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Cost-Effectiveness of Treatment Wetlands for Nitrogen Removal in Tropical and Subtropical Australia

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Citation: Kavehei, E.; Hasan, S.; Wegscheidl, C.; Griffiths, M.; Smart, J.C.R.; Bueno, C.; Owen, L.; Akrami, K.; Shepherd, M.; Lowe, S.; et al. Cost-Effectiveness of Treatment Wetlands for Nitrogen Removal in Tropical and Subtropical Australia. *Water* **2021**, *13*, 3309. https://doi.org/10.3390/w13223309

Academic Editor: Jesus Gonzalez-Lopez

Received: 28 October 2021
Accepted: 17 November 2021
Published: 22 November 2021

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Abstract: Treatment wetlands can reduce nitrogen (N) pollution in waterways. However, the shortage of information on their cost-effectiveness has resulted in their relatively slow uptake in tropical and subtropical Australia, including the catchments of the Great Barrier Reef and Moreton Bay. We assessed the performance of constructed treatment wetlands (CW) and vegetated drains (VD) that treat agricultural runoff, and of sewage treatment plant wetlands (STPW), which polish treated effluent. Treatment performance was estimated as changes in concentration (dissolved inorganic nitrogen, DIN, and total nitrogen, TN; mg L^{-1}) and annual load reductions (kg N ha^{-1} yr^{-1}). We calculated their cost-effectiveness by comparing their N removal against the costs incurred in their design, construction, and maintenance. Overall, CWs and VDs reduced DIN concentrations by 44% (0.52 to 0.29 mg L^{-1}), and STPW reduced them by 91% (2.3 to 0.2 mg L^{-1}); STPWs also reduced TN concentrations by 72%. The efficiency varied among sites, with the best performing CWs and VDs being those with relatively high inflow concentrations (>0.2 mg L^{-1} of DIN, >0.7 mg L^{-1} of TN), low suspended solids, high vegetation cover and high length: width ratio. These high performing CWs and VDs removed N for less than USD 37 kg^{-1} DIN (AUD 50 kg^{-1} DIN), less than the end-of-catchment benchmark for the Great Barrier Reef of USD 110 kg^{-1} DIN (AUD 150 kg^{-1} DIN). When adequately located, designed, and managed, treatment wetlands can be cost-effective and should be adopted for reducing N in tropical and subtropical Australia.

Keywords: cost-effectiveness metric; eutrophication; Great Barrier Reef; macrophytes; nitrogen; total suspended solids; treatment systems

1. Introduction

Nitrogen (N) pollution in waterways is one of the most significant environmental challenges of our times, threatening the wellbeing of humanity [1]. More than a century ago, the discovery of artificial N fixation (the Haber–Bosch process) paved the way to industrial fertiliser manufacture, allowing for large-scale food production [2]. Simultaneously, fossil fuel combustion has contributed to elevated levels of reactive N in the atmosphere [3,4]. The increase in N in waterways due to agricultural and urban runoff, atmospheric deposition, and sewage discharge has caused large scale degradation of aquatic ecosystems [4].

Improved water management, soil conservation, optimised fertiliser application, and enhanced point source treatment and control are essential to address this environmental problem. These activities can be complemented with "treatment systems", which are engineered and designed to intercept, slow down, and remove pollutants from surfaces and groundwaters.

Treatment wetlands are one of the most common treatment systems used to reduce N; they have various applications, such as treating leachate during water treatment operations and reducing fertilizer runoff from surface and groundwaters. Treatment wetlands simulate the N removal conditions of natural palustrine wetlands [5]. These conditions include high vegetation cover and waterlogged soils that are low in oxygen and rich in carbon [6]. These characteristics allow N removal to occur mainly through denitrification, which is the bacterial process that reduces nitrate (NO_3^-) to nitrite (NO_2^-), nitric oxide (NO), nitrous oxide (N_2O), and finally to gaseous N_2. Vegetation uptake can also contribute to N removal; however, it only accounts for less than 10% of the total, and unless harvested, is just temporary storage [7,8]. In some regions, treatment wetlands are five times more effective at reducing NO_3^- loads than land management strategies, including optimisation of fertiliser application and land retirement [9]. Treatment wetlands are also commonly used as the tertiary stage for treating municipal and industrial wastewater. They provide a final effluent polishing before discharge into receiving waterways [10]. Thus, significant investments worldwide have been directed towards constructing treatment wetlands for improving water quality in urban and agricultural landscapes [11].

In tropical and subtropical Australia, there is uncertainty around the cost-effectiveness of treatment wetlands for N mitigation in different land uses [12]. A standardised assessment is necessary to determine an accurate and comparable cost-effectiveness (CE) performance metric of treatment wetlands in this region. This metric can provide a cost (USD/AUD) per unit of reduction in a load (kg) of pollutant (e.g., N) and help justify publicly funded investments. CE metrics are used as an alternative to cost-benefit analysis because they overcome the need to estimate the monetary value of the benefit perceived by society, which is difficult to measure [13,14]. Despite its usefulness, only a limited number of studies have used CE metrics to evaluate the performance of treatment wetlands. With few exceptions (e.g., [15]), most have been conducted in temperate regions (e.g., [16]).

In Queensland, Australia, dissolved inorganic nitrogen (DIN) load reduction targets have been set for agriculture-dominated catchments adjoining the Great Barrier Reef [12]. Additionally, state regulations of total nitrogen (TN) are in place for all industries limiting point source discharge (e.g., sewage treatment plants; through the Environmental Protection Act, 1994). These regulations require innovative solutions that are efficient but also economically viable. In this meta-analysis, we compiled information on the cost and effectiveness of treatment wetlands in Queensland. First, we analysed the N (DIN, TN) removed by these treatment wetlands as changes in concentration (mg L^{-1}) between inlet and outlet and as annual reductions of N (kg ha^{-1} yr^{-1}). Then, we calculated their cost-effectiveness by collating data on the costs incurred in their design, construction, and maintenance, along with their effectiveness in N removal. The aims of this research were (1) to investigate if treatment wetlands in Queensland are efficient at removing N, (2) to find environmental factors associated with the N removal efficiency, and (3) to determine whether treatment wetlands are cost-effective. The results from this project are highly relevant when considering investments for improving water quality in tropical and subtropical Australia. Especially for regions where N pollution is an acute stressor of globally significant coastal ecosystems, such as the World Heritage-listed Great Barrier Reef and the Moreton Bay Ramsar site.

2. Materials and Methods

2.1. Study Sites

We collated data from 14 treatment wetlands ranging in size from 0.3 to 10 ha: eight are constructed treatment wetlands (CW) designed to treat agricultural runoff, two are vegetated drains (VD) receiving agricultural runoff, and four are constructed treatment wetlands designed to polish treated effluent as part of sewage treatment plant operations (STPW, Table 1, Figure 1). All treatment wetlands are in Queensland, northeast Australia. The VDs and CWs (1–5) are in the northern part of the state between latitudes 17°17′ and 18°01′ in the region known as the Wet Tropics, which has a humid tropical climate with a mean annual rainfall of 4548 mm and mean minimum and maximum temperature of 19.4 and 27.9 °C, respectively (1881–2020, Innisfail Station, Australian Bureau of Meteorology, ABM, 2021). The remainder of the CWs are within the Dry Tropics and Mackay–Whitsundays regions, between latitudes 19°24′ and 21°09′, which have mean minimum and maximum annual temperatures of 19.4 to 26.5 °C and 18.0 to 29.2 °C, respectively, and lower rainfall with 948 and 1595 mm, respectively (Ayr, 1951–2021 and Mackay City, 1959–2020, ABM, 2021). Finally, the STPWs are in the subtropics between latitudes 26°34′ and 28°00′, with a mean annual rainfall of 1090 mm and mean annual minimum and maximum temperatures of 15.8 and 25.4 °C, respectively (Brisbane City, 1889–2020, ABM, 2021).

