and reduce contaminant loads entering freshwater and terrestrial environments. Biodiversity enhancement should be considered a potential co-benefit unless directly assessed.
SDG 6/SDG 13	Water security and climate resilience	Water reuse potential, water retention and evapotranspiration	CWs may contribute to water conservation and climate resilience through wastewater treatment and potential water reuse
SDG 6: Clean Water and Sanitation	Target 6.4: Increase water-use efficiency and sustainable water management	Treated wastewater quality and potential for reuse	Treatment of wastewater through CWs can facilitate safe water reuse where appropriate treatment standards are achieved

6.2. Potentially Lower Operational and Maintenance Costs

The relatively simple design and passive operation of some CWs may reduce energy and routine operational requirements compared with highly mechanised treatment systems. CWs are not maintenance-free. Vegetation management, removal of accumulated solids, hydraulic maintenance, substrate replacement, inlet and outlet maintenance, and

monitoring of clogging and treatment performance may be required [21,23]. Claims of minimal operational costs should be avoided unless supported by comparable life-cycle economic assessments.

6.3. Land Requirement as a Major Trade-Off

One of the principal limitations of CWs is their potentially high land requirement, particularly for systems treating large wastewater volumes or requiring extended hydraulic retention times. Land availability, land cost, soil characteristics, and proximity to the wastewater source can therefore strongly influence their economic feasibility [52]. The relatively low energy demand of CWs may provide an operational advantage, this benefit may be offset in areas where land is expensive or scarce. Compact configurations, intensified CWs, vertical-flow systems, and hybrid designs may reduce the required footprint, but may involve greater construction complexity or energy inputs.

6.4. Potential Contribution to Circular-Economy Principles

CWs may potentially contribute to circular-economy approaches by enabling water reuse, nutrient recovery, biomass production, and the incorporation of alternative substrates or waste-derived materials. For example, selected waste-derived sorbents or biochars may provide opportunities for resource recovery while supporting EC retention. However, the reuse of wetland biomass or spent substrate requires careful consideration because ECs and transformation products may accumulate within these materials [52,84]. Therefore, biomass and spent-substrate management must be based on contaminant characterisation and appropriate risk assessment, and reuse should not be assumed to be environmentally safe.

6.5. Potential Biodiversity Benefits

Vegetated CWs can provide habitat and ecological niches for microorganisms, plants, invertebrates, and other organisms and may therefore potentially enhance local ecological diversity. However, biodiversity benefits depend on wetland design, vegetation composition, landscape context, invasive-species management, and water-quality conditions. Accumulation of contaminants in sediments or biomass may also create ecological risks. Thus, the presence of vegetation alone should not be interpreted as evidence of a net biodiversity benefit [84].

6.6. Climate Resilience and Climatic Dependence

CWs may offer potential advantages in decentralised wastewater treatment and can continue to function using natural biological and physicochemical processes when appropriately designed. Their treatment performance is strongly influenced by temperature, precipitation, evapotranspiration, hydraulic loading, seasonal variation, drought, flooding, and extreme weather events. Low temperatures can reduce microbial activity, whereas excessive rainfall may cause hydraulic overloading and reduced contaminant retention. High temperatures and evapotranspiration can alter hydraulic conditions and contaminant transformation rates. Consequently, climate resilience should be considered context-specific rather than a universal characteristic of CWs.

6.7. Greenhouse-Gas Emissions

Although CWs may have relatively low direct energy consumption, they can generate CO_2, CH_4, and N_2O through microbial processes such as organic-matter decomposition, methanogenesis, nitrification, and denitrification. The magnitude of these emissions depends on wetland type, redox conditions, organic loading, vegetation, water depth, temperature, and operational conditions. Therefore, evaluating CW sustainability solely

on electricity consumption may underestimate their overall climate impact. Life-cycle greenhouse-gas assessment is required for a comprehensive comparison with conventional WWTPs [3,21,52,67].

6.8. Biomass and Contaminant Management

Plant uptake can contribute to the attenuation of selected ECs, but contaminants retained in plant tissues are not necessarily destroyed. Harvesting and subsequent disposal or reuse of contaminated biomass therefore represent important aspects of the treatment process. Sorbed contaminants may accumulate in substrates and sediments and could potentially be released during changing environmental conditions [52]. Long-term sustainability consequently requires consideration of the complete contaminant life cycle, including retention, transformation-product formation, accumulation, desorption, biomass harvesting, and final disposal or reuse.

6.9. Potential Contribution to Sustainable Development Goals

CWs may potentially contribute to several Sustainable Development Goals (SDGs), particularly SDG 6 (Clean Water and Sanitation) through wastewater treatment and potential water reuse; SDG 9 (Industry, Innovation and Infrastructure) through nature-based treatment technologies; SDG 11 (Sustainable Cities and Communities) through decentralised wastewater management; SDG 12 (Responsible Consumption and Production) through resource-recovery approaches; SDG 13 (Climate Action) where low-energy operation and appropriate design reduce life-cycle impacts; and SDG 15 (Life on Land) where CWs provide suitable ecological habitat. However, these contributions are potential rather than automatic, and the achievement of individual SDG targets depends on local environmental, economic, and social conditions.

