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PERSPECTIVE ARTICLE

Biological carbon sequestration in constructed wetlands: A nature-based strategy for climate mitigation and wastewater treatment

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Abstract

Constructed wetlands are engineered ecosystems composed of substrates, vegetation, and microorganisms that serve as a sustainable alternative for wastewater treatment. Despite their benefits, greenhouse gas emissions continue to pose a challenge. Consequently, increasing research efforts focus on developing cleaner strategies to reduce overall global warming potential while enhancing carbon sequestration. This perspective outlines the current status of carbon sequestration and wastewater treatment, highlighting the close coupling between pollutant removal and carbon cycling. It also describes existing limitations and explores future prospects.

Keywords: Carbon sequestration; Constructed wetlands; Greenhouse gas emissions; Pollutants; Wastewater treatment

1. Introduction

Wetlands are found across diverse landscapes and may contain either permanent or temporary shallow water. Their soils, substrates, and biological communities are specially adapted to flooding and waterlogging, which promotes the development of anaerobic conditions.[1] Wetlands occupy approximately 5–8% of the Earth's land surface. They typically consist of freshwater, soil, vegetation such as macrophytes and microphytes, and diverse microbial communities.[2]

Wetlands are generally classified into two categories: Natural wetlands and constructed (man-made) wetlands. The most widely used classification systems categorize wetlands into five types: marine (coastal wetlands), estuarine (including mangrove swamps, deltas, and tidal marshes), lacustrine (associated with lakes), riverine (along rivers and streams), and palustrine (bogs and swamps). These divisions are based on hydrological, ecological, biological, and environmental characteristics.[2]

Wetland vegetation plays a key role in wastewater treatment, as plants can directly absorb nutrients from the water, thereby enhancing the overall efficiency of contaminant removal. Natural wetlands—such as wet grasslands, mangroves, and salt marshes—are characterized by static or flowing water, and their soils are rich in carbon (C), functioning as major C sinks. These ecosystems also serve as hotspots for biogeochemical cycling of C.[3] The C sequestration rate of wetlands, expressed as the amount of C stored per

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hectare per year, ranks just below that of forests and is comparable to agroecosystems.[4] In coastal wetlands, frequent tidal inundation alters the environment and slows the decomposition of plant organic matter, which can enhance long-term C retention; however, site-specific hydrodynamics can also limit soil C density.[5] Studies have shown that methane (CH_4) fluxes from natural wetlands (15.6–49.5 mg CH_4/m^2/day) are substantially higher than those observed in disturbed wetlands (−1.4–4.0 mg CH_4/m^2/day).[4] Despite these emissions, natural wetlands remain prominent C sequesters, underscoring the importance of their protection in the context of a growing global population.

In contrast, constructed wetlands (CWs) are engineered systems designed to harness natural processes involving soils, vegetation, and associated microbial communities for wastewater treatment.[6] For instance, vertical flow CWs equipped with gravel substrates achieve less than 50% removal of emerging organic contaminants, including acesulfame, carbamazepine, benzotriazole, and naproxen.[7] Similarly, Muduli *et al.*[8] reported that a baffled horizontal subsurface-flow CW efficiently removed the following contaminants: Chemical oxygen (O_2) demand (COD; 68.1%), total suspended solids (86.5%), total phosphorus (64.8%), total nitrogen (N) (78.25%), and NH_4^+–N (95.2%). CWs generally exhibit shorter C accumulation periods and lower soil C density compared with coastal wetlands. However, due to the high net primary productivity of wetland plants and the continuous input of organic matter and nutrients from wastewater, CWs achieve high C sequestration rates in soil, resulting in considerable C storage potential.[5]

As roots release C and N, they stimulate microbial activity in the soil. These interactions often lead wetlands to function as C sinks, where C storage exceeds C release. The root zone, or rhizosphere, is defined as the soil region influenced by hydraulic, chemical, and microbial processes associated with roots and has been shown to enhance immobilization of contaminants.[9] CWs are recognized as crucial systems for wastewater treatment, where the influx of diverse pollutants often creates redox gradient that can increase CH_4 emissions, and under certain conditions, promote N_2O production, thereby contributing to global warming.[10]

Research on wetlands primarily focuses on their ecological functions, conservation, biodiversity, water quality improvement, nutrient cycling, and ecosystem restoration. This perspective highlights the critical role of CWs as platforms for both wastewater treatment and biological C sequestration. However, their net climate benefit is constrained by greenhouse gas (GHG) emissions, particularly N_2O and CH_4. Addressing these trade-offs through essential design and management measures is necessary to maximize climate mitigation while ensuring the primary treatment function remains intact.

