Constructed floating wetlands cut greenhouse gas emissions from wastewater lagoons

Lukas Schuster[1,2*], Peter I. Macreadie[1], John Awad[3], Divina Navarro[3], Martino E. Malerba[1,2]

[1] Centre for Nature Positive Solutions, School of Science, RMIT University, Melbourne, Victoria 3000, Australia

[2] School of Life and Environmental Sciences, Deakin University, Burwood, Victoria 3125, Australia

[3] Commonwealth Scientific and Industrial Research Organisation, Environment, Waite Campus, Urrbrae, South Australia 5064, Australia

* Corresponding author:

Email: lukas.schuster@rmit.edu.au

Mailing address: RMIT University, 124 La Trobe Street, Melbourne VIC 3000, Australia

Abstract

Wastewater treatment is a significant, yet often overlooked, contributor to global greenhouse gas emissions, accounting for ~1.6% of anthropogenic emissions (~0.77 Gt CO_2-equivalent per year), including 7-10% of methane (CH_4) and nitrous oxide emissions (N_2O). However, scalable mitigation options remain scarce. Constructed floating wetlands (CFWs) are widely used to reduce nutrient loads in wastewater systems, but their potential to reduce greenhouse gas emissions at full operational scales remains unclear. We conducted a two-year, full-scale trial of a CFW in a wastewater holding lagoon in southeastern Australia. The lagoon was divided into paired treatment (with the CFW) and control channels using baffle curtains. At the start and end of each channel, we monitored greenhouse gas emissions continuously and water quality monthly. On average, carbon dioxide (CO_2)-equivalent emissions in the treatment channel were 22-31% lower than in the control, primarily driven by reduced CH_4 emissions (32-66%), with additional reductions in CO_2 (24-36%) and N_2O emissions (18%). Total Kjeldahl nitrogen levels (primarily organic nitrogen) declined by 12%, whereas nitrate and phosphorus remained unaffected. Notably, emission reductions emerged within four to seven months of CFW installation, preceding detectable nutrient reductions, which only became evident after 12 months. Our findings provide the first full-scale evidence that CFWs can substantially reduce greenhouse gas emissions from wastewater lagoons and may offer a promising nature-based pathway to support net-zero targets in the water sector. Nevertheless, wide-scale adoption will likely depend on further validation across treatment systems and configurations, as well as on addressing upfront costs and regulatory barriers.

Keywords: Wastewater treatment plants; wastewater lagoons; emission reduction; nature-based solutions; sustainable development goals; environmental engineering

Introduction

Wastewater treatment plants are an important source of greenhouse gas emissions, releasing large amounts of carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) during treatment processes (Nguyen et al., 2019). Globally, wastewater treatment accounts for ~1.6% of total anthropogenic emissions, which equates to 0.77 Gt (0.77×10^9 tonnes) CO_2-equivalent per year (IPCC, 2014). Considering CH_4 and N_2O emissions alone, wastewater treatment accounts for an estimated 7-10% of global anthropogenic emissions (Maktabifard et al., 2023). These non-CO_2 greenhouse gases are of particular concern because their global warming potentials (GWPs) are 27 (CH_4) and 273 (N_2O) times greater than CO_2 on a 100-year timescale (IPCC, 2021). As a result, wastewater treatment plants are considered hotspots for high-impact emissions.

In recent decades, emissions from the water sector have increased substantially. According to IPCC inventory estimates, CH_4 emissions increased by ~50% between 1990 and 2020, with N_2O emissions increasing by ~25% (IPCC, 2007). This increase in emissions has been largely driven by rapid urbanisation and infrastructure expansion associated with economic and population growth (U.S. Environmental Protection Agency, 2012). The sector's growing climate footprint has prompted widespread decarbonisation pledges, with several countries, including Australia, members of the European Union, and China, committing to net-zero targets by 2050-2060 (Maktabifard et al., 2023). However, meeting these targets requires innovative, evidence-based strategies to deliver scalable emission reductions across the global water sector.

Wastewater is characterised by high nutrient concentrations, particularly nitrogen and phosphorus (Carey & Migliaccio, 2009). Treatment processes largely rely on microbial breakdown to remove these nutrients, a process that also generates most of the sector's direct

("Scope 1") emissions (Wu et al., 2022). Specifically, microbial respiration of organic matter produces CO_2, methanogenesis produces CH_4, and nitrification and denitrification both produce N_2O (Li et al., 2021; Zhu et al., 2025). During enclosed treatment stages, some of these emissions can be mitigated through capture and utilisation. For example, CH_4 that is produced during anaerobic digestion can be captured and used for flaring or energy recovery (Campos et al., 2016). In contrast, emissions from open lagoons cannot be mitigated as easily. After primary treatment, wastewater effluent is typically stored in large, open holding lagoons that are used for flow equalisation, seasonal storage, and additional settling to remove residual suspended solids before discharge into the ocean, reuse for irrigation, or further treatment (Shammas et al., 2009). Nevertheless, treated effluent typically retains substantial amounts of nutrients and organic matter, which can fuel further microbial activity and associated greenhouse gas production (Carey & Migliaccio, 2009). In particular, nutrient-rich inflows, long hydrological retention times, and the development of anaerobic conditions make wastewater holding lagoons substantial, yet often under-recognised, greenhouse gas emission hotspots (Ho et al., 2021; Song et al., 2023). Tackling emissions from these open lagoons by targeting nutrient reductions could therefore deliver some of the most immediate and cost-effective gains toward net-zero goals in the water sector, while simultaneously supporting compliance with increasingly stringent nutrient discharge regulations (Preisner et al., 2020).

Constructed floating wetlands (CFWs) are a promising nature-based solution for reducing nutrient loadings in open-water systems, including wastewater holding lagoons. CFWs consist of buoyant platforms that support the growth of emergent wetland plants, with plant roots growing directly into the water column (Pavlineri et al., 2017). The extensive root network provides ample surface area for microbial biofilms to establish, thereby facilitating nutrient uptake by plants, trapping of suspended particles, and biofilm-mediated breakdown and transformation of organic materials and other pollutants (Nichols et al., 2016; Shahid et al., 2020). To date, CFWs have been used in a wide range of settings, including eutrophic lakes, wastewater effluents, agricultural and stormwater runoffs, among other applications (reviewed in Shahid et al., 2018). Across these systems, reported average nutrient removal efficiencies ranged from 49% for total phosphorus to 58% for total nitrogen and 73% for ammonium, although performance is strongly dependent on vegetation coverage, a lagoon's hydrological retention times, and nutrient concentrations in the influent (Pavlineri et al., 2017). In a wastewater lagoon specifically, Benvenuti et al. (2018) reported nitrogen and phosphorus reductions of 41% and 37%, respectively, within 12 months of deploying a full-surface CFW. Crucially, reducing nutrient levels using CFWs may have a strong co-benefit for climate change mitigation by limiting microbial greenhouse gas production, including microbial respiration, methanogenesis, nitrification, and denitrification (Audet et al., 2024; Li et al., 2021). For instance, a 25% reduction in nitrate concentrations has been shown to cut CO_2-equivalent emissions from agricultural ponds, on average, by 54% (Ollivier et al., 2018). Yet, despite the wide use of CFWs for nutrient management and the clear link between nutrient availability and greenhouse gas emissions, the potential of CFWs to directly reduce emissions at full operational scales has not yet been investigated.

In this study, we evaluated the performance of a CFW in reducing nutrient concentrations and greenhouse gas emissions from a wastewater holding lagoon in southeastern Australia. We used baffle curtains to create two channels inside the lagoon: one containing a CFW and the other serving as an unmodified control (Fig. 1). This channelised, within-lagoon design provides a direct, side-by-side comparison that attributes differences in greenhouse gas emissions to the presence of the CFW under shared hydrological and climatic conditions, rather than inferring treatment effects from whole-lagoon temporal comparisons. Over a two-year monitoring period, we measured nutrient concentrations and greenhouse gas fluxes, including CO_2, CH_4, and N_2O, to determine: (i) the effectiveness of CFWs in mitigating dissolved nutrient loads and greenhouse gas emissions in open wastewater lagoons, (ii) the timeframe required to achieve measurable nutrient and emission reductions, and (iii) the relationships between nutrients, other water quality parameters, and greenhouse gas emissions to infer the mechanisms by which the CFW influences greenhouse gas dynamics. To support decision-making and cost planning for water utilities considering the installation of CFWs in wastewater lagoons, we also quantified the capital (CAPEX) and operational (OPEX) expenditures associated with the installation, monitoring, and maintenance of the CFW system over the two-year study period. Altogether, our study provides one of the first comprehensive, full-scale assessments of CFW performance for nutrient and greenhouse gas emission mitigation in commercial wastewater lagoons, elucidating the magnitude and timing of observed CFW effects.

Material and methods

Site description

We conducted our study at the effluent holding lagoon of the Westernport Water wastewater treatment plant, located on Phillip Island, Victoria, Australia (38° 29' 13.7" S, 145° 13' 12.3 "E) between April 2023 and April 2025. The holding lagoon is used to store class C treated wastewater that is characterised by high nutrient levels (particularly nitrogen and phosphorus) and is used for agricultural irrigation, discharged into the ocean, or further filtered and treated for other reuse purposes. The lagoon is 4,500 m^2 in size and can hold up to 14 ML (million litres) of treated wastewater. Across our monitoring period, water levels in the lagoon ranged between 1.08 and 3.22 m (2.1 ± 0.01 m). The average water inflow rate into the lagoon was 45.3 ± 0.19 L sec^{-1}, with hydraulic retention times ranging between 1 and 2.3 days (1.66 ± 0.06 days). The study region is characterised by a temperate Mediterranean climate, with long-term (1981-2017) average monthly temperatures ranging from 10°C in winter to 18°C in summer, and an average yearly precipitation of 738 mm (Bureau of Meteorology).

Experimental design

To assess the efficacy of a CFW in reducing nutrient levels and greenhouse gas emissions, we relied on a paired-channel design, where the lagoon was subdivided into two 9.4 m wide channels using three 40 m-long vinyl baffle curtains in March 2023: a treatment channel with the CFW and an unmodified control channel (no CFW). The baffle curtains were designed to minimise mixing and water exchange between channels by weighing down the bottom with heavy chains, whereas the top was filled with a 90 mm polyvinyl chloride (PVC) stormwater pipe and fitted with polyethylene foam floats ('pool noodles') to keep it afloat. All baffle curtains were tensioned on both sides of the lagoon to keep the zones separate and securely anchored. To ensure that both channels received similar amounts of water, a splitter box was installed at the lagoon inlet that equally distributed the inflow water between the control and treatment channels (Fig. 1). Because the channels were located at the lagoon inlet with continuous flow-through, their hydrological retention time was much shorter than that of the lagoon (9.68 h vs. 1.66 days), reflecting their smaller average water volumes (788,096 ± 1,969 L) combined with high inflow rates (45.3 ± 0.19 L sec^{-1}). Note that despite secure and tensioned installation of the baffle curtains, some limited water exchange between channels may have occurred, particularly near the lagoon inlet. However, any such mixing would likely have diluted rather than exaggerated the observed CFW effects. Consequently, any reported treatment effects are likely conservative estimates.

Prior to the installation of the CFW, we conducted baseline monitoring to quantify pre-existing differences in greenhouse gas fluxes and water quality parameters between the treatment and control channels. Specifically, baseline monitoring ensured that any observed post-installation changes could be attributed to the CFW rather than inherent channel variation. To do so, we collected one-time grab samples approximately 10 cm below the water surface at the inlet (start of the channel) and outlet (end of the channel) of each channel (2 channels × 2 sampling points × 1 sample each = 4 samples) for water quality analyses in April 2023. We also deployed two floating greenhouse gas sensors ('Pondi', see below) at the start and end of each channel (2 channels × 2 sampling points × 2 sensors each = 8 sensors), taking hourly readings for one month to determine baseline CO_2, CH_4, and N_2O fluxes (April-May 2023). We conducted greenhouse gas flux monitoring and water quality sampling at the start and end of each channel to capture the net effect of the CFW on water quality and emissions. This inflow-outflow approach aligns with conventional wastewater treatment assessments, where treatment performance is evaluated based on changes between the entry and exit points, rather than internal spatial variability (Lucke et al., 2019).

