Vegetated drains for water quality improvement in the Wet Tropics

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For:

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Summary

Nitrogen (N) release from agricultural land is transported through drains and can cause degradation of receiving waters. Drains can be managed to reduce N loads before leaving the farm; however, there are few studies in tropical locations that have addressed this problem. In this study, we investigated the potential of drains to remove nutrients (dissolved inorganic nitrogen, DIN [NO_x^--N + NH_4^+-N] and PO_4^--P) in a sugarcane farm and identified the conditions that promote nutrient removal. The overall goal of the project was to find management actions that could be conducted on-farm to mitigate nutrient leaching from agricultural land.

We worked together with the landholder to sample five drains that had a range of physicochemical and hydrological characteristics. We sampled each of these drains five times during the wet season in April 2021. We conducted denitrification experiments and measured the physicochemical characteristics of each drain as well as their hydrology. Finally, we assessed the isotopes of water flowing through the drains (δD and ^{18}δO) to determine their origin: rainfall/groundwater or runoff.

Our results showed that all the drains had the potential for NO_3^--N removal through denitrification. However, only one drain (D5) had the adequate residence time, measured at water velocities around 0.04 m s^{-1}, for this potential to be realised. The D5 drained a combination of rainfall/groundwater and runoff, which flowed through a patch of shallow water that was densely vegetated. As a result, as water moved through D5, DIN concentrations decreased by 0.05 mg L^{-1} between the inlet and outlet. If these conditions were maintained throughout the year, the equivalent DIN reduction would be 320 kg DIN ha^{-1} yr^{-1}, from which 120 kg would be NO_x^--N.

In conclusion, managing drains to achieve similar conditions as those from D5 could be a strategy to decrease DIN leaching from agricultural land. We identified activities to improve the management of drains, such as modifying drain slope to reduce flow and increase retention time, avoiding stagnation, and creating vegetation "nodes" with dense macrophytes.

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1 Introduction

Nitrogen (N) release from agricultural land is an environmental problem that can cause severe degradation of receiving waters. Nitrogen can be transported through groundwater or through drainage systems that link the farms with the coastal environment (Sharpley et al., 2007; Vymazal and Březinová, 2018). Farmers build channels to drain their farms and increase production. These drains could be managed to improve water quality by modifying their water flow, sediment characteristics, and vegetation cover (Collins et al. 2016, Iseyemi et al., 2016).

Many studies in temperate climates, such as in North America, Europe and Asia, have been conducted to understand how drains can be managed to remove N (Vymazal and Březinová, 2018). Still, studies in tropical regions are very scarce. Vegetated drains can remove nitrogen through denitrification, converting nitrate (NO_3^--N) to N gas (N_2). Rates of denitrification in temperate drains can be substantial, with values ranging from 350 to 1,278 kg of N per ha every year (Castaldelli et al., 2015). The highest denitrification rates are usually found in soils rich in carbon, vegetated, which receive have high NO_3^--N concentrations at relatively slow water flows.

Mitigating N pollution is a management priority in Queensland to protect sensitive freshwater and marine environments, such as the Great Barrier Reef. In recent years, treatment systems have been built to test their capacity for N removal. However, the role of the drainage network in N removal is not fully understood and has yet to be incorporated into current management practices. This project investigated the potential of drains to remove nutrients (dissolved inorganic nitrogen, DIN [NO_x^--N + NH_4^+-N] and PO_4^--P) in a sugarcane farm in collaboration with Angelo Crema, the farm owner. We examined the following questions:

1- What are the conditions that promote denitrification in these drains?
2- How does denitrification within the drain reflect changes in nutrient concentration between the inflow and outflow?
3- How can we manage drains to improve water quality?

2 Methodology

2.1 Study site

The sites were located within the lower reaches of the Tully catchment in a sugarcane farm. We selected five drains along with Angelo Crema. The drains were selected to be of similar length (335 – 460 m) and depth (0.5 – 1.5 m). The measuring points were selected at locations to avoid water inflow into the drains between the inlet and outlet. The drains were selected with a range of physicochemical characteristics (vegetation density, water flow velocity and soil type, Fig. 1,2) as described below:

Drain 1: It is 335 m long and drains a large area of rainforest and a farm. The drain is 2.3 m wide at the inlet with a maximum water depth of 0.55 m. At the outlet, It is 3.3 m wide with a maximum water depth of 0.3 m. The drain has a surface area of 970 m^2. The water is clear, and the drain is fringed by large trees on both sides of the drain. The soil is coarse, light brown with some leaf litter and no macrophytes.

Drain 2: The drain is 335 m long and is the continuation of Drain 1, located 250 m apart. This drain is 3.5 m wide with a maximum water depth of around 1.4 m both at the inlet and outlet monitoring points. This drain has a surface area of 1172 m^2, submerged vegetation and is fringed by few trees. The soil is dark brown, rich in organic matter.

Drain 3: The drain is 460 m long with a slower water flow than Drain 1 and 2 and has a surface area of 1465 m^2. The soil is dark brown, high in organic matter. The inlet of the drain has shallow water with low macrophyte cover and is fringed by few trees. The drain is 2.2 m wide at the inlet with a maximum water depth of 0.45 m. Towards the drain outlet, the water flow slows as the drain gets wider and deeper, and macrophyte cover increases along with the number of fringing trees. At the outlet, the drain is 4.4 m wide with a has a maximum water depth of 0.9 m.

Drain 4: This drain flows around a paddock and is 400 m long. The drain is vegetated at the inlet, with macrophyte cover decreasing towards the outlet; there are no fringing trees. This drain is the widest among the five drains at 7.6 m, with a maximum water depth of 1.6 m and a surface area of 3300 m^2. The water was almost stagnant during sampling. The soil was sandy, light brown, with some organic matter.