The CWs were constructed by private companies and regional natural resource management organisations. The VDs were constructed and maintained by a sugarcane farmer, and the STPWs were constructed and operated by three local wastewater management authorities. The treatment wetlands were constructed and monitored at different intervals since 2008 by their various owners and organisations, although for CW4, there was no water quality monitoring, only modelling and denitrification experiments. This resulted in diverse monitoring methodologies, water quality parameters, and records of costs incurred for each site (see Tables S1 and S2 for details on each site). Data were standardised to enable analysis and inter-site comparisons as described below and in Supplementary Materials.

Figure 1. Location of treatment wetlands analysed in this project in Queensland, Australia. Circles show the location of constructed treatment wetlands (CW), triangles of vegetated drains (VD) and squares of sewage treatment plant wetlands (STPWs). The pictures are examples of a CW in Russell River catchment, a VD in Tully River catchment (Wet Tropics), a CW in Barratta Creek (Dry Tropics), and an STPW in the Mary River catchment (Southeast Queensland).

Table 1. Climate and characteristics of constructed treatment wetlands (CW), vegetated drains (VD), and sewage treatment plant wetlands (STPW). n = number and date of sampling occasions for water quality included in the study; w:c = wetland size to catchment area ratio; vegetation cover = approximate extent (%) during sampling. CW4 had no water quality data. n.a. = data not available.

Climate/Region		Size (ha)	w:c (%)	Vegetation Cover (%)	Characteristics
Humid-tropical/Wet Tropics	CW1 $n = 25$ 2019–2021	1.6	11	>50	Converted drain in sugarcane farm. Groundwater dominated, with a very high length: width ratio.
	CW2 $n = 45$ 2019–2021	1.2	3	<25	A square-shaped wetland in a banana farm. It has a sediment basin at the inlet with a low length: width ratio. Macrophytes includes *Nymphaea* sp. on the edges.
	CW3 $n = 45$ 2020–2021	8.5	2	>50	Large sinous landscape wetland with two inlet points draining sugarcane farms.
	CW4	10	n.a.	>50	Sugarcane paddock converted to wetland; water level regulated by multiple manually operatedoutlet gates.
	CW5 $n = 8$ 2011–2013	2.5	9	<25	System draining a banana farm. Designed with a retention time of two days, very high length: width ratio. Most reeds and sedges planted did not survive and it became dominated by invasive grasses *Urochloa mutica* and *Hymenachne amplexicaulis*.
Semi humid-tropical/ Dry Tropics, Mackay-Whitsundays	CW6 $n = 15$ 2018–2020	1.8	0.5	>50	Square shaped (low length: width) system draining sugarcane. It has two internal berms to increase residence time, a sediment basin, and an outlet wall. Two inlet points. Densely vegetated mostly by *Typha* spp.
	CW7 $n = 32$ 2017–2020	2.1	0.4	25–50	Treatment train system draining sugarcane with multiple ponds and a high length:width ratio. Vegetation community includes *Schoenoplectus* sp., *Lomandra* sp., and *U. mutica*.
	CW8 $n = 9$ 2017–2019	1.3	3	25–50	Treatment train draining sugarcane with with multiple ponds and a high length: width ratio. Vegetation community includes *U. mutica* and *Typha* sp.
Humid-tropical/Wet Tropics	VD1 $n = 5$ 2021	0.3	n.a.	25–50	400 m-long widened drain in sugarcane farm with stagnant water during sampling and high length: width ratio.
	VD2 $n = 5$ 2021	0.5	n.a.	>50	460 m-long, partly widened drain in a sugarcane farm with patches of dense *U. mutica* constant water flow and high length: width ratio.
Subtropical/Southeast Queensland	STP1 $n = 96$ 2020–2021	7.3	n.a.	>50	Surface flow wetland part of a sewage treatment plant; three parallel cells with similar hydraulic loading rate.
	STP2 $n = 348$ 2018–2020	7.5	n.a.	>50	Part of an irrigated forest and treatment system of a sewage treatment plant.
	STP3 $n = 23$ 2015–2017	0.2	n.a.	>50	Surface flow wetland with two parallel cells, part of a sewage treatment plant.
	STP4 $n = 17$ 2015–2016	0.4	n.a.	>50	Surface flow wetland with two parallel cells, part of a sewage treatment plant.

2.2. N Removal

Water quality data was obtained as dissolved inorganic N (DIN = NO_3^- + NO_2^- + NH_4^+ + NH_3), dissolved organic N (DON = total dissolved nitrogen (TDN) − DIN), total N (TN = DIN + DON + particulate nitrogen (PN)), total phosphorus (TP), and total suspended solids (TSS). For CW1, CW2, and CW3, TN did not include PN; thus, it was considered TDN. For CW4, there was no water quality monitoring, and on-site denitrification measurements were used to estimate removal (Table S1). Additionally, we collated information of dissolved oxygen (DO) and pH for inflow and outflow surface water. Water pH was available for all sites except CW7 and CW8, while DO was available for CW1, CW2, CW3, STPW1, STPW2, STPW3, and STPW4. For STPWs, DO was reported as mg L^{-1}, which was converted to %DO assuming a constant water temperature of 25 °C, which is close to the mean annual air temperature for Southeast Queensland. The changes in DO and pH as the water moved through the treatment wetlands were used to investigate if these parameters could explain N removal efficiency.

N removal was estimated as the change in DIN and TN concentration (Δ mg L^{-1}) between the inflow (the point where most water enters the treatment wetland, usually a drain or pipe) and the outflow (the point where water leaves the treatment wetland) [5,10]. All sites had different monitoring procedures (see Table S1 for details); thus, assumptions were made to standardise the data without compromising the accuracy of the results, always selecting the most conservative scenario (less N removal). First, the inflow concentrations were averaged when there was more than one inflow (CW3, CW6). Second, CW1 was highly influenced by groundwater; thus, we used the average of groundwater and surface N concentrations as there was no information on the percentage of contribution for each water source. Because groundwater N concentrations were an order of magnitude higher than surface runoff concentrations in CW1, we may have underestimated the N concentrations and removal efficiency for this site. Groundwater contribution was not quantified for the rest of the sites and was therefore not included; while none of the other sites were groundwater-dominated, we acknowledge this shortcoming of the analyses. Third, the sampling of in- and outflow of CWs and VDs was usually conducted simultaneously; thus, N removal estimations for these sites are likely to be underestimated as this methodology may not have provided the adequate retention time for nutrient removal to be captured. Finally, statistically high outliners in N concentrations (n = 2) were removed from the analyses as contamination was suspected.

The inlet and outlet flow discharge were assessed with different methodologies by the organisations in charge of each site. These included hydrological models, estimations from changes in water depth, and direct monitoring of water flow (see Table S1 for details on each site). For the sites with no flow measurements (CW3 and CW4), the areal rate of denitrification (NO_3^--N removed in mg m^{-2} h^{-1}, [17], and MFAdame, unpublished data) was used to estimate the annual loads of DIN removal.