Following baseline monitoring, the CFW was installed by Clarity Aquatic (Queensland, Australia) in mid-May 2023, and we continued to monitor water quality and greenhouse gas fluxes at the start and end of the channels monthly. The CFW consisted of 60 modules (2.35 m × 2.35 m) and was 331 m^2 in size, covering approximately 7.4% of the lagoon's water surface. Each module contained 15 interchangeable polypropylene baskets (57 cm × 38 cm × 17 cm) that were half-filled with aggregate and planted with two plants of the same species. The CFW was initially planted with two native wetland plant species, *Baumea articulata* and *Phragmites australis*, with each species occupying half of the CFW (30 modules per species × 15 baskets × 2 plants per basket × 2 species = 1,800 plants). However, approximately half of the *B. articulata* plants failed to establish within the first year and were, therefore, replaced with *Bolboschoenus caldwellii* in May 2024. Following this replanting, the CFW was comprised of 50% *P. australis*, 25% *B. articulata*, and 25% *B. caldwellii*. Across the monitoring period, the CFW was maintained through weeding in November 2023 (removal of non-target plant species) and plant harvesting by trimming the shoots in September 2024 (*P. australis* and weeding) and March 2025 (all wetland plant species; Table S1). These periodic harvesting events were implemented to promote regrowth to maximise nutrient uptake from the wastewater. All harvested plant material was disposed of by burning.

Altogether, we monitored water quality and greenhouse gas fluxes for 21 months across the two-year monitoring period, with four months (October 2023 and June to September 2024) omitted due to weather-related access limitations and temporary logistical interruptions.

Water quality parameters

For water quality testing, we collected 500 mL grab samples approximately 10 cm below the water surface at the start and end of both channels once a month across the monitoring period (4 samples per month × 21 months = 84 samples). We stored all samples at 4°C overnight and sent them to Nutrient Advantage Laboratory (Victoria, Australia) the next day for the quantification of nitrate nitrogen (NO_3^- in mg L^{-1}; APHA method 4500-NO3-I), ammonium (NH_4^+ in mg L^{-1}; APHA method 4500-NH3-H), total Kjeldahl nitrogen (in mg L^{-1}; s-TKN™ method), which includes ammonium and organic nitrogen contents, and total phosphorus (in mg L^{-1}; APHA methods 3120 B; 3030 B) levels in the surface water samples. Note that ammonium concentrations were below the detection limit (<1 mg L^{-1}) at all sampling locations and monitoring time points. As a result, the total Kjeldahl nitrogen values reported here primarily reflect organic nitrogen concentrations. We also measured dissolved oxygen concentration (in %), pH, and temperature (in °C) approximately 10 cm below the water surface using a Hanna HI98194 portable multi-meter at each sampling point.

Greenhouse gas flux measurements

We used ‘Pondi’ loggers to quantify greenhouse gas fluxes at the start and end of the treatment and control channels (Malerba et al., 2025). Pondi loggers consisted of a 16 L floating chamber (45 cm in diameter × 15.5 cm in height) fitted with a Sensirion SCD40 sensor to quantify CO_2 levels within the gas collection chamber (measurement range: 0-40,000 ppm), a Figaro TGS2611-E00 sensor for CH_4 (measurement range: 0-10,000 ppm), and a Dynament Platinum P/N2OP/NC/4/P sensor for N_2O (measurement range: 0-1,000 ppm). Floating chambers can capture constant fluxes (diffusion) and stochastic releases of gas bubbles (ebullition), which are particularly relevant for CH_4 fluxes. Thus, both flux types were included in our greenhouse gas flux estimates.

Each Pondi was powered by two solar panels and battery cells and used Telstra's Cat-M1 network to transfer data to a cloud in real-time. To mitigate gas saturation effects, each Pondi was equipped with an external air pump configured to automatically vent the chamber headspace once per week by injecting ambient air. This periodic venting ensured internal gas concentrations were reset to ambient levels, preventing saturation-induced suppression of fluxes. Field trials indicated that CH_4 levels do not typically saturate for several weeks, whereas CO_2 and N_2O gases typically reach saturation within one to three days (Fig. S1). Consequently, we selected weekly venting as a practical compromise across all gases, while also conserving battery life. Finally, we checked all weekly gas flux trajectories manually and removed any segments showing signs of saturation (i.e., lack of a linear increase in gas concentration over time) from flux calculations.

To minimise sensor drift and biofouling, we retrieved all Pondi loggers every four weeks and replaced them with a newly calibrated set (refer to Malerba et al., 2025 for details on sensor calibrations). At each sampling location, we deployed two Pondi units (8 loggers in total) and monitored greenhouse gas concentrations within the chamber hourly. In the treatment channel, we tethered the Pondi to the CFW using ropes. In the control channel, we used a rope and pulley system to align the sensor placement with the corresponding locations in the treatment channel, thus ensuring spatial comparability (Fig. 1).

We estimated greenhouse gas fluxes from the water surface to the atmosphere (F; g m^{-2} day^{-1}) as:

$$F = \frac{slope \times V \times P \times M \times F_1}{A \times R \times T \times F_2} \qquad (1)$$

where *slope* is the linear rate of change in gas concentration over time within the chamber (ppm min^{-1}), V is the chamber volume (0.01309 m^3), P is the average atmospheric pressure in Pascals within the chamber during the measurement period, M is the molar mass (44.01 g mol^{-1} for CO_2, 16.04 g mol^{-1} for CH_4, and 44.013 g mol^{-1} for N_2O), F_1 is the conversion factor from minutes to days (1,440), A is the surface area of the chamber (0.1282 m^2), which is equivalent to the water surface area, from which gas fluxes were measured, R is the ideal gas constant (8.314 J mol^{-1} K^{-1}), T is the average temperature in Kelvin within the chamber, and F_2 is the conversion factor to convert ppm (µmol mol^{-1}) to mol mol^{-1} (1,000,000).

To calculate CO_2-equivalent fluxes, we multiplied CH_4 and N_2O fluxes by their respective 100-year GWPs (27 for CH_4 and 273 for N_2O) and then summed them with the measured CO_2 fluxes (IPCC, 2021).

Statistical analyses

We used linear models to test for the effects of channel (control or treatment channel) and sampling location (start or end of channel) on baseline (T0; before the CFW was installed) greenhouse gas fluxes (CO_2, CH_4, and N_2O) and water quality parameters (total Kjeldahl nitrogen, nitrate, phosphorus, surface water temperature, dissolved oxygen, and pH).

To test for the effects of channel and sampling location on greenhouse gas fluxes and water quality parameters after the CFW was installed, we used linear mixed effects models and included time point (T1 to T19) as a random effect in all models to account for repeated measures. We included sampling year as a fixed effect in all models to differentiate between the establishment phase (year 1; May 2023-April 2024) and the mature phase, when most plants on the CFW were established (year 2; May 2024-April 2025).

We used linear mixed-effects models with time point as a random effect to test for the effects of total nitrogen (the sum of total Kjeldahl nitrogen and nitrate concentrations), dissolved oxygen, channel, and sampling location on greenhouse gas fluxes. To avoid bias due to multicollinearity among environmental predictors, we calculated variance inflation factors (VIFs) and selected a cut-off value of 5 (Zuur et al., 2009). We found that including total Kjeldahl nitrogen, nitrate levels, and phosphorus resulted in VIFs > 5 for these predictors. As a result, we calculated total nitrogen concentrations by summing total Kjeldahl nitrogen and nitrate concentrations to estimate their combined effects on greenhouse gas fluxes and removed phosphorus from the final model. Similarly, including pH and water surface temperature in our model resulted in VIFs > 5 for these predictors. As a result, we only included dissolved oxygen and total nitrogen in the final model, given their importance in modulating greenhouse gas fluxes (Huang et al., 2020; Li et al., 2021). When modelling CH_4 fluxes, we excluded spikes observed after plant harvesting (at T6, T13, and T19; see Results), since these events were most likely attributable to harvesting disturbance rather than water quality dynamics. We centred and scaled all predictors before fitting linear mixed-effects models and identified the best-fitting model using Akaike information criteria corrected for small sample sizes (AICc; Burnham & Anderson, 2004).

We performed all analyses and visualisations in R v4.5.0 (R Core Team, 2013), using the *nlme* v.3.1-168 (Pinheiro et al., 2024) and *ggplot2* v.3.5.2 (Wickham, 2016) packages. When standardised residuals showed unequal variances or systematic trends, we included sampling year-, channel- and/or sampling location-specific variance coefficients in the model (function “varIdent”). Errors reported throughout are standard errors.

Results

Baseline water quality and greenhouse gas fluxes

Prior to the installation of the CFW, all water quality parameters, including total Kjeldahl nitrogen, nitrate, phosphorus, surface water temperature, dissolved oxygen, and pH, were comparable (<6.7% difference) between channels at the respective sampling locations (start and end of the channels) (Fig. S2). Similarly, baseline CO_2, CH_4, and N_2O fluxes did not significantly differ between channels (CO_2: $P = 0.58$; CH_4: $P = 0.47$; N_2O: $P = 0.57$; Fig. S3, Table S2).

Effects of the CFW on dissolved nutrient levels and other water quality parameters

After the CFW was installed in the treatment channel, we continued monitoring water quality in both channels monthly. Although nutrient levels fluctuated over time, we found that total Kjeldahl nitrogen levels started to trend lower at the end of the treatment channel 12 months after the installation of the CFW (time point T11; Fig. 2). When considering the full monitoring period, we found that overall total Kjeldahl nitrogen levels were, on average, 12% lower at the end of the treatment compared to the end of the control channel (channel × sampling location: $P = 0.04$; Fig. 3, Table S3). Contrastingly, the CFW had no detectable effect on nitrate or phosphorus levels.

Other water quality parameters, including surface water temperature and pH, did not significantly differ between channels or sampling locations (Fig. S4, Table S4). In contrast, overall dissolved oxygen concentrations were, on average, 9% higher at the end of the treatment than in the control channel (79.3 vs. 72.9%; channel × sampling location: $P = 0.02$; Fig. S4, Table S4).

Effects of the CFW on greenhouse gas emissions

We next evaluated the effects of the CFW on CO_2, CH_4, and N_2O emissions. CH_4 fluxes in the treatment channel began to decrease relative to the control four months after CFW deployment (T4), whereas reductions in CO_2 and N_2O fluxes became apparent seven months after the CFW was installed (T7; Fig. 4). Nevertheless, these differences in greenhouse gas emissions between channels were not always apparent across the monitoring period (Fig. 4). For example, CO_2 fluxes in the treatment channel were lower relative to the control channel at 12 out of the 19 monitoring time points. Similarly, CH_4 and N_2O fluxes in the treatment channel were lower at eight and seven time points, respectively. Notably, we found that plant harvesting induced sharp but short-lived spikes in CH_4 fluxes in the treatment channel. Specifically, CH_4 emissions at the end of the treatment channel reached 3.3-12.5 times those measured in the control channel after harvesting in November 2023, September 2024, and March 2025. Nevertheless, CH_4 emissions returned to pre-harvest levels within one month (Fig. 4).