2. CW designs and their functional variants

CWs are established according to specific guidelines to function as buffer zones, protecting aquatic ecosystems and nearby species from both point and non-point source pollution.[11] CWs for C sequestration are primarily classified into free-water-surface flow and subsurface-flow systems. A hybrid system can also be created by combining these two designs. The classification of these wetlands generally depends on the water flow path and the type of vegetation planted.[12] Floating treatment wetlands replicate natural floating islands using buoyant platforms planted with emergent vegetation.[13]

In surface flow CWs, water moves through open channels with diverse vegetation, where factors such as depth, plant types, and flow velocity influence C storage and treatment efficiency.[14] Subsurface flow CWs include horizontal and vertical systems. In horizontal subsurface flow wetlands, wastewater flows beneath the substrate through porous media, passing through O_2-rich and anoxic zones that enhance pollutant removal.[12] Vertical subsurface flow CWs use gravel or sand beds with intermittent batch feeding; "fill-and-drain" operations improve contact between wastewater and microbes, increasing purification efficiency.[1] Vertical subsurface flow CWs are widely applied for domestic, dairy, landfill leachate, and food processing wastewater, requiring less space than horizontal systems. Variants include vertical upflow and downflow configurations.[12,13] While effective, vertical systems face challenges such as clogging under high wastewater loads.

3. Wetland ecosystems as carbon sinks and sources

Wetlands serve as major hotspots for C sequestration. Water saturation creates anoxic conditions that slow decomposition and lead to the accumulation of total soil C. However, alterations in evapotranspiration and precipitation patterns can significantly influence C cycling in CWs.[15] C sequestration is further mediated by vegetation through photosynthesis, in which atmospheric carbon dioxide (CO_2) is converted into glucose and subsequently transformed into complex compounds such as cellulose and lignin, which are deposited in plant tissues. Thus, it is essential to understand the potential of C sequestration in CWs while simultaneously addressing C emissions.[4]

Some studies report that wetlands acquire 200 times more C in soils than in vegetation. C can accumulate rapidly as organic matter, while anoxic conditions slow decomposition, increasing C residence time and reducing C losses through mineralization. The potential of C sequestration through wetland restoration has been estimated at approximately 0.4 ton C/ha/year over a 50-year period, based on direct system measurements.[2] CH_4 emissions in CWs occur through three primary pathways: Molecular diffusion, ebullition, and plant-mediated transport through aerenchyma. Among these, plant-mediated release is the dominant process, accounting for approximately 70% of total CH_4 emissions. Ebullition contributes nearly three times more flux than diffusion. Plant transport occurs either by diffusion or by faster convective processes, which deliver O_2 to the roots while venting microbial byproducts to the atmosphere.[16] Figure 1 provides an overview of C sequestration in CWs.

During CH_4 transport in CWs, oxidation can occur through either anaerobic or aerobic pathways, influencing overall fluxes. Aerobic oxidation, carried out by methanotrophic bacteria, converts CH_4 into CO_2 at zones where CH_4 and O_2 overlap, including the water–air interface, substrate–air interface, rhizosphere, and internal plant tissues,[17] depending on O_2 availability.[18] This process is regarded as the principal CH_4 sink in wetland systems and is capable of oxidizing more than 50% of CH_4 production in CWs.[19]

Anaerobic oxidation of methane (AOM) refers to the microbial oxidation of CH_4 in the absence of oxygen, utilizing alternative electron acceptors such as sulfate (sulfate-reduction-dependent AOM), metal oxides like Fe^{3+} and Mn^{4+}, and nitrite or nitrate (nitrite-dependent AOM), as well as through direct interspecies electron transfer.[20,21] Although often overlooked, AOM is increasingly recognized as an important CH_4 sink. For instance, Guerrero-Cruz *et al.*[21] demonstrated that AOM reduced CH_4 emissions by more than 50% in freshwater wetlands. It is also estimated that wetlands are significant C sinks, sequestering around 830 Tg C/year.

Since CWs have been shown to sequester more C than natural wetlands, it is essential for researchers to investigate the factors influencing C storage in these systems, particularly those that can reduce CH_4 emissions. Ma *et al.*[22] demonstrated that CWs act as both C sources and sinks, thereby contributing to the global warming potential. For example, in most intact wetlands (about 67%) where CH_4 fluxes remain low, the initial surge of CH_4 released during wetland establishment is offset over time by continuous CO_2 uptake, resulting in a net CO_2-equivalent C sink. By contrast, intact wetlands with high CH_4 fluxes act as net C sources, although their cumulative source strength increases only modestly over a 500-year timescale.