For sites with no water flows and those with limited temporal sampling (CW1, VD1, and VD2- all sites within the Wet Tropics), rainfall was used as an indicator of water flow through the wetlands. During the wet season (January–June), we assumed that their soils would be waterlogged and that >1 mm of rainfall would induce soil denitrification and, thus, N removal. During the dry season (July to December), we assumed that rainfall events of at least 10 mm were required to achieve similar conditions. The average monthly number of days with rainfall for the past 30 years was obtained from the nearest meteorological station (ABM, 1991–2020). This resulted in 129 days of N removal for CW1 and CW4, and 89 days for CW3, VD1, and VD2. This assumption, although uncertain, likely underestimates N removal, as we know from previous studies [17] and personal observation that at least CW1, CW3 and both VD1 and VD2 flow all year round, albeit with variable N concentrations. Additionally, daily flow monitoring at CW2 (February–June 2020; >130 days), confirmed continuous flow during this period. Finally, CW4 is manually operated so flooding can be controlled for any desired period.

To compare the differences in nutrient concentrations, DO and pH between inflow and outflow, the Mann–Whitney U test was used as the variables were not normally distributed (tested with Shapiro–Wilk and Kolmogorov–Smirnov tests), even with transformations. The Spearman's rank-order was used to determine the relationship between inflow–outflow concentrations, physicochemical parameters, and removal. Stepwise multiple regression was performed after checking for auto collinearity among parameters to assess the influence of DO and pH and removal rates. Values are shown as mean ± standard error. Statistical analyses were performed with SPSS (v24, IBM, New York, NY, USA).

2.3. Cost-Effectiveness

Treatment wetland costs were categorised as "upfront" and "ongoing". Upfront costs were those incurred once during the planning, design, and construction phase. These included consultancy fees for site surveys, soil tests, field mapping, modelling, earthworks, machine hire, materials (including native seedlings for revegetation), and labour. Project management costs were also included during the design and construction phase; these costs were regarded as a lump sum amount. Ongoing costs were recurrent and sporadic post-construction, and occurred until the end of the project evaluation period. i.e., the time for which the wetland is maintained and assumed to remain effective for N removal. Ongoing costs comprised operation, maintenance, and repair costs. For example, weed control, hydraulic structure maintenance, dredging of sediments, structural repairs, and replanting.

Where relevant, in-kind contributions were included in upfront and ongoing costs because we considered in-kind contributions as essential for successful project completion. Development approvals were not included for all treatment wetlands except for CW4 where they were a key component of the upfront costs. We excluded opportunity costs, i.e., the fore-gone gross margin from the previous land use (e.g., sugarcane production). These costs are highly site-specific and were not relevant to all sites; including them would have impeded transferability and cross-site comparison. Additional site-specific costs can be included in future studies where relevant information is available. See details of available cost information for each site in Table S2, and the methodology in Box S1 for revising the cost-effectiveness estimates produced in this study to accommodate additional site-specific costs.

Each treatment wetland was constructed at a different time; thus, each cost item was expressed in the current financial year (FY 2020/21) in Australian dollars (AUD) using the Consumer Price Index (Australian Bureau of Statistics, 2021), and then converted to US dollars (USD) using an exchange rate of 0.73 on 15 November 2021. The costs were aggregated and discounted over a 20-year project evaluation period to produce an annualised equivalent present value cost (APVC; USD yr^{-1}). The APVC was calculated in two steps.

(1) The total present value cost (TPVC) incurred in constructing and maintaining site over the evaluation period (20 years) is calculated by summing upfront costs incurred in year $t = 0$ and discounted future costs incurred in $t = 1, 2, \ldots$ as they arise over the remainder of the evaluation period.

$$\text{TPVC of wetland j} = W_j = C_j + \sum_{t=1}^{T} \frac{M_{j,t}}{(1+r)^t} \tag{1}$$

where:

W_j = TPVC for site j
C_j = upfront costs of construction for site j (t = 0)
$M_{j,t}$ = one-off, annual, and periodic costs for site j incurred in t = 1, 2, . . .
r = real discount rate
T = evaluation period

(2) TPVC (USD, Equation (1)) is converted into an annualised equivalent present value cost (APVC) in USD yr^{-1}:

$$\text{APVC of wetland j} = AW_j = W_j \left[\frac{r}{1-(1+r)^{-T}} \right] \tag{2}$$

Following standard practice, the TPVC was annualised at a 5% real discount rate over an evaluation period of 20 years (e.g., [16]). This timeframe was selected as it represents the period that we are confident the treatment wetland will remain effective, assuming appropriate maintenance and repair. However, the timeframe could be modified if required (e.g., to obtain Reef Credits requiring 25-year lifetime projects, see Box S1). Annualisation of TPVC into USD yr^{-1} directly compares the treatment wetland effectiveness expressed as annual equivalent reductions in kg DIN or TN.

We calculated the cost-effectiveness metric using annualised equivalent present value cost (AW_j; USD yr^{-1}) and the annual DIN or TN load reductions (kg yr^{-1}) as shown by the following ratio [18]:

$$CE_j\left(\text{USD kg}^{-1}\right) = \frac{AW_j\left(\text{USD yr}^{-1}\right)}{N_j\left(\text{kg yr}^{-1}\right)} \text{ for all } j = 1,\ 2,\ \ldots J \tag{3}$$

where:

CE_j = cost-effectiveness for site j

N_j = annual DIN or TN load removal for site j

Sensitivity analyses were conducted with various discount rates (3, 5 and 7% per annum) and evaluation periods (15, 20, and 25 years) to test the robustness of our results. We evaluated cost-effectiveness at the site scale (e.g., [19,20]), with no adjustments for DIN or TN transport to the coast.

3. Results

3.1. N Removal

The inflow DIN concentrations of the CW and VDs (0.1 to 1.4 mg L^{-1}) were lower than those for STPWs, which ranged between 2 and 17 mg L^{-1}. For TN, concentrations ranged from 1.5 to 2.5 mg L^{-1}, except for STPW3 and STPW4, where concentrations were much higher at 14 and 31 mg L^{-1}, respectively (Table 2).

Table 2. Nitrogen inflow concentrations (NH_4^+-N, NO_3^--N, DIN, TN or TDN; mg L^{-1}); and outflow values for dissolved oxygen (DO%) and pH of constructed treatment wetlands (CW), vegetated drains (VD), and sewage treatment plant wetlands (STPW). n.a. = data not available. Values are mean ± standard error. * NO_x^--N.

	Inflow (mg L^{-1})				Outflow	
	NH_4^+-N	NO_3^--N	DIN	TN or TDN	DO (%)	pH
CW1	0.20 ± 0.03	0.53 ± 0.09	0.72 ± 0.08	TDN: 0.93 ± 0.09	66.3 ± 7.3	6.0 ± 0.1
CW2	0.06 ± 0.02	0.09 ± 0.02	0.15 ± 0.02	TDN: 0.48± 0.04	42.1 ± 5.6	6.1 ± 0.1
CW3	0.03 ± 0.00	0.52 ± 0.06	0.54 ± 0.06	TDN: 0.62 ± 0.06	24.4 ± 3.1	6.2 ± 0.0
CW4	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
CW5	0.48 ± 0.12	0.97 ± 0.25 *	1.44 ± 0.31	2.53 ± 0.43	n.a.	7.3 ± 0.1
CW6	0.10 ± 0.06	0.16 ± 0.03	0.21 ± 0.07	1.52 ± 0.24	n.a.	7.7 ± 0.2
CW7	0.14 ± 0.04	0.79 ± 0.28	0.97 ± 0.33	2.86 ± 0.38	n.a.	n.a.
CW8	0.06 ± 0.01	0.27 ± 0.23	0.87 ± 0.60	1.56 ± 0.40	n.a.	n.a.
VD1	0.12 ± 0.02	n.a.	0.13 ± 0.02	n.a.	n.a.	n.a.
VD2	0.07 ± 0.02	n.a.	0.09 ± 0.02	n.a.	n.a.	n.a.
STPW1	0.57 ± 0.12	1.29 ± 0.11	1.86± 0.16	2.46 ± 0.16	110 ± 1.4	7.5 ± 0.1
STPW2	0.06 ± 0.00	n.a.	2.17 ± 0.23	2.81 ± 0.27	47.8 ± 6.0	6.8 ± 0.1
STPW3	7.44 ± 0.53	0.59 ± 0.11	7.90 ± 0.6	13.88 ± 0.74	115 ± 23	7.8 ± 0.1
STPW4	19.12 ± 1.42	0.99 ± 0.14	18.73 ± 1.4	31.21 ± 1.56	88.1 ± 22.0	7.7 ± 0.0