When averaged across the entire monitoring period, greenhouse gas emissions were significantly lower in the treatment than in the control channel (Table S5). On average, CO_2 fluxes in the treatment channel were 24% lower than in the control channel during the first sampling year (T1-T10; 1.17 vs. 1.53 g CO_2 m^{-2} day^{-1}) and 36% lower during the second year (T11-T19; 1.17 vs. 1.82 g CO_2 m^{-2} day^{-1}; sampling year × channel: $P = 0.04$). CH_4 fluxes were also lower in the treatment channel, but the magnitude of reduction depended on the sampling location (channel × sampling location: $P = 0.01$). Specifically, average CH_4 fluxes at the start of the treatment channel were 66% lower than at the start of the control channel (0.004 vs. 0.01 g CH_4 m^{-2} day^{-1}), whereas fluxes at the end of the treatment channel were

32% lower than at the end of the control channel (0.02 vs. 0.03 g CH_4 m^{-2} day^{-1}). When harvesting-related CH_4 spikes were excluded, overall CH_4 emission reductions in the treatment channel strengthened to 60% compared to a 32% reduction when all measurement time points were included. Average N_2O fluxes were 18% lower in the treatment channel across both sampling years (0.016 vs. 0.02 g N_2O m^{-2} day^{-1}, $P = 0.07$; Fig. 5, Table S5).

In terms of CO_2-equivalent fluxes, the combined emissions of all three greenhouse gases across the monitoring period were, on average, 22% lower in the treatment than the control channel (6.79 vs. 8.74 g CO_2-equivalent m^{-2} day^{-1}, $P = 0.002$). When excluding harvesting-related CH_4 spikes, average CO_2-equivalent fluxes from the treatment channel were 31% lower than from the control channel (6.51 vs. 9.39 g CO_2-equivalent m^{-2} day^{-1}, $P < 0.0001$; Table S5).

Drivers of greenhouse gas emissions in wastewater

We next assessed the effects of total nitrogen (total Kjeldahl nitrogen + nitrate) and dissolved oxygen on greenhouse gas fluxes to identify key drivers of emissions. We found that CO_2 fluxes were significantly influenced by both total nitrogen and dissolved oxygen, but the effects varied between channels. In the control channel, CO_2 fluxes increased with increasing total nitrogen concentrations (channel × total nitrogen: $P = 0.01$) but tended to decrease with increasing dissolved oxygen levels (channel × dissolved oxygen: $P = 0.09$). Contrastingly, CO_2 fluxes in the treatment channel were not significantly affected by either total nitrogen or dissolved oxygen concentrations (Fig. 6, Table S6). Neither CH_4, N_2O, nor CO_2-equivalent fluxes were significantly affected by total nitrogen or dissolved oxygen levels in either the control or treatment channel (Table S6).

Costs associated with the installation, monitoring, and maintenance of the CFW

To support decision-making and cost planning for utilities and water managers considering the deployment of CFWs in wastewater lagoons for nutrient and greenhouse gas emission mitigation, we provide a detailed breakdown of the capital (CAPEX) and operational (OPEX) expenditures associated with the installation, monitoring, and maintenance of the CFW over the two-year study period (Table 1). The total project cost, based on actual expenditures and supplier quotes, was AU$360,016 (US$234,010). The majority of costs (~75%) was attributed to CAPEX, including the design, construction, and installation of the CFW, as well as greenhouse gas flux equipment (Pondi loggers) and water quality assessments. The remaining OPEX (~25%) covered labour for CFW maintenance, plant harvesting, and data analysis and reporting of flux and water quality data. When expressed relative to CFW area (331 m^2) and monitoring duration, CAPEX equated to AU$873 m^{-2} (US$567 m^{-2}), and OPEX to AU$107 m^{-2} $year^{-1}$ (US$70 m^{-2} $year^{-1}$).

Discussion

Wastewater treatment plants are significant emitters of anthropogenic greenhouse gases to the atmosphere (IPCC, 2014). With widespread decarbonisation pledges across the global water sector, innovative and evidence-based strategies to deliver scalable emission reductions are urgently needed to meet net-zero targets. Using a channelised experimental design, our findings demonstrate that installing a CFW in a wastewater holding lagoon significantly reduced CO_2, CH_4, and N_2O emissions, with emission reductions becoming evident within four to seven months after the installation of the CFW. Considering the sum of all three greenhouse gases, average CO_2-equivalent emissions in the treatment channel (with the CFW) were 22-31% lower than in the control channel, with the lower value reflecting the effects of short-term CH_4 spikes after plant harvesting on overall emission reductions. We also observed significant reductions in organic nitrogen levels (total Kjeldahl nitrogen) and increased dissolved oxygen concentrations at the end of the treatment channel (after the CFW), whereas nitrate and phosphorus levels were not affected by the CFW. Finally, CO_2 fluxes were positively associated with total nitrogen (total Kjeldahl nitrogen + nitrate) and tended to decline with increasing dissolved oxygen levels, whereas CH_4 and N_2O fluxes showed no significant relationships with water quality. Altogether, our results highlight the potential of CFWs as a promising nature-based solution to mitigate greenhouse gas emissions from wastewater lagoons and provide an evidence base to evaluate their cost-effectiveness and practicality for wider adoption.

The efficacy of CFWs in reducing nutrient concentrations in wastewater lagoons is strongly dependent on hydrological retention times, which control the contact time between water and plant roots and microbial biofilms (Pavlineri et al., 2017). In systems with low hydrological retention times, such as in our study (9.68 h within the channels), nutrient removal efficiency tends to be limited, since inflows can rapidly transport nitrogen and phosphorus through the lagoon before substantial uptake or transformation occurs. We observed detectable reductions in organic nitrogen levels only after 12 months, yielding an overall reduction of 12% across our monitoring period. Contrastingly, phosphorus and nitrate levels remained unaffected, indicating relatively modest total nutrient removal under our experimental conditions. To maximise nutrient removal, deploying CFWs in wastewater lagoons with longer hydrological retention times may thus lead to earlier and stronger nutrient reductions, which, in turn, could also enhance their capacity to mitigate greenhouse gas emissions by limiting nutrient-driven production pathways. Alternatively, expanding CFW coverage across a larger lagoon surface area may achieve similar improvements by increasing the scale of plant and microbial interactions with the water column, although such modifications may have operational implications (Pavlineri et al., 2017).

In aquatic systems, net greenhouse gas emissions are governed by microbial production and consumption as well as physical gas transfer processes (Bastviken, 2009; Hallin et al., 2018; Raymond et al., 2012). Given the significant reductions in CO_2, CH_4, and N_2O emissions even in the absence of strong CFW effects on dissolved nutrient levels, reduced substrate availability was unlikely to be the primary driver of these reductions, at least during early stages. Instead, biological consumption and physical mechanisms likely contributed to the observed mitigation. By creating extensive root and biofilm habitats, CFWs can support diverse microbial assemblages, including algae, methanotrophs and N_2O reducers, which play a critical role in regulating net greenhouse gas emissions from aquatic ecosystems (Huang et al., 2025; Tranvik et al., 2009; Wang et al., 2024). These root systems can also trap and extend the residence time of dissolved gases or rising bubbles (e.g., the ebullitive release of CH_4), thereby enhancing opportunities for microbial oxidation or algal uptake before release to the atmosphere (Kosten et al., 2016; Wang et al., 2024). Consequently, enhanced greenhouse gas consumption within these established root-associated habitats may provide a plausible biological pathway for rapid emission mitigation. Simultaneously, partial coverage of the water surface by the CFW may have reduced wind shear and surface turbulence, lowering gas transfer velocities and constraining water-air exchange (Raymond et al., 2012). Nevertheless, short-term variability in microbial activity or physical forcing may temporarily reduce mitigation effects, leading to emissions comparable to those observed in the control channel at some measurement time points. Yet overall greenhouse gas fluxes were substantially lower in the treatment channel, highlighting the potential of CFWs to reduce long-term greenhouse gas emissions, even when nutrient removal is limited. Future studies should couple water-air flux measurements with dissolved gas profiles, direct estimates of gas transfer velocities, and targeted sediment and water-column process assays, including microbial analyses, to mechanistically distinguish altered physical gas exchange from shifts in greenhouse gas production, consumption, or in-water transformation. Such integration will enable more precise attribution of emission reductions and inform the optimisation of CFW design to maximise greenhouse gas mitigation.

The relationships between nutrient dynamics and greenhouse gas emissions are often complex and vary across gases (Zhang et al., 2024). We found that CO_2 fluxes were positively associated with total nitrogen levels, whereas CH_4 and N_2O emissions appeared largely decoupled from dissolved nutrient levels. CH_4 is primarily produced under anoxic conditions in the sludge (Conrad, 2009), which may explain why overall CH_4 fluxes were largely unaffected by fluctuating nutrient levels in the water column. Likewise, the absence of detectable ammonium levels in the water column points to anaerobic denitrification (the conversion of nitrate to nitrogen) in the sludge as the dominant production pathway of N_2O, rather than aerobic nitrification (the conversion of ammonium to nitrate) in the water column, which is consistent with previous studies indicating denitrification in anaerobic sludge as a dominant source of N_2O under similar conditions (Massara et al., 2017). These patterns indicate that CFW-driven reductions in dissolved nutrients are likely most relevant for regulating CO_2 emissions, whereas sustained CH_4 and N_2O mitigation likely relies on microbial consumption or physical processes. Consistent with these assumptions, we observed stronger CO_2 emission reductions in the second monitoring year, coinciding with more pronounced organic nitrogen level reductions. In contrast, the magnitudes of CH_4 and N_2O emission reductions were comparable between monitoring years. Together, these findings suggest that CFW-driven emission reductions likely arise from a combination of enhanced greenhouse gas consumption (CO_2, CH_4, and N_2O) and nutrient-driven suppression of production (CO_2 only) over longer timescales. Nevertheless, further emission reductions in CH_4 and N_2O may be achieved through targeted operational management actions such as periodic sludge removal to limit anaerobic production.

Wetland plant harvesting is a critical management practice to sustain plant growth and maximise nutrient uptake by the plants on the CFW (Huth et al., 2021). Across our study period, a total of 969.65 kg of plant biomass was removed through harvesting (refer to Table S1). Notably, immediately after harvesting events, we observed significant, albeit temporary, increases in CH_4 emissions in the treatment channel after the CFW. These short-lived spikes in CH_4 emissions may reflect the release of accumulated CH_4 due to the physical disturbance of the root mats and/or due to reduced plant-mediated CH_4 oxidation in the root systems following the removal of plant shoots. Comparable short-term spikes in CH_4 emissions after harvesting have been reported in constructed wetlands with rooted vegetation, where disturbance-driven release and reduced plant-mediated CH_4 oxidation were identified as the primary mechanisms of short-term spikes in CH_4 emissions (Zhu et al., 2007). Importantly, these temporary spikes in CH_4 emissions can offset some of the long-term emission-reduction benefits of CFWs, with CO_2-equivalent reductions ranging from 22% when including post-harvest CH_4 spikes to 31% without these spikes. Consequently, balancing harvesting frequency to maintain strong nutrient removal while minimising disturbance-based emissions is critical. Beyond emission management, harvested biomass further represents an underutilised carbon resource. In our study, harvested plant material was burned for disposal. However, converting such harvested biomass into stable carbon forms, such as biochar, could further help inset carbon emissions (the implementation of emission reduction or sequestration projects within an organisation's own value chain; Ebersold et al., 2023) through enhanced carbon sequestration and storage, while producing materials that are beneficial for soil amendment or wastewater filtration (Chiaramonti et al., 2024; Xiang et al., 2020). Integrating low-disturbance harvesting (e.g., phased or partial harvesting) with sustainable plant biomass processing could therefore further improve the climate performance of CFWs by simultaneously reducing post-harvest CH_4 release and enhancing long-term carbon storage.