Similarly, CH_4 emissions at the Lankheet CW varied with temperature and vegetation density, averaging 7.8 mg/m^2/h at 15°C and increasing to 24.5 mg/m^2/h at 24°C. When expressed as CO_2 equivalents, the wetland still functions as a net CO_2 sink under current conditions, with an annual sequestration rate of 0.27–2.4 kg/m^2/year. This represents 12–67% of the C fixed in plant biomass.[23]

To ensure maximum societal, environmental, and economic benefits, comprehensive management plans tailored to different CW types and functions should be developed, accounting for both their ecological services and cost-saving potential.

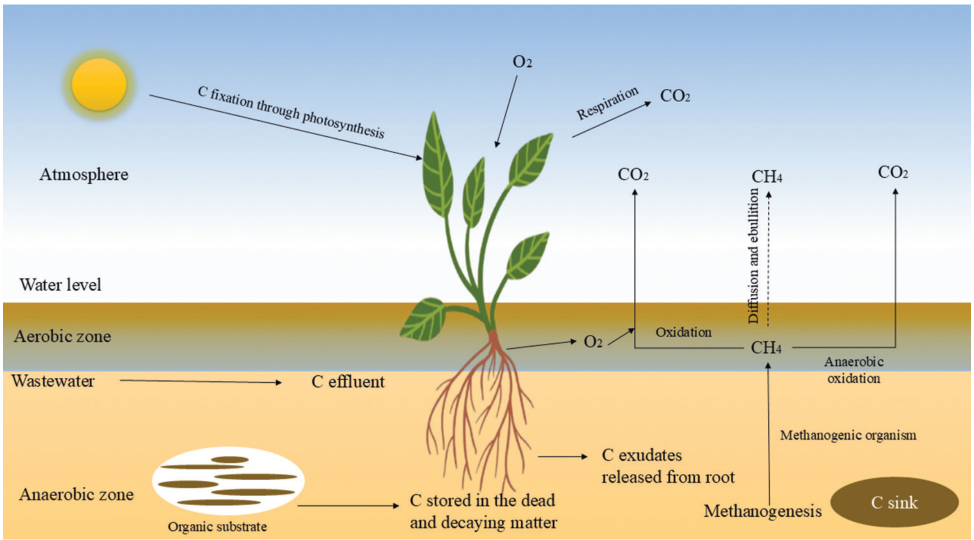

Figure 1. Overview of carbon (C) sequestration in constructed wetlands. Image created by authors using Microsoft powerpoint (version 2019).

4. Dynamics of CWs in wastewater treatment

CWs have emerged as a versatile, nature-based technology for wastewater treatment. Recent advances include hybrid configurations for enhanced N removal, targeted removal of specific contaminants, reactive media for phosphorus retention, characterization of microbial communities, and hydraulic modeling.[24] Growing demand for NH_4^+–N removal has accelerated the adoption of hybrid CWs, which demonstrate superior performance compared to single-unit systems. CWs have been applied not only to municipal sewage but also to industrial effluents from tanneries, wineries, composting facilities, and aquaculture. Researchers have highlighted the effectiveness of CWs in removing emerging pollutants, such as linear alkylbenzenesulfonates and pharmaceutical or personal care products, underscoring them as a sustainable, nature-based solution for diverse wastewater streams.[1]

For instance, Li *et al.*[25] reported that phytoremediation in CWs is regarded as an effective secondary wastewater treatment approach for pharmaceutical contaminants, particularly within decentralized systems. Their study examined rhizosphere dynamics and the associated microbial communities, demonstrating that the removal of pharmaceutical compounds such as ibuprofen occurs through the co-metabolism of root exudates, including lipids, fatty acids, amino acids, and organic acids. Furthermore, they established a positive correlation between rhizosphere microbial activity and COD, indicating that COD serves as a C source that supports microbial nutrition and enhances pollutant degradation.