DIN concentrations were significantly higher in the inflow compared to the outflow when aggregating all treatment wetlands (U = 10,944, $Z = -5.0$, $p < 0.001$ for CWs and VDs and U = 1153.5, $Z = -18.2$, $p < 0.001$ for STPWs, Figure 2), suggesting some DIN removal. The difference was significant in CW1, CW3, and VD1 ($p < 0.001$). Overall, CWs and VDs reduced DIN concentrations by 44%, from 0.52 to 0.29 mg L^{-1}, and STPWs reduced them by 91% from 2.3 to 0.2 mg L^{-1} (Figures 2 and 3). For TN, concentrations were higher in the inflow than the outflow for four of the seven CWS analysed, with significant differences for CW1 ($p < 0.001$) and CW8 ($p < 0.05$). When aggregated, the difference was only significant for the STPWs ($U = 13{,}962.5$, $Z = -1.3$, $p = 0.2$ for CWs and VDs, and $U = 1856.5$, $Z = -17.3$, $p < 0.001$ for STPWs; Figure 3), which resulted in TN concentrations reduced by 72% (Figures 2 and 3).

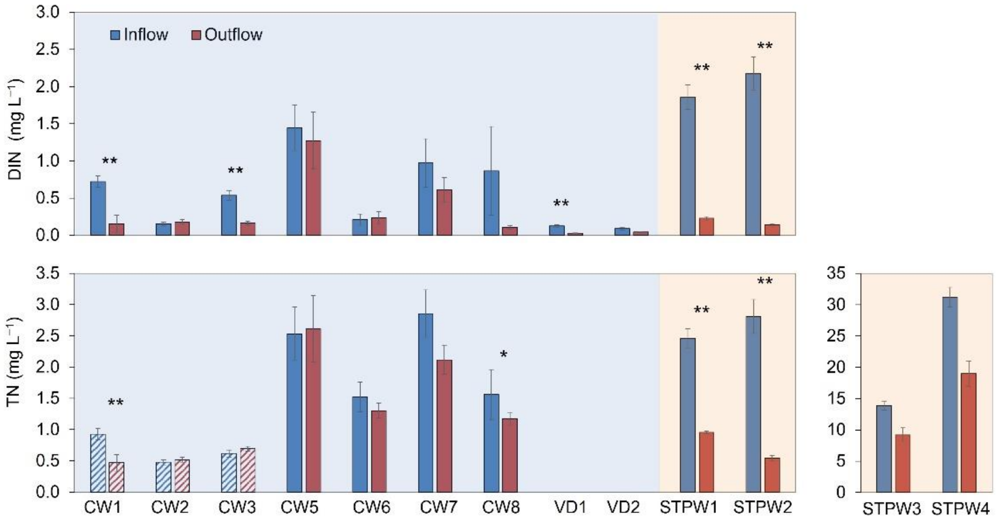

Figure 2. Inflow and outflow concentrations (mg L^{-1}) of DIN and TN in constructed treatment wetlands (CW), vegetated drains (VD), and sewage treatment plant wetlands (STPW) in tropical and subtropical Australia. Hatched bars are sites where TN only includes the dissolved fraction (TDN). TN was not measured in VDs. Significant differences between the inflow and outflow are shown as * $p < 0.05$ and ** $p < 0.001$. Values are mean ± standard error.

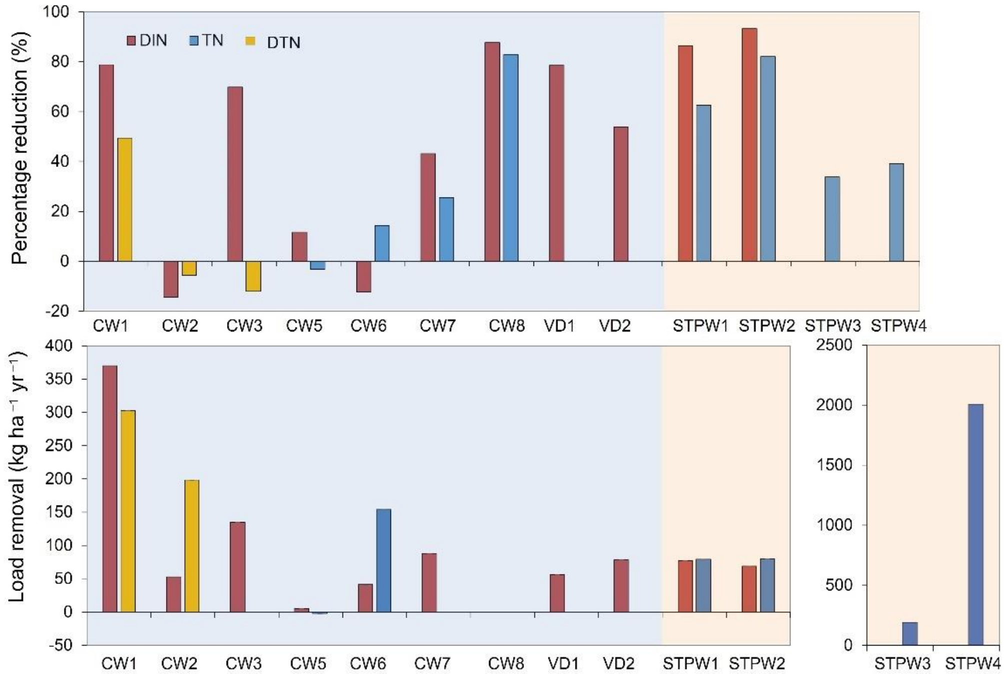

Figure 3. Nitrogen (DIN, TN, and TDN) removal (% and kg ha^{-1} yr^{-1}) by constructed treatment wetlands (CW), vegetated drains (VD) and sewage treatment plant wetlands (STPW) in tropical and subtropical Australia. Values are mean ± standard error.

There was high variability among sites for DIN removal: CW1, CW3, and CW8 achieved the highest DIN reductions, which were consistently above 70%, and the VDs reduced DIN concentrations by 40%. In contrast, CW2 and CW6 did not reduce DIN concentrations significantly during the sampling period. STPW1 and STPW2 had DIN and

TN percentage reductions of 90% and 54%, respectively. However, STPW3 and STPW4, which received TN inflow concentrations an order of magnitude higher than the other sites, had a higher annual load removal but a lower percentage of reduction with 34% (Figure 3). Most N reductions were achieved for the DIN fraction, while the proportion of PN and DON to TN were either similar or higher in the outflow than the inflow (Figure S1).