As with other emission reduction approaches, the economic feasibility of installing and maintaining CFWs is an important consideration for their large-scale implementation in wastewater treatment systems globally. In our study, the CAPEX and OPEX costs equated to AU\$873 m^{-2} (US\$567 m^{-2}) and AU\$107 m^{-2} $year^{-1}$ (US\$70 m^{-2} $year^{-1}$), respectively, with total project costs accumulating to AU\$360,016 (US\$234,010) over the two-year monitoring period. These values are largely comparable to cost ranges reported by Awad et al. (2025), who documented CAPEX ranging between US\$15-780 m^{-2} and OPEX between US\$0.5-38 m^{-2} $year^{-1}$. However, our CAPEX and OPEX estimates also included the quantification and reporting of greenhouse gas fluxes in addition to standard CFW construction and maintenance costs and nutrient monitoring. Although our channel-scale measurements precluded direct calculation of the cost per tonne of CO_2-equivalent mitigated, the observed 22-31% reduction under limited surface coverage suggests that lagoon-wide implementation could yield substantial emission abatement if appropriately scaled. Compared to capitalintensive alternatives such as aeration upgrades, lagoon covers, or biogas capture systems, which often require substantial investments of tens or hundreds of millions of dollars, depending on the scale of upgrades (Guo et al., 2014), CFWs may offer a relatively low-cost, low-infrastructure complement for addressing both nutrient loads and greenhouse gas emissions. Future work integrating whole-lagoon emission quantification with economic modelling will be essential to refine cost-effectiveness estimates and support sector-wide adoption.

Limitations and future directions

Our study provides valuable insights into the potential of CFWs to mitigate greenhouse gas emissions from wastewater lagoons. Nevertheless, there are several limitations that warrant consideration. First, we quantified emission reductions using a paired, channelised design within a single lagoon. This approach allowed for a robust comparison between treatment and control conditions under identical hydraulic and climatic settings, but it does not capture whole-lagoon emission responses. Scaling these findings to lagoon-wide emission reductions will require studies that explicitly account for CFW coverage, lagoon hydrodynamics, and spatial heterogeneity in emissions. Second, the study was conducted at a single site using one CFW configuration, vegetation assemblage, and surface coverage (~7.4% of lagoon area). Although this design reflects realistic operational constraints, additional studies across a broader range of lagoon types, climates, hydraulic retention times, and CFW coverages are needed to develop predictive, generalisable estimates of greenhouse gas mitigation potentials across the water sector. Third, the mechanistic understanding of observed greenhouse gas reductions remains limited. Although relationships among nutrient concentrations, dissolved oxygen, and greenhouse gas fluxes suggest possible pathways for emission reductions, we did not directly investigate microbial community composition or process-level rates (e.g., CO_2 uptake, methanotrophy, or N_2O reduction). Future work integrating microbial, isotopic, or sediment-based measurements would help disentangle production versus consumption pathways and strengthen mechanistic attribution. Finally, although we quantified the capital and operational costs associated with the CFW installation, monitoring, and maintenance, the lack of whole-lagoon emission estimates precluded the calculation of cost-effectiveness metrics such as cost per tonne of CO_2-equivalent mitigated. Integrating full-scale emission measurements with economic assessments will be essential to robustly evaluate the cost-effectiveness of CFW deployment and to support decision-making for large-scale implementation in wastewater treatment systems.

Conclusions

Our findings demonstrate that CFWs can deliver rapid and substantial reductions in greenhouse gas emissions from open wastewater holding lagoons. Across our two-year monitoring period, the installation of a CFW resulted in an average reduction of 22-31% in CO_2-equivalent emissions from the treatment channel, driven primarily by reduced CH_4 emissions (32-66%), with additional reductions in CO_2 (24-36%) and N_2O emissions (18%). We further found an average reduction of 12% in dissolved organic nitrogen, whereas nitrate and phosphorus levels remained unaffected. Notably, emission reductions emerged even in the absence of strong nutrient effects, suggesting that enhanced microbial and algal greenhouse gas consumption, coupled with reduced surface gas transfer due to partial water coverage by the CFW, may have been key drivers. To fully realise the mitigation potential of CFWs, future research should focus on scaling to whole-lagoon applications under operational conditions and disentangling the microbial and physical mechanisms driving

CFW-associated greenhouse gas reductions. Importantly, the installation of CFWs can have other potential benefits beyond emission reductions, including mitigating pollutant concentrations like per- and polyfluoroalkyl substances (PFAS), toxic metals, pharmaceuticals, and other contaminants (Awad et al., 2024; Shahid et al., 2018), as well as providing habitat for a range of native species or acting as stepping stones to improve habitat connectivity in urban areas (Calheiros et al., 2025; Karstens et al., 2022), making them a multifunctional and comprehensive nature-based solution within the water sector. However, broader adoption across the water sector will likely depend on the ability to address practical barriers such as upfront capital costs, ongoing maintenance requirements, and the need for stronger regulatory guidance and empirical evidence under full-scale operational conditions. With targeted research, policy support, and demonstration projects, CFWs could become a viable and multifunctional nature-based option to reduce Scope 1 emissions from wastewater treatment plants while delivering wider benefits for water quality and biodiversity.

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Figures

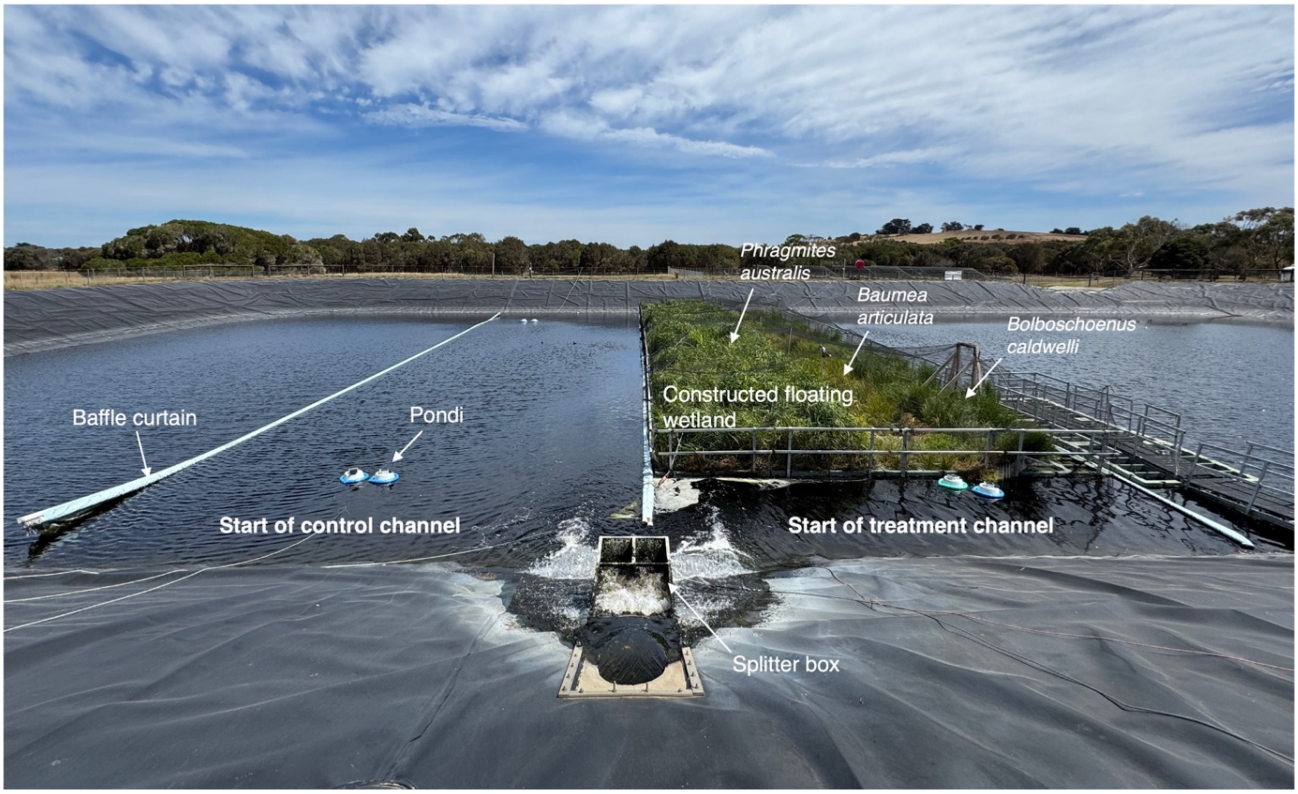

Figure 1: Control (on the left) and treatment (on the right, with the CFW) channels created by baffle curtains in the wastewater holding lagoon. The inflowing water is distributed between the two channels by a splitter box. The floating greenhouse gas sensors (Pondi) were deployed at the start (inlet) and end (outlet) of each channel.

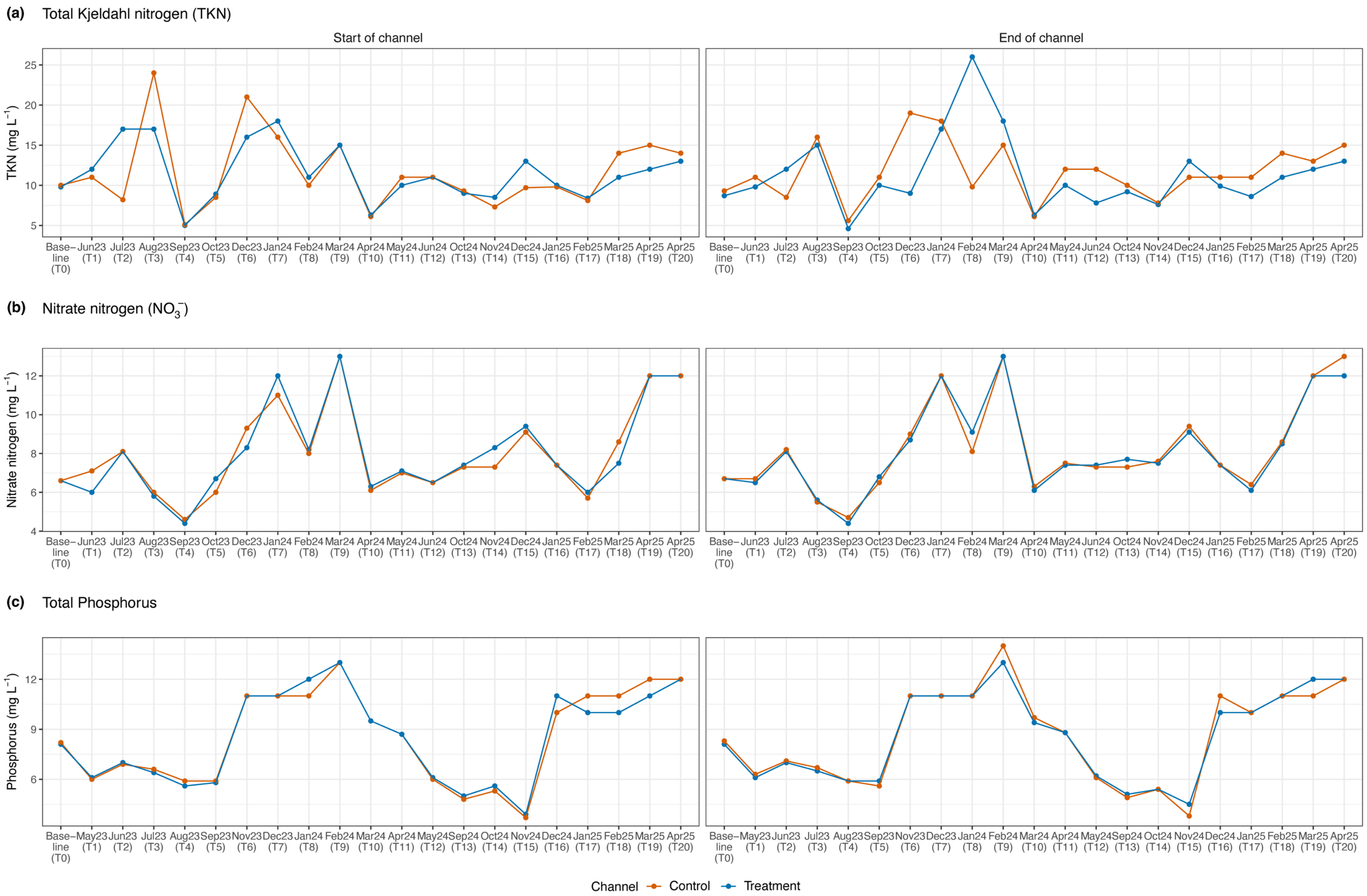

Figure 2: (a) Total Kjeldahl nitrogen, (b) nitrate nitrogen (NO_3^-), and (c) total phosphorus levels (all in mg L^{-1}) at the start and end of the control (orange) and treatment (blue) channels over the sampling period. Baseline (T0) indicates baseline measurements before the CFW was installed.