7. Challenges and Future Prospects

A limited number of ECs have been used as model pollutants to evaluate the treatment performance of CWs. Greater emphasis is needed on investigating the occurrence of ECs in wastewater, especially those categorized as high to medium risk. In addition, more research is required to optimize CW designs to enhance the efficiency of multiple removal processes for diverse ECs [65]. There is insufficient understanding of the fundamental mechanisms behind EC removal in CWs, which involve the interplay between substrates, plants, and microbial communities. Several knowledge gaps remain regarding plant-mediated mechanisms of EC removal, including phytoextraction, phytotransformation, and rhizosphere-mediated interactions. Although available evidence suggests that EC accumulation in CW vegetation through phytoaccumulation is generally limited, the processes governing EC uptake, translocation, transformation, and degradation within plants remain insufficiently understood [18,98]. The specific microbial taxa and functional consortia responsible for EC transformation and removal in CWs have yet to be comprehensively identified. Understanding the relationship between EC removal and microbial community composition is essential for elucidating the underlying biodegradation mechanisms in CWs. Advances in molecular biology, including high-throughput pyrosequencing, real-time PCR, and metagenomic sequencing, provide powerful approaches for characterizing microbial communities and identifying functional genes associated with EC transformation. Chen et al. [56,99] demonstrated a strong association between chloroform biodegradation rates and the abundance of functional genes involved in sulfate reduction and methanogenesis in microcosm-scale CWs using real-time PCR.

The contribution of biodegradation to overall EC removal in CWs is generally estimated indirectly using mass balance methods. Compound-specific isotope analysis (CSIA)

provides a valuable means to quantify biodegradation relative to other removal pathways, such as photodegradation and sorption [43]. Bioassays complement chemical analyses by assessing changes in biotoxicity before and after wastewater treatment. CWs are advantageous as natural treatment systems, though their widespread use is limited by the need for large land areas and the relatively slow, less efficient passive processes compared to conventional treatment technologies. Limited oxygen availability and inadequate oxygen transfer rates remain important constraints that can hinder the effective application of CWs for EC removal. Several enhancement strategies, including hybrid CW configurations, active aeration, and substrate modification, have been proposed to overcome these limitations and improve EC removal efficiency [47]. Further research is required to elucidate the mechanisms underlying these enhancement approaches, optimize their operational conditions, and evaluate their long-term performance, cost-effectiveness, and scalability under field conditions. Future research should also focus on clarifying the roles of microbial communities and functional genes in EC biodegradation under varying redox conditions using advanced omics approaches. Greater emphasis is also needed on identifying transformation products, assessing their toxicity, and developing mass-balance approaches to evaluate treatment effectiveness [65]. The removal, retention, and precursor transformation of PFAS and other persistent contaminants require systematic investigation, including optimized substrates and hybrid treatment configurations. Research should further develop climate-resilient CW designs capable of maintaining stable performance under changing temperature, precipitation, hydraulic loading, and redox conditions. Systematic optimization of wetland type, substrate, vegetation, hydraulic retention time, loading rate, and aeration is recommended. Finally, long-term field-scale studies should evaluate seasonal variability, microbial adaptation, substrate saturation, plant development, and maintenance requirements, supported by standardized methodologies and performance indicators.

8. Conclusions

Constructed wetlands demonstrate considerable potential for the attenuation of emerging contaminants from wastewater and can be applied as standalone, secondary, or tertiary treatment systems. The reviewed studies indicate substantial variability in EC removal performance, with reported removal efficiencies generally ranging from approximately 20% to 95%, depending on the contaminant, wetland configuration, substrate, hydraulic conditions, vegetation, and environmental conditions. Several compounds, including acetaminophen, salicylic acid, sulfadimethoxine, and caffeine, exhibited high average removal efficiencies (>85%), whereas compounds such as tramadol, gemfibrozil, and iopromide showed comparatively low removal. These findings demonstrate that CW performance cannot be generalized across all ECs and instead reflect strong compound- and system-specific behaviour. The removal of emerging contaminants in constructed wetlands involves a combination of interconnected physical, chemical, and biological processes, including sorption/adsorption, microbial biodegradation, photodegradation, sedimentation, plant uptake, and other transformation pathways. The relative contribution of these mechanisms is governed by the physicochemical properties of individual contaminants, including hydrophobicity, ionization state, solubility, and molecular structure, together with substrate characteristics, vegetation, redox conditions, hydraulic retention time, oxygen availability, and wetland configuration. The reviewed evidence indicates that EC removal is compound- and system-specific, with biodegradation, sorption, photodegradation, and phytoremediation contributing to varying degrees depending on the contaminant, CW configuration, and prevailing environmental and operational conditions. Hybrid CWs, particularly combinations of vertical- and horizontal-flow configurations, offer an important approach for improving treatment performance by providing complementary aerobic and anaerobic

microenvironments. The reviewed evidence includes hybrid systems achieving a high removal of compounds such as ketoprofen (81.25%), diclofenac (87.47%), and caffeine (96.64%), demonstrating the potential advantage of integrating different hydraulic and biogeochemical conditions. Negative or incomplete removal has also been reported for some ECs, indicating that apparent removal may sometimes involve transformation, desorption, or precursor conversion rather than complete degradation.