DIN load removal rate for CWs was 115 $\pm$ 54 kg ha^{-1} yr^{-1}, and for VDs was 67 $\pm$ 11 kg ha^{-1} yr^{-1}. The STPW1 and STPW2 had similar values of 73 $\pm$ 4 kg ha^{-1} yr^{-1}. Additionally, CW1 removed TDN at a rate of 300 kg ha^{-1} yr^{-1}. For TN, STPW1 and STPW2 removed 70 kg ha^{-1} yr^{-1}, while STPW3 and STPW4 had much higher rates at 193 kg ha^{-1} yr^{-1} and 2007 kg ha^{-1} yr^{-1}, respectively (Figure 3). The latter were small systems that received very high TN concentrations of 14 $\pm$ 0.7 mg L^{-1} and 31 $\pm$ 1.6 mg L^{-1}, which were reduced to 9.2 $\pm$ 1.1 mg L^{-1} and 19 $\pm$ 2 mg L^{-1}, in the outflow, respectively.

There were significant differences in the physicochemical parameters of water between the inflow and outflow for the sites investigated. Data on DO were available for three treatment wetlands and the values decreased from the inflow to the outflow in CW1 and CW3 (U = 624, Z = $-$0.654, p < 0.001). Overall, in CWs, the DO decreased from 69 $\pm$ 4.8 to 43 $\pm$ 3.6% (U = 2781.5, Z = $-$4.0, p < 0.001); and in STPWs, DO decreased from 80 $\pm$ 1.7 to 73 $\pm$ 2.5% (U = 17,379.5, Z = $-$2.3, p < 0.05; Figure 4). The difference was significant for STPW1 and STPW2 (p < 0.001).

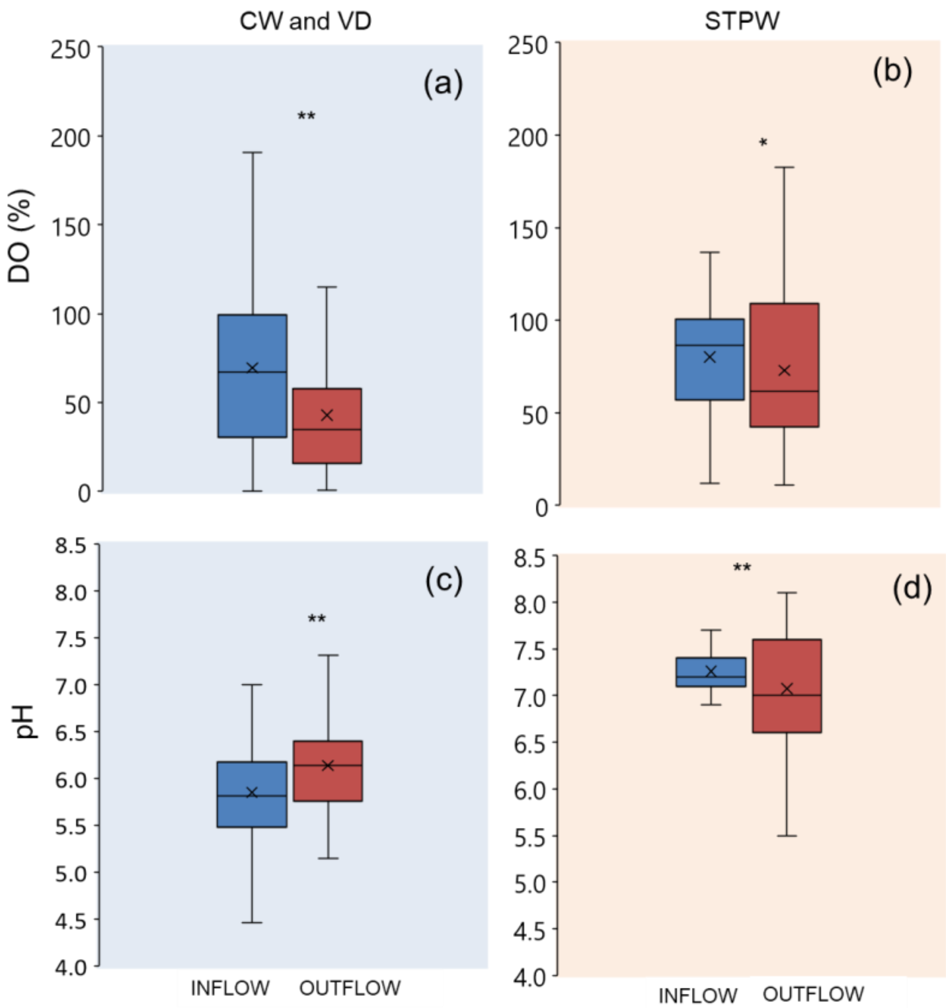

Figure 4. Difference between the inflow (blue) and outflow (red) in pH and dissolved oxygen (DO%) for (**a**,**c**) constructed treatment wetlands (CW) and vegetated drains (VD) and (**b**,**d**) for sewage treatment plant wetlands (STPW). Significant differences between the inflow and outflow are shown as * p < 0.05 and ** p < 0.001.

Water pH significantly increased as it flowed through the CWs from 5.9 ± 0.1 to 6.1 ± 0.1 (U = 4518.5, Z = −3.5, p < 0.001) with significant differences in CW1, CW2, and CW3 (p < 0.05). However, it decreased in the STPWs from 7.3 ± 0.0 to 7.1 ± 0.0 (U = 16,141.5, Z = −4.3, p < 0.001; Figure 4) with significant differences in STPW1 and STPW2 (p < 0.001).

3.2. Factors Associated with N Removal

The removal of DIN, TN, TP, and TSS was positively associated with their inflow concentrations (DIN: $r_s(344) = 0.92$, TN: $r_s(236) = 0.84$, TP: $r_s(336) = 0.61$, TSS: $r_s(344) = 0.49$; $p < 0.001$; Figure 5 and Figure S2) for all treatment wetlands and species except for TSS in STPWs. Removal peaked at higher concentrations, and when concentrations in the inflow were very low, removal was not detected. Thresholds for removal in CWs and VDs were 0.2 mg L^{-1} of DIN, 0.7 mg L^{-1} of TN, 0.1 mg L^{-1} of TP and 10 mg L^{-1} of TSS. For STPWs, thresholds were 0.2 mg L^{-1} of DIN, 0.8 mg L^{-1} of TN and 0.2 mg L^{-1} of TP (Figure 5).