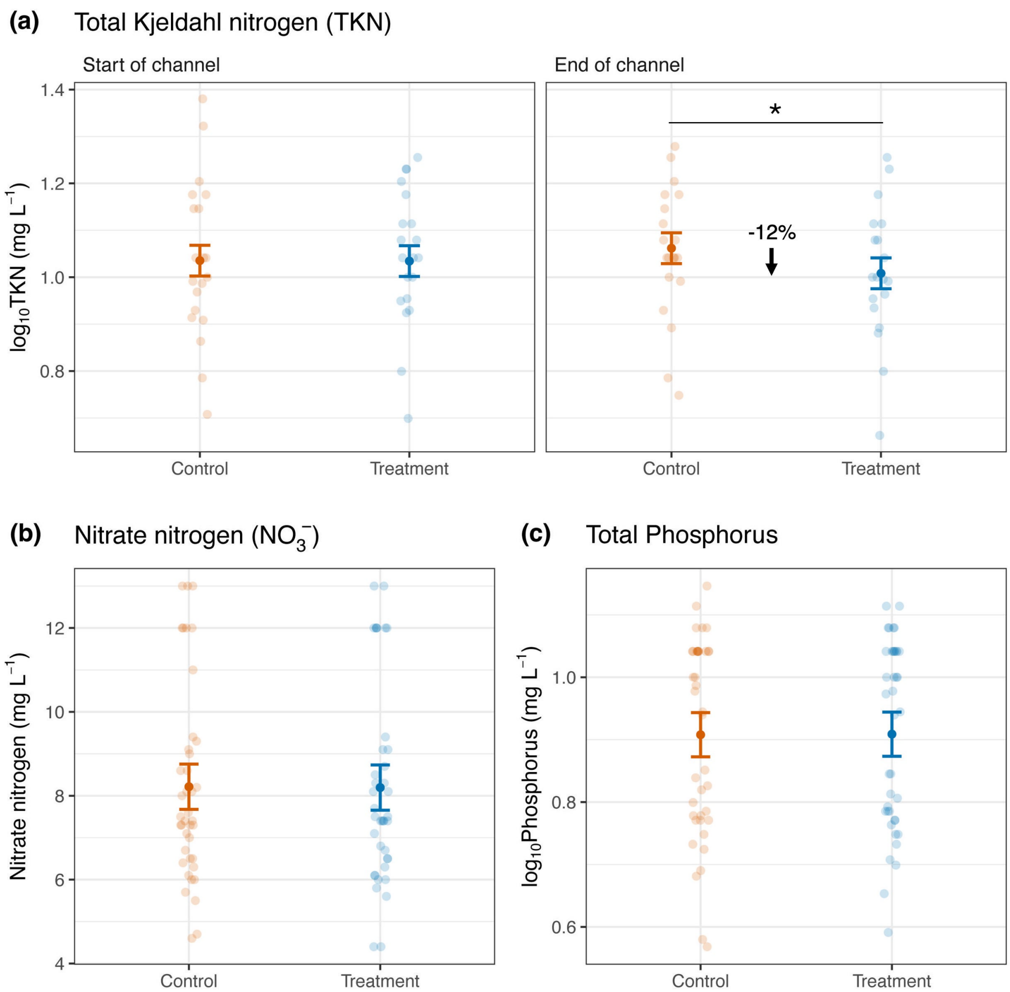

Figure 3: Overall (a) total Kjeldahl nitrogen, (b) nitrate nitrogen (NO_3^-), and (c) total phosphorus levels (all in mg L^{-1}) in the control (orange) and treatment (blue) channels across the sampling period. Semitransparent points are the raw data, opaque points are the predicted means and standard errors from the best-fitting statistical models.

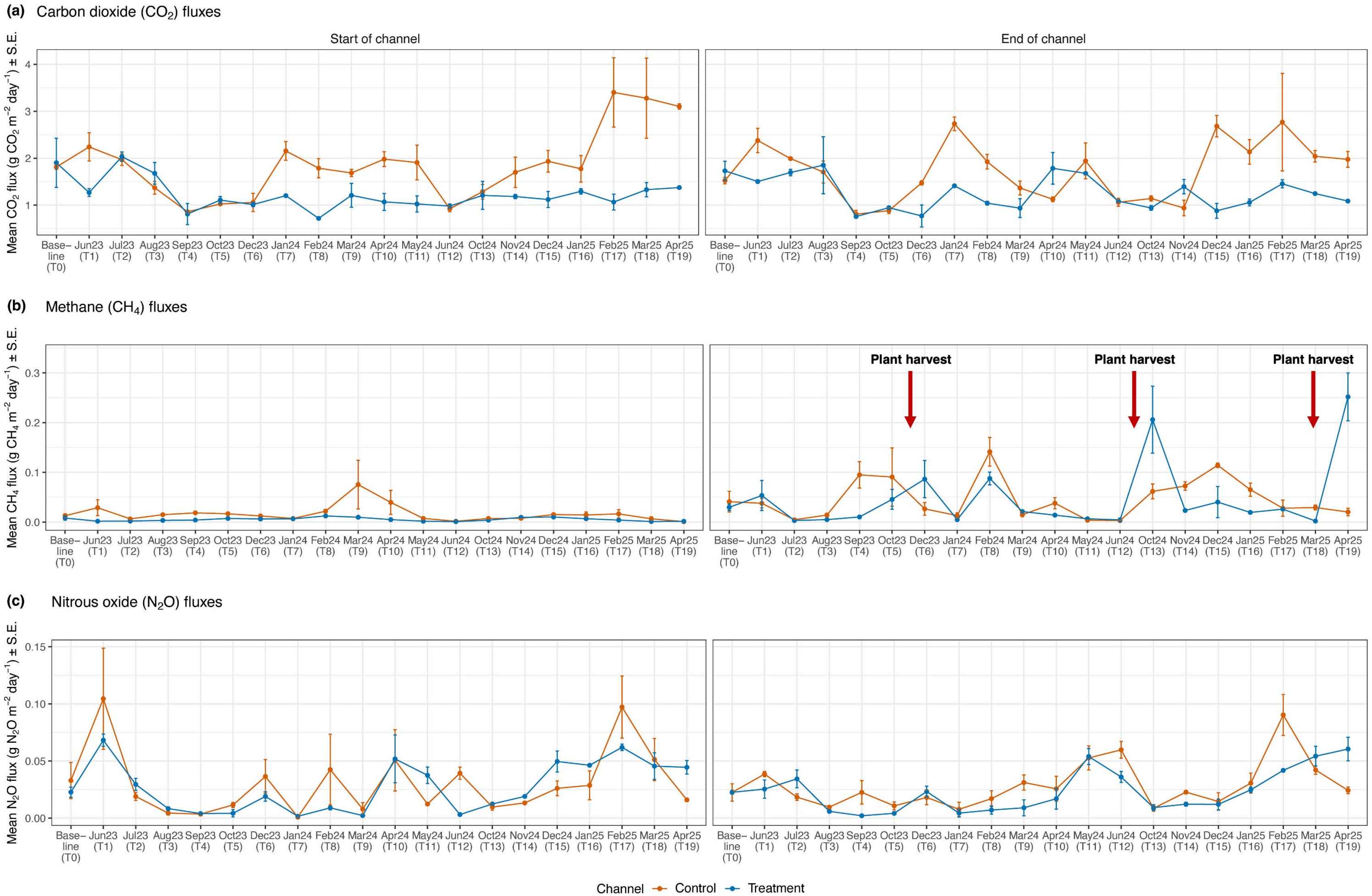

Figure 4: Average (a) carbon dioxide (CO_2; in g CO_2 m^{-2} day^{-1}), (b) methane (CH_4; in g CH_4 m^{-2} day^{-1}), and (c) nitrous oxide (N_2O; g N_2O m^{-2} day^{-1}) fluxes at the start and end of the control (orange) and treatment (blue) channels over the sampling period. Baseline (T0) indicates baseline measurements before the CFW was installed. Red arrows indicate plant harvesting events. Points are means, error bars indicate standard errors.

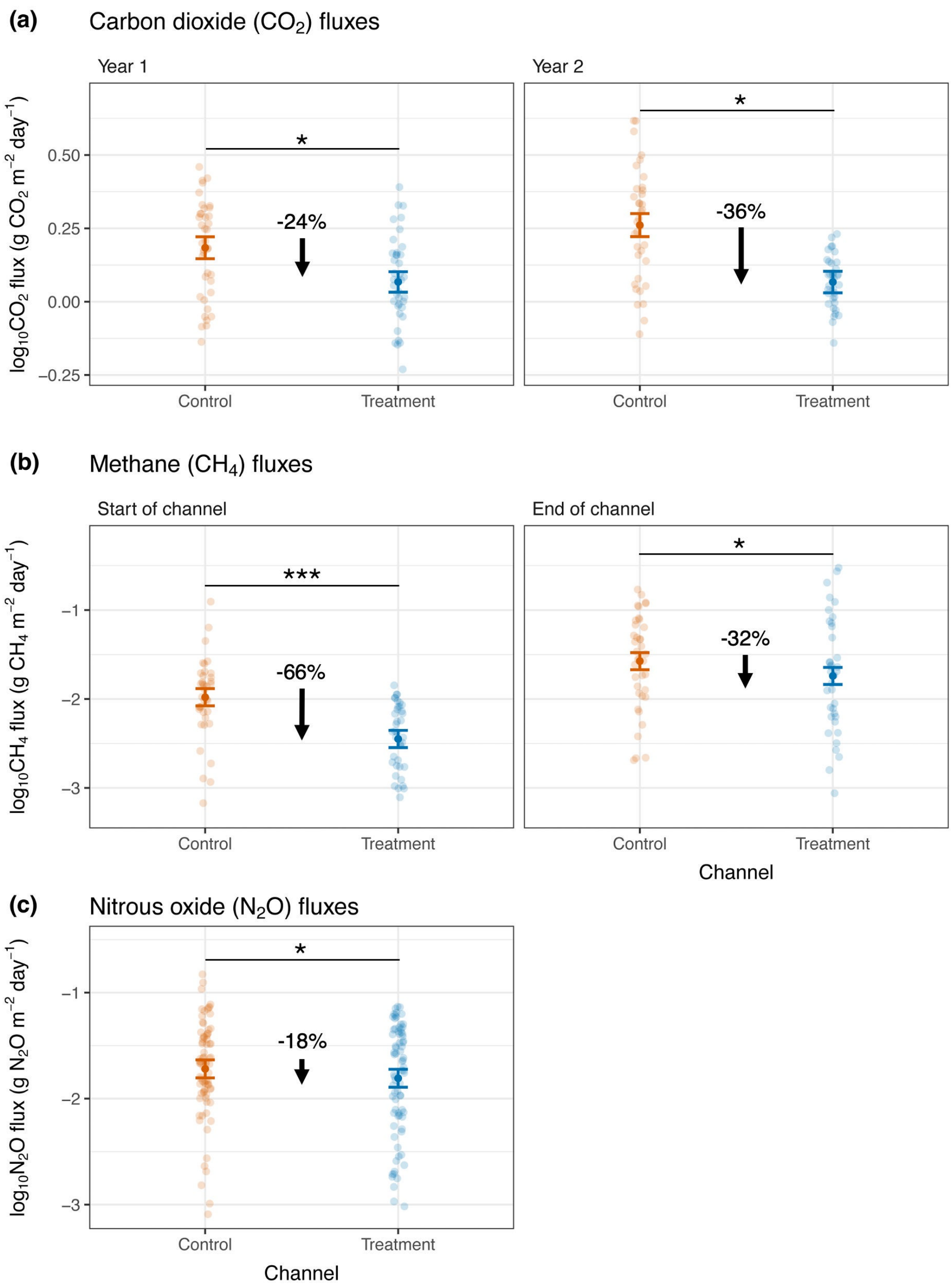

Figure 5: (a) Carbon dioxide fluxes (CO_2; in g CO_2 m^{-2} day^{-1}) during sampling year 1 (T1-T10) and sampling year 2 (T11-T19), (b) methane fluxes (CH_4; in g CH_4 m^{-2} day^{-1}) at the start and end of the control (orange) and treatment (blue) channels, and (c) nitrous oxide fluxes (N_2O; in g N_2O m^{-2} day^{-1}) across both sampling years in the control (orange) and treatment (blue) channels. Semitransparent points are the raw data, opaque points are the predicted means and standard errors from the best-fitting statistical models.