Microbes, including cyanobacteria, algae, and heterotrophic microorganisms, are collectively referred to as periphyton and are located on the submerged surfaces of aquatic plants. They can account for the removal of up to approximately 90% of contaminants from wastewater; in turn, aquatic vegetation provides nutrients to the microbial community.[12] In addition, the selection of suitable bed material supports vegetation growth and biofilm formation, playing a crucial role in wastewater treatment.[12]

Zhang *et al.*[26] reported that CWs provide C sequestration and wastewater treatment simultaneously. For example, a system planted with *Trema orientalis* achieved high pollutant removal efficiencies, reducing NH_4^+–N by 89.07%, COD by 79.32%, total phosphorus by 66.33%, and total N by 98.31%. At the same time, C fixation was enhanced, with a sequestration capacity of 1,806.94 g C/m^2/year. GHG fluxes included CH_4 at 2.37 mg/m^2/h (14.35%), CO_2 at 362.69 mg/m^2/h (81.36% of global warming potential), and N_2O at 0.07 mg/m^2/h (4.29%). These findings highlight that nutrient removal from wastewater directly fuels plant growth and microbial activity, thereby enhancing C storage while simultaneously influencing net GHG emissions. This demonstrates that wastewater purification and C cycling are tightly coupled, with hydrophytes driving both pollutant removal and C sequestration.

Furthermore, CWs have been widely explored to decontaminate nitrogenous substances through processes such as denitrification, anaerobic ammonia oxidation (anammox), and dissimilatory nitrate reduction to ammonium (DNRA). The efficient removal of N is mediated by rhizospheric microbes, including anaerobic, aerobic, and anoxic bacteria.[27] Nitrification occurs near the rhizosphere due to radial oxygen loss mediated by root activities, while denitrification is facilitated in the non-rhizosphere zone under anaerobic conditions. Recent evidence suggests that DNRA and anammox processes are hindered in the rhizosphere. This inhibition is linked to the presence of humic acid-like substances in root exudates and the accumulation of hydroxylamine during ammonia oxidation, which interferes with anammox activity and the expression of key functional genes.

In addition, the disruption of the symbiotic relationship between anammox and DNRA microorganisms further contributes to reduced N cycling efficiency in the rhizosphere.[27] Moreover, wastewater management through CWs facilitates nutrient recovery, promotes the reuse of treated effluent, and enables the extraction of valuable byproducts; it is also economically sustainable compared to conventional treatment infrastructure, thereby advancing the principles of the circular bioeconomy and supporting urban irrigation[24] (Figure 2). The key findings are summarized in Table 1, which presents selected studies demonstrating the role of CWs in wastewater treatment and C sequestration.

5. Challenges and future scope

Advanced CWs are widely recognized as reliable technologies for treating a broad range of wastewater types. However, limitations remain, particularly the relatively low efficiency of phosphorus removal, which must be considered when selecting treatment technologies. Earlier concerns regarding their safety and dependability have largely been dispelled. In addition, studies have demonstrated that CWs perform efficiently even under cold climatic conditions, and their relatively modest land requirements make them suitable for use in densely populated countries such as Denmark and the Netherlands.[38] CWs are also increasingly harnessed for their ability to mitigate non-point source pollution from agricultural runoff and livestock effluents,

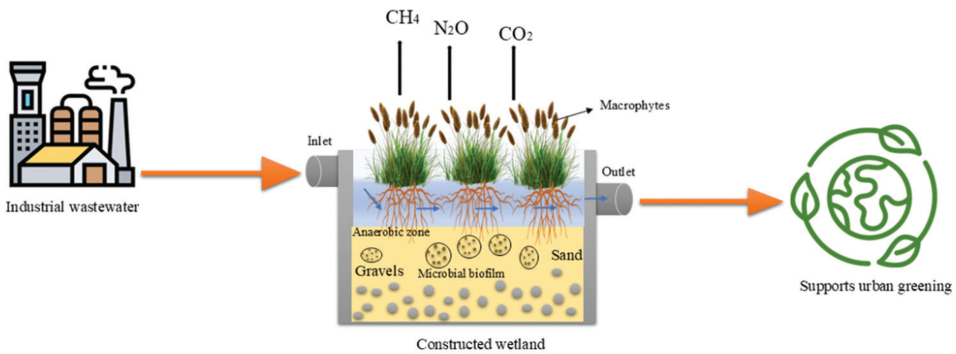

Figure 2. Representation of wastewater treatment through constructed wetlands. Image created by the authors by Microsoft power point (version 2019).