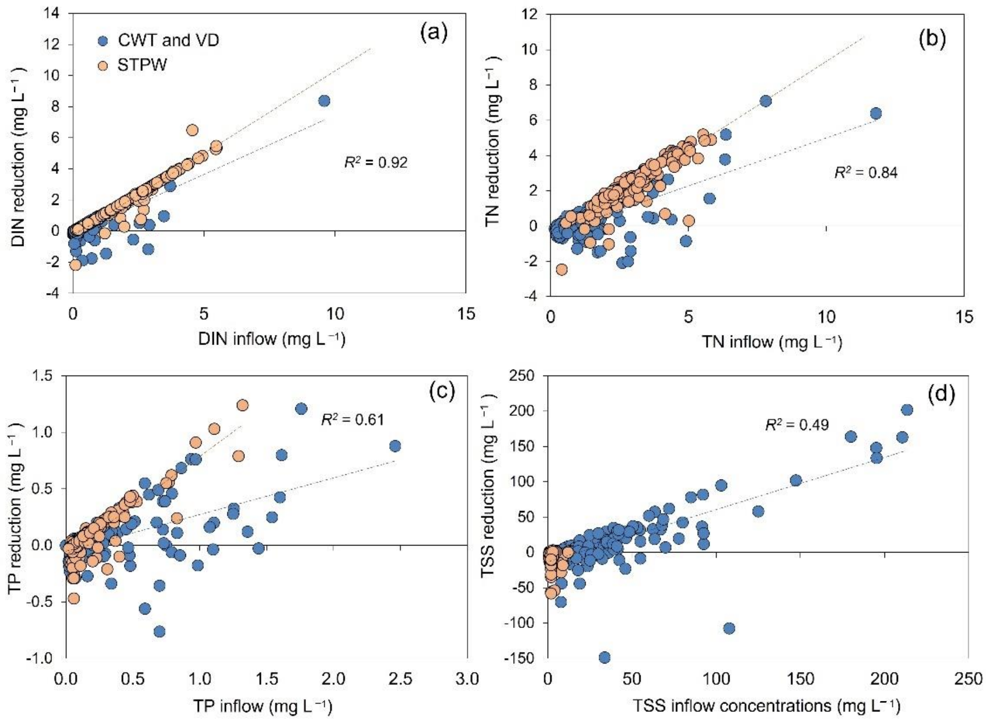

Figure 5. Correlations between (**a**) DIN, (**b**) TN, (**c**) TP, and (**d**) TSS reduction in concentration (mg L^{-1}) and inflow concentrations in constructed treatment wetlands (CW), vegetated drains (VD) and sewage treatment plant wetlands (STPW) in tropical and subtropical Australia. Correlations are for all treatment wetlands aggregated and are significant (p < 0.001).

Finally, inflows with high DIN also had high TN ($r_s(279) = 0.86$, $p < 0.001$) and low TSS concentrations ($r_s(371) = -0.48$, $p < 0.001$), while inflows with high TSS, had high TP ($r_s(371) = 0.31$, $p < 0.001$). Sites with high TSS had lower DIN and TN removal and vice versa ($r_s(350) = -0.31$, $p < 0.001$ and $r_s(354) = -0.2$; $p < 0.001$; Figure S2) (Figure 5). Therefore, conditions that favour TSS and TP removal (e.g., sedimentation) appear to contrast with those that favour N removal (denitrification).

The pH and DO in the outflow explained some of the N removal of the treatment wetlands analysed. For the STPWs, higher DIN and TN removal was associated with lower pH in the outflow (DIN: $r_s(199) = -0.39$; $p < 0.001$ for DIN, and: $r_s(189) = -0.41$; $p < 0.001$ for TN), with higher removal achieved when pH was between 6 and 7. Finally, higher removal in all treatment systems occurred when DO in the outflow was between 40 and 70%.

3.3. Cost-Effectiveness

In present value terms, the annualised cost per hectare for CWs and VDs ranged from USD 2809 for VD2 to USD 31,445 for CW2 (Table 3). The cost of reducing one kg of DIN by CWs and VDs ranged between USD 17 and USD 596 while reducing one kg of TN via an STPW ranged between USD 81 and USD 747. Four treatment systems (CW1, CW3, CW4, and VD2) removed one kg of DIN for less than USD 37, followed by another two treatment systems (CW7 and VD1) that reduced it for less than USD 74.

Table 3. Annualised present value cost (APVC) and cost-effectiveness (CE) for constructed wetlands (CW), vegetated drains (VD) and sewage treatment plant wetlands (STPW) in tropical and subtropical Australia. CE was estimated with a 5% real discount rate per annum over a 20-year evaluation period. n.a. = data was not available. Sites with CE below the USD110 benchmark are highlighted in grey. Values in brackets are in AUS dollars.

	APVC USD ha^{-1} yr^{-1}	DIN Removed kg ha^{-1} yr^{-1}	TN Removed kg ha^{-1} yr^{-1}	CE USD kg DIN^{-1}	CE USD kg TN^{-1}
CW1	6225 (8527)	370	303	17 (23)	21 (28)
CW3	3064 (4197)	135	n.a.	23 (31)	n.a.
CW4	7370 (10,096)	250	n.a.	30 (40)	n.a.
VD2	2809 (3848)	78	n.a.	36 (49)	n.a.
CW7	5558 (7614)	88	n.a.	63 (87)	n.a.
VD1	4164 (5704)	56	n.a.	74 (101_	n.a.
CW6	4996 (6844)	41	154	121 (166)	33 (45)
CW5	4941 (6769)	11	13	452 (619)	368 (504)
CW8	5813(7963)	n.a.	n.a.	n.a.	n.a.
CW2	31,445 (43,076)	53	198	596 (817)	159 (217)
STPW4	162,773 (222,976)	n.a.	2007	n.a.	81 (111)
STPW2	68,938 (94,436)	135	158	510 (699)	437 (599)
STPW1	57,343 (78,551)	77	79	741 (1015)	722 (990)
STPW3	143,740 (196,904)	n.a.	193	n.a.	746 (1023)

Cost-effectiveness (USD kg^{-1} DIN) generally improved (decreased) as DIN removal per hectare increased (Figure 6), with the three most cost-effective wetlands (CW1, CW3, and CW4) removing > 100 kg DIN ha^{-1} $yr^{-1.}$ The APVC per hectare (USD ha^{-1} yr^{-1}) remained relatively unchanged as the area of the wetland area increased, suggesting that economies of scale were not evident across the range of wetland sizes in our study (0.3 to 10 ha; Figure 6). Cost-effectiveness also did not improve as APVC per hectare was reduced (Figure 6). Thus, larger wetlands were not cheaper per hectare, and the cost-effectiveness of wetlands for N removal did not correlate with their cost per hectare. Thus, optimising N removal performance, not reducing area or minimising costs, appears to be critical for improving cost-effectiveness.

The sensitivity analysis indicated that cost-effectiveness decreased as the discount rates increased for a given evaluation period (Tables S3 and S4). More extended evaluation periods improved cost-effectiveness at each discount rate. For example, the cost-effectiveness of CW1 for a 20-year evaluation period ranged between USD 15 and USD 19 kg^{-1} DIN for discount rates of 3 and 7%, respectively (Table S3). However, the differences caused by evaluation periods and discount rates were relatively small compared to the effects of the wetland performance for improving cost-effectiveness.

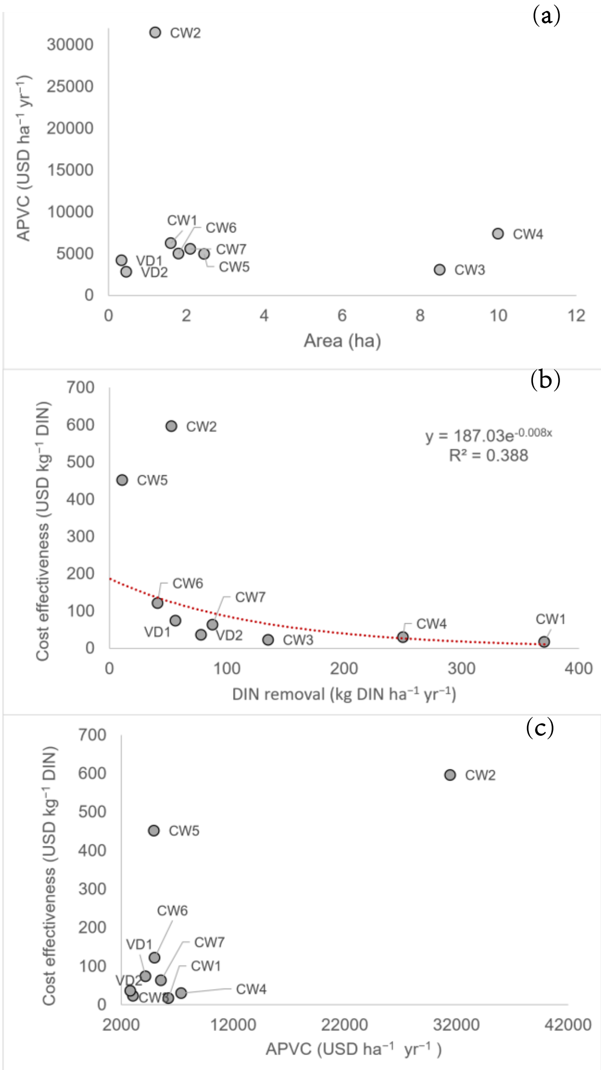

Figure 6. Variation in (**a**) APVC (annualised present value cost per annum, USD ha^{-1} yr^{-1}) against treatment wetland area (ha), and (**b**) cost-effectiveness (USD kg^{-1} DIN) against DIN removal, and (**c**) cost-effectiveness against APVC of constructed treatment wetlands (CW) and vegetated drains (VD) in agricultural catchments of the Great Barrier Reef.