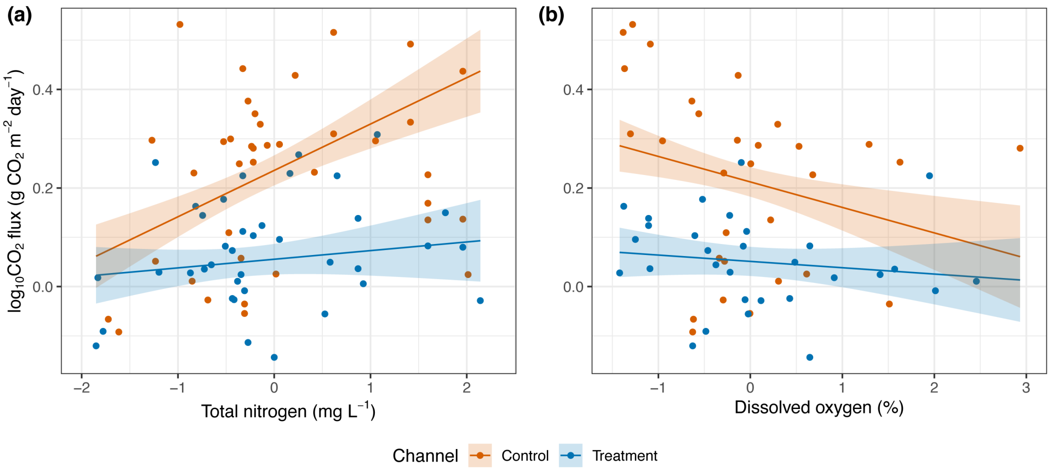

Figure 6: Relationships between carbon dioxide fluxes (CO_2; in g CO_2 m^{-2} day^{-1}) and (a) total nitrogen (in mg L^{-1}) and (b) dissolved oxygen (in %) in the control (in orange) and treatment (in blue) channels. Points are the raw data, lines of best fit are the predicted linear relationships from the best-fitting statistical models ± 95% confidence intervals.

Table 1: Estimated capital (CAPEX) and operational (OPEX) costs associated with the installation, monitoring, and maintenance of the CFW over a two-year period. All costs are presented in Australian (AU$) and United States dollars (US$) using 2025 conversion values (AU$1 = US$0.65). Reported costs are based on actual project expenditures and supplier quotes from Victoria, Australia (2025).

Item	Details	CAPEX (AU$ / US$)	OPEX (AU$ / US$)
CFW modules	60 modules at AU$550 m^{-2} (US$356 m^{-2}), including substrate and plants. Price also includes design, manufacturing, and transport.	179,916 / 116,945	
Handrails	Installed around the perimeter of the CFW for operational health and safety (OH&S) compliance.	6,000 / 3,900	
Bird netting	Protective netting to reduce plant predation during establishment phase.	8,000 / 5,200	
Access ramp	Access to CFW for maintenance and plant harvesting.	7,000 / 4,550	
Baffle curtains	Vinyl baffle curtains to create separate control and treatment channels. Note that these components are not typically associated with CFW installation but were specific to this experimental setup.	8,000 / 5,200	
CFW installation	Labour: five staff over five days.	10,000 / 6,500	
Plant harvesting & other CFW maintenance	Labour: three days for plant harvesting plus periodic inspections and general upkeep.		21,100 / 13,715
Pondi loggers	16 Pondi for continuous greenhouse gas flux measurements (manufacturing included).	64,000 / 41,600	
Water quality analysis	Routine nutrient and general water quality monitoring.	6,000 / 3,900	
Greenhouse gas flux monitoring and data analysis	Labour: staff time for five days per month over two years for flux data analysis, reporting, and instrument maintenance.		50,000 / 32,500

Total estimated cost	288,916 / 187,795	71,100 / 46,215
Grand total for 2 years	**AU$360,016 / US$234,010**	

References

Audet, J., Levi, E.E., Jeppesen, E., Davidson, T.A., 2024. Nutrient enrichment—but not warming—increases nitrous oxide emissions from shallow lake mesocosms. Limnol. Oceanogr., 70(2), 43-54.

Awad, J., Navarro, D., Kirby, J., Walker, C., Juhasz, A., 2024. Long-term management of PFAS contaminated water using constructed floating wetlands: Opportunities, limitations, and implementation considerations. Crit. Rev. Environ. Sci. Technol., 54(24), 1709-1733.

Awad, J., Walker, C., Page, D., Arslan, M., White, S.A., Lucke, T., Beecham, S., Winston, R.J., Strosnider, W.H.J., Nicodemus, P., Streb, C., van Leeuwen, J., 2025. Assessing the costs of constructed floating wetlands for the treatment of surface waters and wastewater. ACS ES&T Water, 5(8), 4737-4747.

Bastviken, D., 2009. Methane. in: G.E. Likens (Ed.), Encyclopedia of Inland Waters, Vol. 2, Elsevier. Oxford, UK, pp. 783-805.

Benvenuti, T., Hamerski, F., Giacobbo, A., Bernardes, A.M., Zoppas-Ferreira, J., Rodrigues, M.A., 2018. Constructed floating wetland for the treatment of domestic sewage: A real-scale study. J. Environ. Chem. Eng., 6(5), 5706-5711.

Bureau of Meteorology, Climate statistics for Australian locations. Retrieved 30 July 2025 from http://www.bom.gov.au/climate/averages/tables/cw_086354.shtml.

Burnham, K.P., Anderson, D.R., 2004. Multimodel inference: understanding AIC and BIC in model selection. Sociol. Methods Res., 33(2), 261-304.

Calheiros, C.S., Ilarri, M., Godinho, M., Castro, P.M., Pereira, S.I., 2025. Biodiversity assessment in a floating treatment wetland established in a stormwater pond. Ecol. Eng., 215, 107598.

Campos, J.L., Valenzuela-Heredia, D., Pedrouso, A., Val del Río, A., Belmonte, M., Mosquera-Corral, A., 2016. Greenhouse gases emissions from wastewater treatment plants: Minimization, treatment, and prevention. J. Chem., 2016, 1-12.

Carey, R.O., Migliaccio, K.W., 2009. Contribution of wastewater treatment plant effluents to nutrient dynamics in aquatic systems: a review. Environ. Manage., 44, 205-217.

Chiaramonti, D., Lehmann, J., Berruti, F., Giudicianni, P., Sanei, H., Masek, O., 2024. Biochar is a long-lived form of carbon removal, making evidence-based CDR projects possible. Biochar, 6(1), 81.

Conrad, R., 2009. The global methane cycle: recent advances in understanding the microbial processes involved. Environ. Microbiol. Rep., 1(5), 285-92.

Ebersold, F., Hechelmann, R.-H., Holzapfel, P., Meschede, H., 2023. Carbon insetting as a measure to raise supply chain energy efficiency potentials: Opportunities and challenges. Energy Convers. Manag.: X, 20, 100504.

Guo, T., Englehardt, J., Wu, T., 2014. Review of cost versus scale: water and wastewater treatment and reuse processes. Water Sci. Technol., 69(2), 223-234.

Hallin, S., Philippot, L., Löffler, F.E., Sanford, R.A., Jones, C.M., 2018. Genomics and ecology of novel N_2O-reducing microorganisms. Trends Microbiol., 26(1), 43-55.

Ho, L., Jerves-Cobo, R., Morales, O., Larriva, J., Arevalo-Durazno, M., Barthel, M., Six, J., Bode, S., Boeckx, P., Goethals, P., 2021. Spatial and temporal variations of greenhouse gas emissions from a waste stabilization pond: Effects of sludge distribution and accumulation. Water Res., 193, 116858.

Huang, F., Shen, W., Zhang, X., Seferlis, P., 2020. Impacts of dissolved oxygen control on different greenhouse gas emission sources in wastewater treatment process. J. Clean. Prod., 274, 123233.

Huang, Y., Deng, M., Jiang, X., Li, L., Zhou, S., Wang, Z., Song, K., Wu, F., 2025. Biofilm-mediated mitigation of N_2O emissions in submerged macrophytes area of freshwater lake. Chem. Eng. J., 521, 166896.

Huth, I., Walker, C., Kulkarni, R., Lucke, T., 2021. Using constructed floating wetlands to remove nutrients from a waste stabilization pond. Water, 13(13), 1746.

IPCC, 2007. Climate change 2007: Synthesis report. Contribution of working group I, II and III to the fourth assessment report of the Intergovernmental Panel on Climate Change.

IPCC, 2014. Climate change 2014: Mitigation of climate change. Contribution of working group III to the fifth assessment report of the Intergovernmental Panel on Climate Change. Cambridge University Pres, Cambridge, United Kingdom and New York, NY, USA.

IPCC, 2021. Climate change 2021: The physical science basis. Contribution of working group I to the sixth assessment report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Karstens, S., Dorow, M., Bochert, R., Stybel, N., Schernewski, G., Mühl, M., 2022. Stepping stones along urban coastlines—improving habitat connectivity for aquatic fauna with constructed floating wetlands. Wetlands, 42(7), 76.

Kosten, S., Piñeiro, M., de Goede, E., de Klein, J., Lamers, L.P., Ettwig, K., 2016. Fate of methane in aquatic systems dominated by free-floating plants. Water Res., 104, 200-207.

Li, Y., Shang, J., Zhang, C., Zhang, W., Niu, L., Wang, L., Zhang, H., 2021. The role of freshwater eutrophication in greenhouse gas emissions: A review. Sci. Total Environ., 768, 144582.

Lucke, T., Walker, C., Beecham, S., 2019. Experimental designs of field-based constructed floating wetland studies: A review. Sci. Total Environ., 660, 199-208.

Maktabifard, M., Al-Hazmi, H.E., Szulc, P., Mousavizadegan, M., Xu, X., Zaborowska, E., Li, X., Mąkinia, J., 2023. Net-zero carbon condition in wastewater treatment plants: A systematic review of mitigation strategies and challenges. Renew. Sustain. Energy Rev., 185, 113638.

Malerba, M.E., Edwards, B., Schuster, L., Odebiri, O., Glen, J., Kelly, R., Phan, P., Grinham, A., Macreadie, P.I., 2025. Technical note: Pondi – a low-cost logger for long-term monitoring of methane, carbon dioxide, and nitrous oxide in aquatic and terrestrial systems. Biogeosciences, 22(18), 5051-5067.

Massara, T.M., Malamis, S., Guisasola, A., Baeza, J.A., Noutsopoulos, C., Katsou, E., 2017. A review on nitrous oxide (N_2O) emissions during biological nutrient removal from municipal wastewater and sludge reject water. Sci. Total Environ., 596, 106-123.

Nguyen, T.K.L., Ngo, H.H., Guo, W., Chang, S.W., Nguyen, D.D., Nghiem, L.D., Liu, Y., Ni, B., Hai, F.I., 2019. Insight into greenhouse gases emissions from the two popular treatment technologies in municipal wastewater treatment processes. Sci. Total Environ., 671, 1302-1313.