From a sustainability perspective, CWs offer advantages including relatively low energy requirements, potentially lower operational costs, and compatibility with decentralized wastewater treatment. However, their broader implementation is constrained by land requirements, relatively slow treatment processes, oxygen-transfer limitations, and uncertainties associated with long-term EC fate. The identification of transformation products, determination of their toxicity, characterization of functional microbial communities, and quantitative separation of individual removal pathways remain important research needs. From a circular economic perspective, CWs offer potential for nutrient recovery (nitrogen and phosphorus), biomass production, and integration with other nature-based solutions, supporting closed-loop systems. Climate-resilient CW designs can adapt to variable hydraulic and pollutant loading conditions, making them suitable for decentralised wastewater treatment in both developed and developing regions. From a sustainability perspective, constructed wetlands (CWs) support the achievement of several United Nations Sustainable Development Goals, including SDG 6 (Clean Water and Sanitation), SDG 11 (Sustainable Cities and Communities), SDG 13 (Climate Action), and SDG 14 (Life Below Water). To enhance the long-term performance and sustainability of constructed wetlands (CWs), future studies should focus on system optimisation, life cycle assessment, and scalability under real-world conditions.

9. Novelty

The novelty of this review lies in its integrated assessment of emerging contaminant (EC) removal by constructed wetlands (CWs), establishing explicit relationships between contaminant physicochemical properties, wetland configurations, design and operational parameters, and the underlying removal mechanisms. The review critically compares CW performance with conventional wastewater treatment technologies, with particular emphasis on compound-specific removal behaviour, including negative removal and contaminant transformation. It further evaluates the influence of substrates, vegetation, hydraulic conditions, and aeration on treatment performance and identifies critical research gaps concerning transformation products, microbial processes, ecotoxicological implications, and long-term system sustainability. By integrating these aspects, the review provides a comprehensive framework for the development and optimization of efficient, resilient, and sustainable CW systems for EC removal. The findings contribute to several United Nations Sustainable Development Goals (SDGs), particularly SDG 3 (Good Health and Well-being), SDG 6 (Clean Water and Sanitation), SDG 12 (Responsible Consumption and Production), SDG 13 (Climate Action), and SDG 14 (Life Below Water), by supporting improved water quality, pollution control, resource-efficient treatment, and sustainable wastewater management.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/w18222878/s1. Table S1. Comparison of the removal efficiencies (%) of selected ECs in CWs and conventional wastewater treatment plants (WWTPs); Table S2. Relative contribution (%) of the major removal mechanisms: phytoremediation, biodegradation, sorption, and photodegradation in the removal of selected ECs in CWs; Table S3. Literature-based comparison of the reported removal efficiencies (%) of selected emerging contaminants in horizontal flow constructed wetlands (HFCWs), vertical flow constructed wetlands (VFCWs), and hybrid constructed wetlands (Hybrid CWs) [3–6,9,11,15,18–21,23,25,26,35,41,43,47–54,58–60,66–68,71–73,

77–79,84–86,88,90–92]. Table S4. Literature-based comparison of the removal efficiencies (%) of selected ECs in planted and unplanted CWs; Table S5. Seasonal variation in the removal efficiencies (%) of selected ECs during summer and winter in constructed wetland systems.

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Abbreviations

EC	Emerging contaminant	WWTP	Wastewater treatment plant	RQ	Risk quotient
ECs	Emerging contaminants	PhACs	Pharmaceutical active compounds	S^w	Water solubility
PhCs	Pharmaceutical compounds	PPCPs	Pharmaceutical and personal care products	pKa	Acid–base dissociation constant
CW	Constructed wetland	ARB	Antibiotic-resistant bacteria	EU	European Union
CWs	Constructed wetlands	PCBs	Polychlorinated biphenyls	SDGs	Sustainable Development Goals
		HFCWs	Horizontal flow constructed wetlands		
ARGs	Antibiotic resistance genes	VFCWs	Vertical flow constructed wetlands	$FSCW_S$	Free surface constructed wetlands
EDCs	Endocrine-disrupting compounds	PhCs	Pharmaceutical contaminants	SSFCWs	Subsurface free constructed wetlands
DOM	Dissolved organic matter	PCR	Polymerase chain reaction	NOM	Natural organic matter
FWS	Free water surface	MCRA	Methyl-com reductase A	CSIA	Compound-specific isotope analysis
VSSF-CW	Vertical subsurface flow constructed wetland	HSSF-CW	Horizontal subsurface flow constructed wetland	Log KoW	Water partition coefficient

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