4. Discussion

4.1. N Removal

Treatment wetlands in tropical and subtropical Australia analysed in this meta-analysis removed DIN and TN from the water column when they complied with certain characteristics. In agricultural settings, the best-performing CWs and VDs were those with high DIN concentrations (>0.2 mg L^{-1}) and extensive vegetation cover. In these conditions, DIN concentrations could be reduced by >78% (CW1, CW8, mean for all CW and VD of 44%). For STPWs, high concentrations of DIN and TN in all sites, along with dense vegetation and managed flows, resulted in high DIN (mean for all sites was 90%) and TN (>35%) reductions. These results are similar to other studies, which have found that the high productivity and high temperatures in tropical constructed wetlands favour N removal [21]. Reductions of treatment wetlands in other temperate and tropical locations were also similar, ranging from 39 to 81% for DIN and 31 to 75% for TN [7,10].

Improved performance of the treatment wetlands was also explained by pH, DO, and TSS concentrations. Higher N removal occurred when outflow water was pH neutral (6–7) and when DO concentrations were between 40 and 70%, conditions that favour denitrification [22]. Additionally, sites with high TSS concentrations, which were not managed through sedimentation basins before reaching the treatment wetland, had low DIN removal. This is likely due to sediments smothering and limiting the establishment of macrophytes and nitrifier–denitrifier microbial communities [23,24]. In catchments with high TSS runoff, it may be necessary to design and construct sediment basins as a first step treatment (e.g., Wetland*info* sediment basins: https://wetlandinfo.des.qld.gov.au/wetlands/management/treatment-systems/for-agriculture/treatment-sys-nav-page/sediment-basins/ accessed on 22 November 2021) to improve N removal.

Finally, hydraulic efficiency is crucial to improve retention time [25]. This can be achieved by having a rectangular-shaped wetland instead of squared ones, with a width:length ratio of at least 1:3 [26]. A low hydraulic efficiency may limit the performance of the treatment wetland to a sedimentation basin, where only TSS and TP, but not N, would be removed [25] (Figure 7).

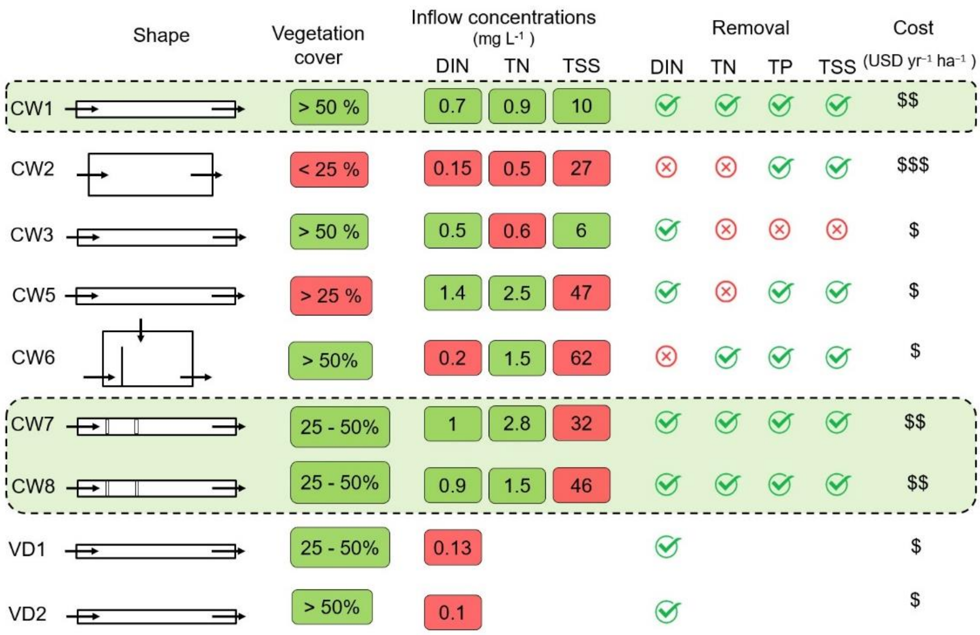

Figure 7. Summary of parameters associated with efficient DIN, TN, TP, and TSS removal and cost (APVC, annualised present value cost) of constructed treatment wetlands (CW) and vegetated drains (VD) in tropical Australia. Sites with a tick mark are those where removal was observed. $ = <USD 5000; $$ = >USD 5000 <USD 10,000; $$$ = >USD 10,000.

A limitation of this study was the diversity of monitoring approaches used and a lack of flow monitoring at some sites, which meant assumptions needed to be applied for estimating N removal loads. Future monitoring programs should include detailed flow measurements to improve load estimations. In addition, water sampling at the inlet and outlet should be timed to account for the retention time in the wetland so that the same water is sampled. Additionally, in the Great Barrier Reef catchments, the first few runoff events after fertiliser application contain the highest DIN concentrations [27]; thus, monitoring should be concentrated and designed to target these events. Finally, groundwater flows were unknown for most sites but likely important [28] and should be incorporated in future studies.

Nevertheless, our assumptions were conservative, with load removals between 11 to 370 kg N yr^{-1}, which are lower than the potential of these sites based on denitrification measurements (170 to 1051 kg N yr^{-1} [17]). Our removal load estimates are also lower than the median of 930 kg N yr^{-1}, from 230 treatment wetlands analyses in a recent review [5], but within the lower range of their estimations (17–5840 kg N yr^{-1} [5]). We expected to have lower loads in our sites as the N concentrations of the CW and VDs were lower (0.5 to 3 mg L^{-1}) than those for other treatment wetlands (0.2 to 100 mg L^{-1} of TN [5]). Because N removal rates were the main factor driving cost-effectiveness (the higher the N loads removed, the more cost-effective), we are confident that our cost-effectiveness estimations are either appropriate or underestimated.

4.2. Cost-Effectiveness

Previous studies in tropical Australia are scarce, but we compared our results with two other studies. The first study reported costs between USD 650,000 and USD 820,000 ha^{-1} (FY2020/21) for establishing small treatment wetlands in the Great Barrier Reef catchment, with costs including site preparation, construction, and ongoing management [29]. A second study reported upfront costs incurred in constructing farm-level drainage lagoons (0.02–2.5 ha) in the Wet Tropics with a mean construction cost of USD 34,117 ha^{-1} (FY2020/21, [19]). In comparison, we estimated that the upfront and ongoing cost of CWs and VDs (5% discount rate; 20-year evaluation period) ranged between USD 35,000 and USD 392,000 ha^{-1} with mean and median costs of USD 98,000 and USD 62,300 ha^{-1}, respectively (FY2020/21). These costs are similar to those reported by the second study, but considerably lower than the costs used by the first. An essential feature of this study, in contrast to others, is that it combines for the first time real (not modelled or estimated) incurred upfront and ongoing costs of treatment wetlands to calculate cost-effectiveness.