Nichols, P., Lucke, T., Drapper, D., Walker, C., 2016. Performance evaluation of a floating treatment wetland in an urban catchment. Water, 8(6), 244.

Ollivier, Q.R., Maher, D.T., Pitfield, C., Macreadie, P.I., 2018. Punching above their weight: Large release of greenhouse gases from small agricultural dams. Glob. Chang. Biol., 25(2), 721-732.

Pavlineri, N., Skoulikidis, N.T., Tsihrintzis, V.A., 2017. Constructed floating wetlands: a review of research, design, operation and management aspects, and data meta-analysis. Chem. Eng. J., 308, 1120-1132.

Pinheiro, J., Bates, D., R Core Team, 2024. nlme: Linear and nonlinear mixed effects models. R package version 3.1-166.

Preisner, M., Neverova-Dziopak, E., Kowalewski, Z., 2020. An analytical review of different approaches to wastewater discharge standards with particular emphasis on nutrients. Environ. Manage., 66(4), 694-708.

R Core Team, 2013. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Raymond, P.A., Zappa, C.J., Butman, D., Bott, T.L., Potter, J., Mulholland, P., Laursen, A.E., McDowell, W.H., Newbold, D., 2012. Scaling the gas transfer velocity and hydraulic geometry in streams and small rivers. Limnology and Oceanography: Fluids and Environments, 2(1), 41-53.

Shahid, M.J., Al-Surhanee, A.A., Kouadri, F., Ali, S., Nawaz, N., Afzal, M., Rizwan, M., Ali, B., Soliman, M.H., 2020. Role of microorganisms in the remediation of wastewater in floating treatment wetlands: A review. Sustainability, 12(14), 5559.

Shahid, M.J., Arslan, M., Ali, S., Siddique, M., Afzal, M., 2018. Floating wetlands: A sustainable tool for wastewater treatment. Clean – Soil, Air, Water, 46(10), 1800120.

Shammas, N.K., Wang, L.K., Wu, Z., 2009. Waste stabilization ponds and lagoons Biological treatment processes, Springer, pp. 315-370.

Song, C., Zhu, J.-J., Willis, J.L., Moore, D.P., Zondlo, M.A., Ren, Z.J., 2023. Methane emissions from municipal wastewater collection and treatment systems. Environ. Sci. Technol., 57(6), 2248-2261.

Tranvik, L.J., Downing, J.A., Cotner, J.B., Loiselle, S.A., Striegl, R.G., Ballatore, T.J., Dillon, P., Finlay, K., Fortino, K., Knoll, L.B., 2009. Lakes and reservoirs as regulators of carbon cycling and climate. Limnol. Oceanogr., 54(6part2), 2298-2314.

U.S. Environmental Protection Agency, 2012. Summary report: Global anthropogenic non-CO_2 greenhouse gas emissions: 1990-2030.

Wang, F., Zhang, S., Hu, X., Lv, X., Liu, M., Ma, Y., Manirakiza, B., 2024. Floating plants reduced methane fluxes from wetlands by creating a habitat conducive to methane oxidation. J. Environ. Sci., 135, 149-160.

Wickham, H., 2016. ggplot2: Elegant graphics for data analysis. Springer.

Wu, Z., Duan, H., Li, K., Ye, L., 2022. A comprehensive carbon footprint analysis of different wastewater treatment plant configurations. Environ. Res., 214, 113818.

Xiang, W., Zhang, X., Chen, J., Zou, W., He, F., Hu, X., Tsang, D.C., Ok, Y.S., Gao, B., 2020. Biochar technology in wastewater treatment: A critical review. Chemosphere, 252, 126539.

Zhang, J., Cao, L., Liu, Z., Wan, L., Cao, X., Zhou, Y., Song, C., 2024. Relationship between eutrophication and greenhouse gases emission in shallow freshwater lakes. Sci. Total Environ., 925, 171610.

Zhu, G., Shi, H., Zhong, L., He, G., Wang, B., Shan, J., Han, P., Liu, T., Wang, S., Liu, C., 2025. Nitrous oxide sources, mechanisms and mitigation. Nat. Rev. Earth Environ., 6, 574-59.

Zhu, N., An, P., Krishnakumar, B., Zhao, L., Sun, L., Mizuochi, M., Inamori, Y., 2007. Effect of plant harvest on methane emission from two constructed wetlands designed for the treatment of wastewater. J. Environ. Manag., 85(4), 936-943.

Zuur, A.F., Ieno, E.N., Walker, N.J., Saveliev, A.A., Smith, G.M., 2009. Mixed effects models and extensions in ecology with R. Springer.

Supplementary material

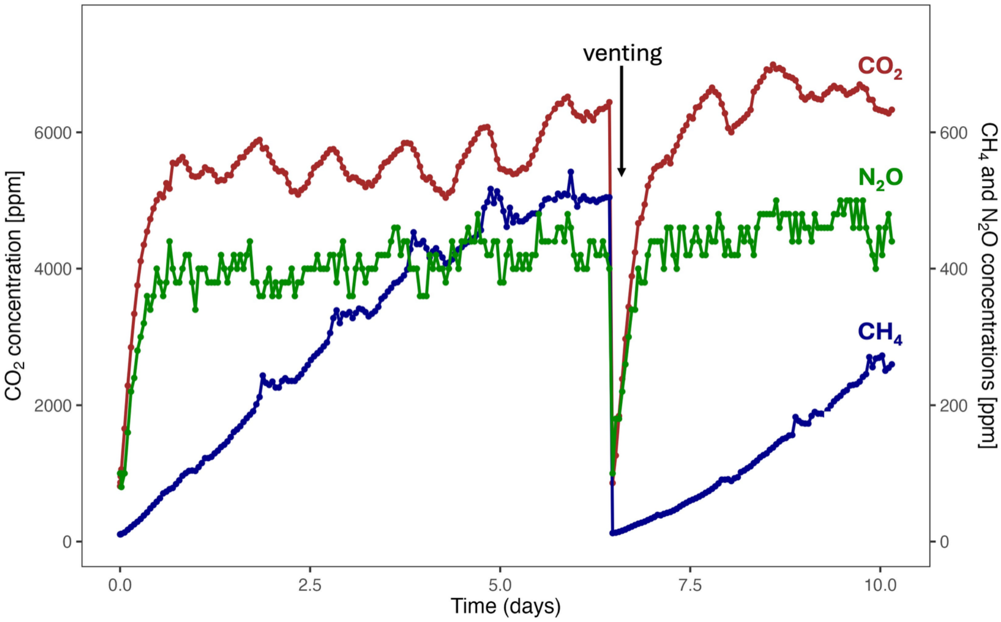

Figure S1: Example of 10 days of hourly carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) flux data collected in the wastewater holding lagoon. The arrow indicates a venting event, where the external air pump injected ambient air into the chamber to reset gas concentrations to environmental levels. Note that greenhouse gas fluxes were calculated from the linear increase in gas concentrations, excluding periods of saturation.

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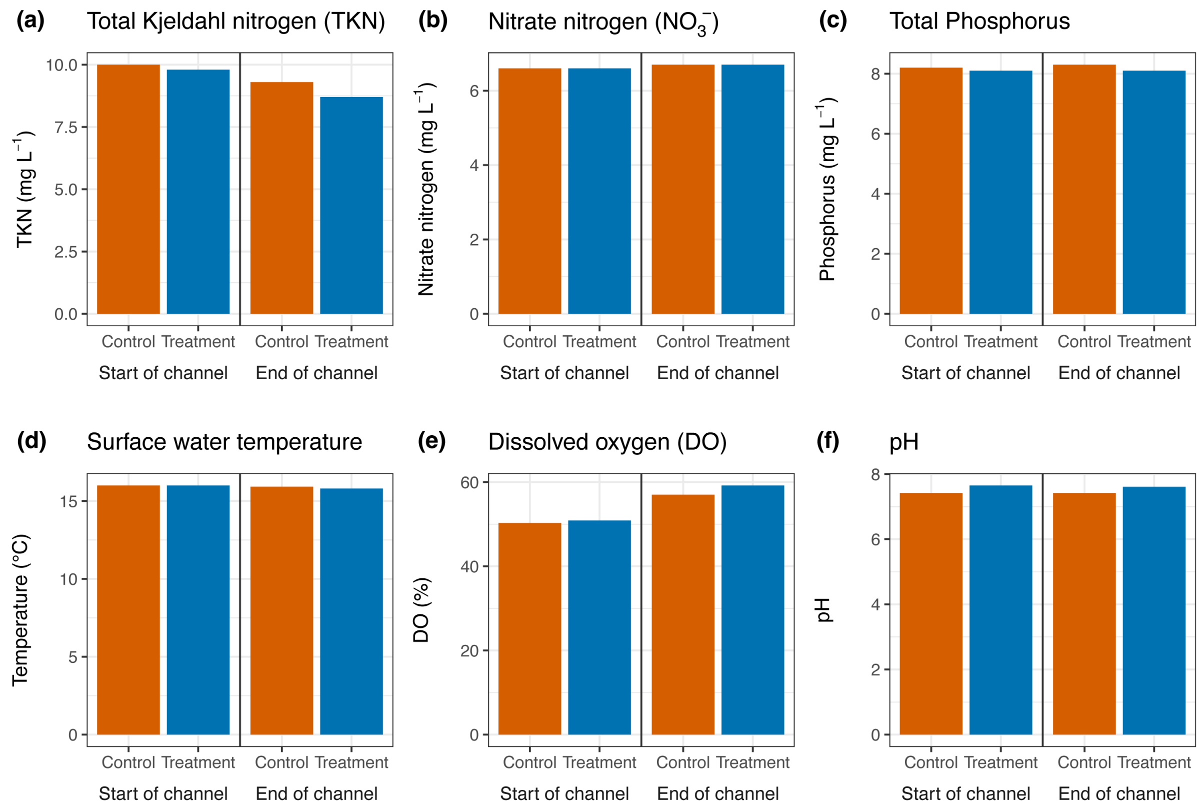

760 **Figure S2:** (a) total Kjeldahl nitrogen (in mg L^{-1}), (b) nitrate nitrogen (NO_3^-; in mg L^{-1}), (c)
761 total phosphorus (in mg L^{-1}), (d) surface water temperature (in °C), (e) dissolved oxygen
762 (DO; in %), and (f) pH at the start and end of the control (orange) and treatment (blue)
763 channels during baseline monitoring.