Table 1. Key findings on constructed wetlands for wastewater treatment and carbon sequestration

Key finding (s)	References
Restored inundated wetlands function as potential carbon sinks; however, their sequestration efficiency is influenced by restoration design, operational practices, and vegetation cover, with implications for climate change mitigation.	28
Coastal wetlands have a strong potential for carbon sequestration, but their efficiency is influenced by climate change, nitrogen inputs, vegetation management, and conservation practices.	29
The Zoige alpine wetland functions as a carbon sink; however, degradation alters environmental factors, reducing its sequestration efficiency.	30
Wetlands in West Bengal act as stronger carbon sinks than upland sites, storing 48.53–143.17 Mg C/ha in soils, with sequestration potential varying by wetland type and positively correlated with macrophyte coverage.	31
Montane fen wetlands in Korea act as significant carbon sinks (58.29–125.31 g C/m^2/y; 14.13–138.00 t C), with sequestration strongly influenced by dominant plant species (notably *Sphagnum palustre*) and reduced by disturbance, underscoring the importance of vegetation composition and conservation in wetland restoration.	32
In the Hangzhou Bay coastal wetlands, long-term reclamation (>1,000 years) increases surface soil carbon storage, with agricultural soils exhibiting the highest organic carbon content due to fertilization. However, SOC declines with depth, highlighting the importance of careful land management during early reclamation stages to minimize carbon loss.	33
CWs vegetated with *Pistia stratiotes* achieved significant pollutant removal (BOD=82%, total dissolved solids=83%, COD=81%, chloride=80%, sulfate=77%, TSS=82%, oil and grease=74%, and NH_3=84%), improved pH by 11.9%, and enhanced color and odor, with optimum treatment at 30 day hydraulic retention time. The effluent met national standards for irrigation use, and the system proved economical with minimal maintenance.	34
A pilot-scale CW-microbial fuel cell system using *Typha angustifolia* achieved high COD removal (97.56% in fedbatch; 82.8% in continuous mode) and enhanced bioelectricity generation (up to 58.55 mW/m^2, 229.6–283.3 mA/m^2) through auxiliary terracotta-based separator–electrode assemblies, demonstrating improved energetics and wastewater treatment efficiency.	35
Long-term monitoring of nature-based solutions in Italy showed that CWs and lagoon systems can improve reclaimed water quality for agricultural reuse, with CWs more effective at nitrogen removal (3.4 mg/L) and phosphorus removal (0.4 mg/L). However, further reductions in *Escherichia coli* (~100 CFU/100 mL), BOD (<25 mg/L), and TSS (up to 40 mg/L in the lagoon system) are needed to meet EU 2020/741 class A standards.	36
The novel biological-CW–microalgal wastewater process (sequencing batch reactor+CW+microalgal photobioreactor) achieved high PAH removal (90.58–97.50%), with adsorption dominating the removal of high-molecular-weight PAHs and microbial degradation increasing for lower-molecular-weight PAHs. Although toxic substituted PAHs formed during degradation, the microalgal unit removed 75.37–88.52% PAHs and 99.56–100% substituted PAHs via cytochrome P_{450} activity, reducing bacterial toxicity by 90.93% and genotoxicity by 93.08%. This highlights the critical role of microalgae in water security and the need to address the ecotoxicity of degradation byproducts.	37

Abbreviations: BOD: Biochemical oxygen demand; CFU: Colony-forming unit; COD: Chemical oxygen demand; CW: Constructed wetland; PAH: Polycyclic aromatic hydrocarbon; SOC: Soil organic carbon; TSS: Total soluble solid.

even under challenging conditions such as high altitudes. However, they still encounter significant challenges related to variable pollutant loads, climatic fluctuations, and long-term operational stability, highlighting the need for future designs that optimize treatment performance while ensuring environmental sustainability. In addition, tailored management strategies for different types and functions of CWs should be formulated to maximize their societal, environmental, and economic benefits, while accounting for the ecological services and cost savings they can provide.

6. Conclusion

CWs are engineered systems that can achieve relatively low net C emissions while enhancing C sequestration, thereby reducing the C footprint and mitigating GHG emissions. In addition, they improve water quality by removing contaminants such as pharmaceuticals, antiseptics, and organic matter. Future designs must address current challenges, particularly phosphorus removal efficiency, resilience to climate change, and long-term operational stability. Optimizing hydrophyte selection and microbial communities will be critical to simultaneously enhance pollutant removal and C sequestration while minimizing CH_4 emissions. By integrating these approaches, CWs can provide resilient ecosystem services, improve water security, and safeguard communities when their deployment is supported by policies that tackle global climate change.

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Conflict of interest

The authors declare that they have no competing interests.

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Consent for publication

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