In terms of cost-effectiveness, a benchmark of USD 110 kg^{-1} DIN (AUD150 kg^{-1} DIN) at end-of-catchment has been recommended for the Great Barrier Reef [30]. Suppose we assume a surface-water transport of 0.8 from the treatment wetland to the Great Barrier Reef lagoon. In that case, the cost-effectiveness for four of the CWs is less than USD 46 kg^{-1} DIN, and for six of the wetlands would be less than USD 92 kg DIN at the end of the catchment. Both figures are well below the proposed end-of-catchment benchmark. These values are also within the lower range of the cost-effectiveness calculated for bioreactors at USD 55 kg^{-1} DIN, an alternative treatment system technology [31].

The cost-effectiveness methodology applied here is consistent with other evaluations in Reef catchments (e.g., [18,19]). A further advantage of our cost-effectiveness calculation is that it allows proponents to adapt costs and timeframes for alternative site scenarios (as explained in Box S1). An important result from this study is that cost-effectiveness was highly driven by the capacity for N removal of the treatment wetland, not its area or the cost incurred per hectare. For example, the 10 ha CW4 had a high per hectare cost but was the third most cost-effective system (USD 29 kg^{-1} DIN removed); whereas the VD2, with only 0.5 ha, had the lowest per hectare cost but was the fourth most cost-effective system (USD 36 kg^{-1} DIN removed).

For STPWs, which were in peri-urban settings, costs were much higher than in agricultural settings. There are two key reasons for this, first, the STPW are designed conservatively and oversized to ensure they achieve their regulatory discharge loads under any circumstances. Second, STPWs are also designed to have sufficient capacity throughout their life expectancy (at least 20 years) for expected population increases, e.g., loads in STPW2 inflow are predicted to double within the next decade. Thus, many of the STPWs analysed in this study were underloaded compared to their ultimate design capacity and would be expected to become more cost-effective as flow and loads increase. The construction of the STPW has provided many co-benefits, including public access for recreation and improved biodiversity in this peri-urban setting [32]. Similarly, other treatment wetlands studied have provided important amenity, water re-use and environmental co-benefits (e.g., CW7 and CW8). The monetary value of these co-benefits was not included in our analysis. In future research, the value of co-benefits could be estimated by market and non-market valuation methods and subtracted from the costs to provide a more comprehensive cost-effectiveness evaluation [33].

Monitoring and ongoing management are key activities that need to be financed and properly implemented to ensure continuous N removal from treatment wetlands. We recommend establishing a monitoring protocol to standardise water quality measurements and upfront and ongoing cost recording, together with a standardised methodology for calculating cost-effectiveness. This will further improve the cost-effectiveness values obtained in this study, allowing comparisons across a broader range of N-mitigation strategies, including agricultural practice change, land-use change and other treatment systems.

5. Conclusions

Treatment wetlands in tropical and subtropical Australia can achieve effective DIN and TN removal when complying with certain characteristics. N-removal occurs when DIN and TN concentrations in the inflow are >0.2 mg L^{-1} for DIN and >0.7 mg L^{-1} of TN. Additionally, removal is highest when vegetation cover is >50%, when the wetland has a length:width ratio of at least 3:1, and when TSS in the inflow are either low, or managed through sedimentation basins.

The costs of designing, constructing, and maintaining treatment wetlands in agricultural settings of the Great Barrier Reef catchments are lower than previously assumed. These wetlands, when effective, can provide a cost-effective option for N-management, delivering DIN removal for less than the benchmark cost-effectiveness of USD 110 kg^{-1} DIN (AUD$150 kg^{-1} DIN) at end-of-catchment recommended for Great Barrier Reef catchments.

For STPW in urban settings, costs were higher, but besides their high capacity for N removal, they can provide additional co-benefits, which were not included in these analyses. Cost-effectiveness of agricultural and STP treatment wetlands would be improved further if effective maintenance and repair prolong operational lifetime beyond 20 years and if the value of additional co-benefits is incorporated.

In the future, a standardised monitoring and cost recording methodology is recommended to improve cost-effective estimations. Importantly, detailed, and long-term measurements of water flows will improve load calculations. Despite uncertainties, our results are conservative and provide clear evidence that treatment wetlands should be included in the mix of options for reducing N pollution in tropical and subtropical Australia.

Supplementary Materials: The following are available online at https://www.mdpi.com/article/10.3390/w13223309/s1, Figure S1: Inflow and outflow concentrations (mg L^{-1}) and contribution to total nitrogen (TN) of the different species: particulate N (PN), dissolved organic N (PON) and dissolved inorganic N (DIN) for treatment wetlands. There were no DON measurements for CW1, CW2 and CW3. CW = constructed wetlands, VD = vegetated drains and STPW= sewage treatment plant wetlands; Figure S2: Correlation ($p < 0.01$) among inflow water parameters (DIN, TN, TSS and TP) and removal (Δ mg L^{-1}), Table S1. Considerations, limitations and uncertainties in nitrogen (N) load reduction estimations for treatment wetlands: constructed treatment wetlands (CW), vegetated drain (VD), and sewage treatment plant wetlands (STPW); Table S2: Summary of cost data categories provided by project proponents for inclusion in the cost-effectiveness analysis. A '✓' indicates data are available and a '✗' indicates data are not available. Project management costs were unavailable for STPW1, STPW2, STPW3 and STPW4 (shown as shaded cells in grey); Table S3: Cost-effectiveness analysis under varying discount rates and evaluation periods for constructed treatment wetlands (CW) and vegetated drains (VD) in tropical Australia. APVC = annualised present value cost, CE = cost-effectiveness metric. Currency is in Australian Dollars (A$); Table S4: Cost-effectiveness analysis under varying discount rates and evaluation periods for sewage treatment plant wetlands (STPW) in subtropical Australia. Currency is in Australian Dollars (A$), Box S1.Adjusting base cost-effectiveness to include site-specific costs.

Author Contributions: E.K., S.H. and J.C.R.S. conceptualized, collected, analysed the data, and wrote the first draft of the manuscript; M.G. and C.W. established the methodology, conceptualization, and administration of the project; C.B., L.O., K.A., M.S. and S.L. provided data, validation of results, and writing—review and editing; M.F.A. was responsible for funding acquisition, project administration, conceptualization, and writing of first and final draft of manuscript. All authors have read and agreed to the published version of the manuscript.

Funding: This study was funded by the Department of Environment and Science and the Department of Agriculture and Fisheries under the Queensland Reef Water Quality Program. MFA is funded by an Advance Queensland Industry Engagement Fellowship.

Data Availability Statement: Data is provided as Supplementary Materials.

Acknowledgments: Thank you to all the people who contributed information and knowledge to this project, in particular to D. Ah Kee (Jaragun EcoServices), C. Algar and F. George (Terrain NRM), A. Rickert (Burdekin Bowen Integrated Floodplain Management Advisory Committee Inc.), K. Crouch (Unity Water), C. Jackson (Urban Utilities), and M. Bayley (Australian Wetlands Consulting) and N. Waltham and others from the Queensland Government.

Conflicts of Interest: The authors declare no conflict of interest.

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