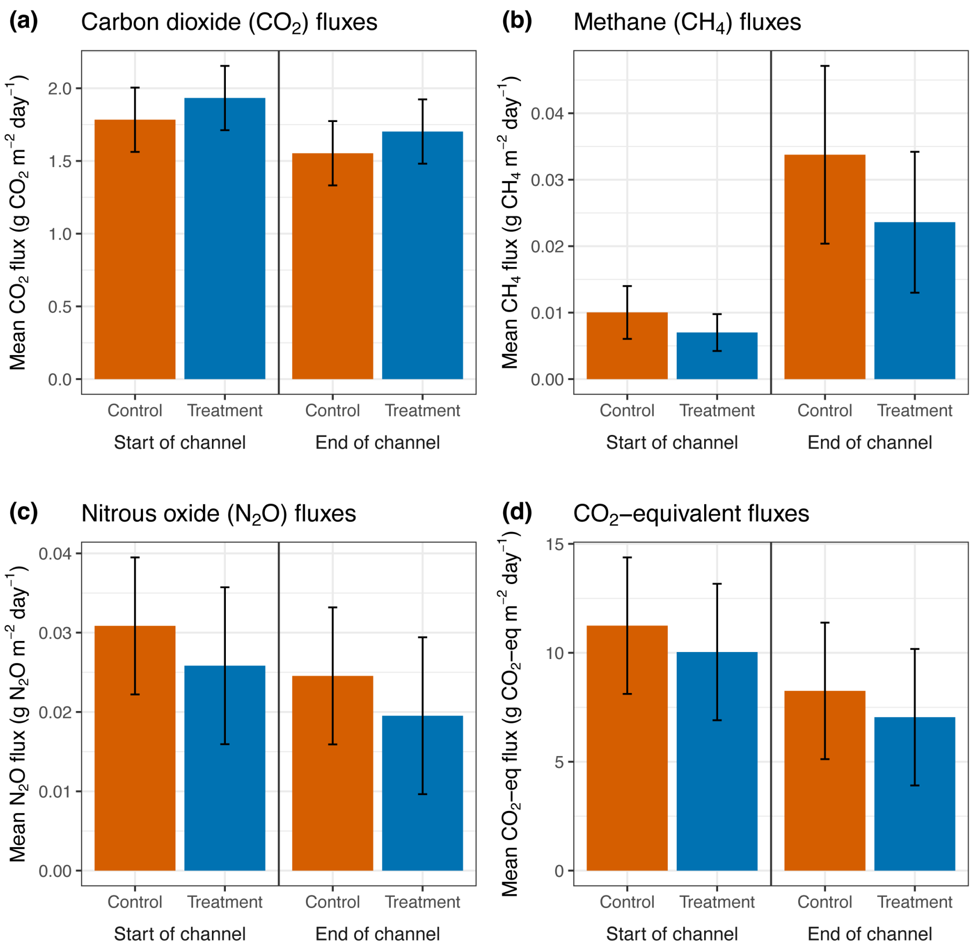

Figure S3: (a) Carbon dioxide (CO_2; in g CO_2 m^{-2} day^{-1}), (b) methane (CH_4; in g CH_4 m^{-2} day^{-1}), and (c) nitrous oxide (N_2O; g N_2O m^{-2} day^{-1}), and (d) CO_2-equivalent (g CO_2-eq m^{-2} day^{-1}) fluxes at the start and end of the control (orange) and treatment (blue) channels during baseline monitoring.

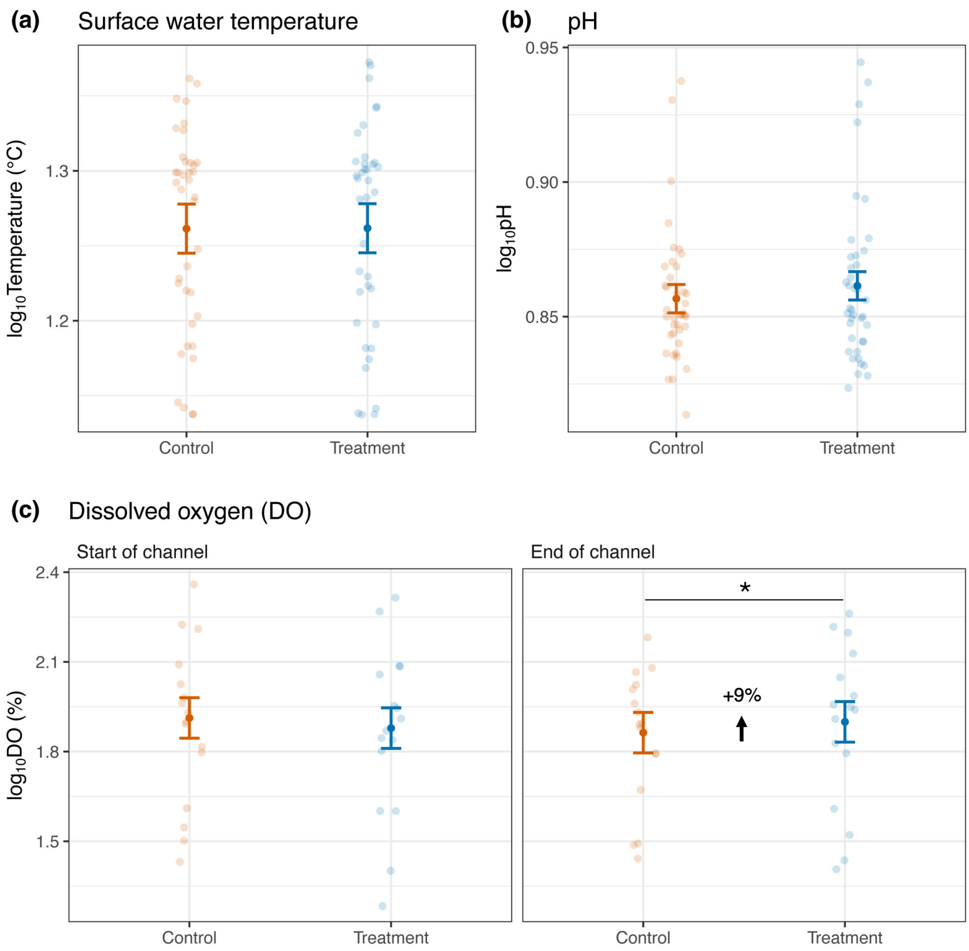

Figure S4: Overall (a) surface water temperature (in °C), (b) pH, and (c) dissolved oxygen concentrations (DO; in %) in the control (orange) and treatment (blue) channels across the sampling period. Semitransparent points are the raw data, opaque points are the predicted means and standard errors from the best-fitting statistical models.

Table S1: Biomasses of wetland plants and weeds that were removed from the CFW during harvesting events.

Time point	Harvested biomass (kg)	Target plant species
November 2023	280	Weeds
September 2024	194	Weeds
	20	*Phragmites australis*
March 2025	120.9	*Bolboschoenus caldwellii*
	27.15	*Baumea articulata*
	327.6	*Phragmites australis*
Total plant biomass removed: 969.65 kg		

Table S2: Outcome of linear models testing for the effects of channel (control or treatment) and sampling location (start or end of channel) on (a) carbon dioxide (CO_2), (b) methane (CH_4), and (c) nitrous oxide (N_2O) fluxes during baseline monitoring. DF are degrees of freedom.

	DF	***F*-value**	***P*-value**
(a) Carbon dioxide (CO_2)			
Intercept	1	65.15	0.0005
Channel	1	0.34	0.58
Sampling location	1	0.82	0.41
Residuals	5		
(b) Methane (CH_4)			
Intercept	1	135.04	<0.0001
Channel	1	0.58	0.47
Sampling location	1	6.58	0.03
Residuals	8		
(c) Nitrous oxide (N_2O)			
Intercept	1	12.77	0.009
Channel	1	0.22	0.66
Sampling location	1	0.36	0.57
Residuals	7		

Table S3: Outcome of linear mixed effects models testing for the effects of sampling year (year 1 or year 2), channel (control or treatment), and sampling location (start or end of channel) on (a) overall total Kjeldahl nitrogen, (b) nitrate nitrogen (NO_3^-), and (c) total phosphorus levels. DF are degrees of freedom.

	Numerator DF	**Denominator DF**	***F*-value**	***P*-value**
(a) Total Kjeldahl nitrogen				
Intercept	1	56	518.79	<0.0001
Sampling year	1	18	0.06	0.81
Channel	1	56	0.004	0.95
Sampling location	1	56	2.27	0.14
Channel × sampling location	1	56	4.45	0.04
(b) Nitrate nitrogen (NO_3^-)				
Intercept	1	58	106.16	<0.0001
Sampling year	1	18	0.21	0.65
Channel	1	58	0.08	0.78
Sampling location	1	58	6.74	0.01
(c) Total phosphorus				
Intercept	1	58	336.23	<0.0001
Sampling year	1	18	0.13	0.72
Channel	1	58	0.08	0.78
Sampling location	1	58	1.37	0.25

Table S4: Outcome of linear mixed effects models testing for the effects of sampling year (year 1 or year 2), channel (control or treatment), and sampling location (start or end of channel) on (a) surface water temperature, (b) pH, and (c) dissolved oxygen levels. DF are degrees of freedom.

	Numerator DF	**Denominator DF**	***F*-value**	***P*-value**
(a) Surface water temperature				
Intercept	1	55	2825.91	<0.0001
Sampling year	1	17	0.03	0.86
Channel	1	55	0.05	0.83
Sampling location	1	55	0.5	0.48
(b) pH				
Intercept	1	58	13951.41	<0.0001
Sampling year	1	18	4.46	0.05
Channel	1	58	1.97	0.17
Sampling location	1	58	0.53	0.47
(c) Dissolved oxygen				
Intercept	1	45	346.06	<0.0001
Sampling year	1	14	0.76	0.39
Channel	1	45	2.79	0.1
Sampling location	1	45	5.84	0.02
Channel × sampling location	1	45	5.94	0.02

Table S5: Outcome of linear mixed effects models testing for the effects of sampling year (year 1 or year 2), channel (control or treatment), and sampling location (start or end of channel) on (a) carbon dioxide (CO_2), (b) methane (CH_4), (c) nitrous oxide (N_2O), and (d) and (e) CO_2-equivalent fluxes. DF are degrees of freedom.

	Numerator DF	**Denominator DF**	***F*-value**	***P*-value**
(a) Carbon dioxide (CO_2) fluxes				
Intercept	1	129	23.31	<0.0001
Sampling year	1	17	2.03	0.17
Channel	1	129	18.92	<0.0001
Sampling location	1	129	0.04	0.84
Year × channel	1	129	4.01	0.04
(b) Methane (CH_4) fluxes				
Intercept	1	129	218.12	<0.0001
Monitoring year	1	17	3.91	0.06
Channel	1	129	31.98	<0.0001
Sampling location	1	129	6.06	0.02
Monitoring year × sampling location	1	129	8.13	0.01
Channel × sampling location	1	129	6.66	0.01
(c) Nitrous oxide (N_2O) fluxes				
Intercept	1	131	267.92	<0.0001
Monitoring year	1	17	5.61	0.03
Channel	1	131	3.14	0.07
Sampling location	1	131	0.88	0.35
(d) CO_2-equivalent fluxes including harvesting-related CH_4 spikes				
Intercept	1	130	105.75	<0.0001
Monitoring year	1	17	5.38	0.03
Channel	1	130	10.34	0.002
Sampling location	1	130	4.17	0.04
(e) CO_2-equivalent fluxes without harvesting-related CH_4 spikes				
Intercept	1	109	92.48	<0.0001
Monitoring year	1	14	5.82	0.03

Channel	1	109	18.51	<0.0001
Sampling location	1	109	2.73	0.1

Table S6: Outcome of linear mixed effects models testing for the effects of channel (control or treatment), sampling location (start or end of channel), dissolved oxygen levels, and total nitrogen (total Kjeldahl nitrogen + nitrate nitrogen) on (a) carbon dioxide (CO_2), (b) methane (CH_4), (c) nitrous oxide (N_2O), and (d) CO_2-equivalent fluxes. DF are degrees of freedom.

	Numerator DF	**Denominator DF**	***F*-value**	***P*-value**
(a) Carbon dioxide (CO_2) fluxes				
Intercept	1	34	28.1	<0.0001
Channel	1	34	9.77	0.003
Sampling location	1	34	5.27	0.03
Total nitrogen	1	34	7.21	0.01
Dissolved oxygen	1	34	4.29	0.04
Channel × total nitrogen	1	34	7.07	0.01
Channel × dissolved oxygen	1	34	2.96	0.09
Channel × sampling location	1	34	4	0.05
(b) Methane (CH_4) fluxes				
Intercept	1	35	118.09	<0.0001
Channel	1	35	26.86	<0.0001
Sampling location	1	35	39.42	<0.0001
Total nitrogen	1	35	0.36	0.55
Dissolved oxygen	1	35	0.27	0.61
(c) Nitrous oxide (N_2O) fluxes				
Intercept	1	41	227.57	<0.0001
Channel	1	41	3.21	0.08
Sampling location	1	41	0.01	0.99
Total nitrogen	1	41	0.37	0.17
Dissolved oxygen	1	41	1.94	0.17
(d) CO_2-equivalent fluxes				
Intercept	1	35	157.78	<0.0001
Channel	1	35	10.19	0.003
Sampling location	1	35	0.22	0.64

Total nitrogen	1	35	0.27	0.6
Dissolved oxygen	1	35	1.95	0.17