Drain 5: This is the main drain of the farm and is 460 m long and a total surface area of 4550 m^2. It runs through 1,900 m of sugarcane before the inlet monitoring point. The drain is 3.6 m wide at the inlet with a maximum water depth of 1.35 m. After the inlet, the water runs through 3,000 m^2 of Paragrass (*Urochloa mutica*), with trees fringe the outlet. At the outlet, the drain is 3.7 m wide with a maximum water depth of 1.5 m. This drain flows all year round. The soil is dense and dark brown.

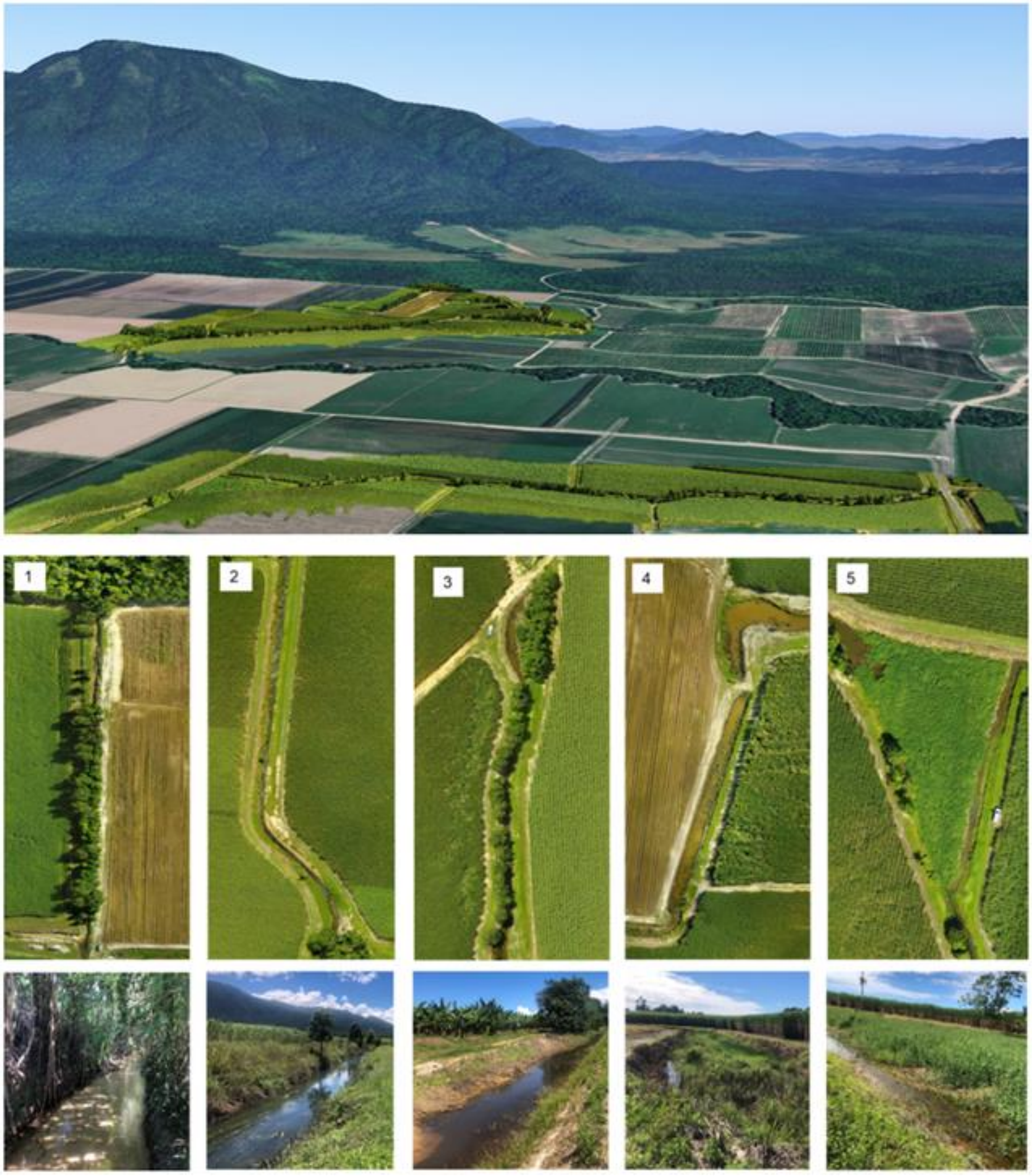

Figure 1. Five drains selected for this study in the Tully River catchment, Wet Tropics, Qld.

The Tully catchment has a tropical climate with monthly mean temperatures ranging from 22 to 34°C (Australia Bureau of Meteorology, ABM, 2019: 1907–2018) and a mean annual rainfall of 2,700mm (ABM, 2019: 1871– 2018). The rivers in this region are characterised by dry periods during winter months and sporadic overbank floods in the summer (between January and May) that inundate the adjacent wetlands from one to 12 days at a time (Karim et al. 2012). Contributions to flow from groundwater are also important (Rasiah et al., 2003). Sampling was conducted between the 3-11 April 2021, during the receding of a large flooding event when 2,707 mm of rainfall fell between January and April (Fig. 2). Ten days before sampling, an additional 196 mm fell, and at the beginning of the sampling, an extra 28 mm fell in the area (ABM, Tully Sugar Mill Station 32042; Fig 2). Temperatures ranged during sampling between a daily minimum of 22.1°C and a maximum of 28.4°C (ABM, Cowley Beach Station 32194).

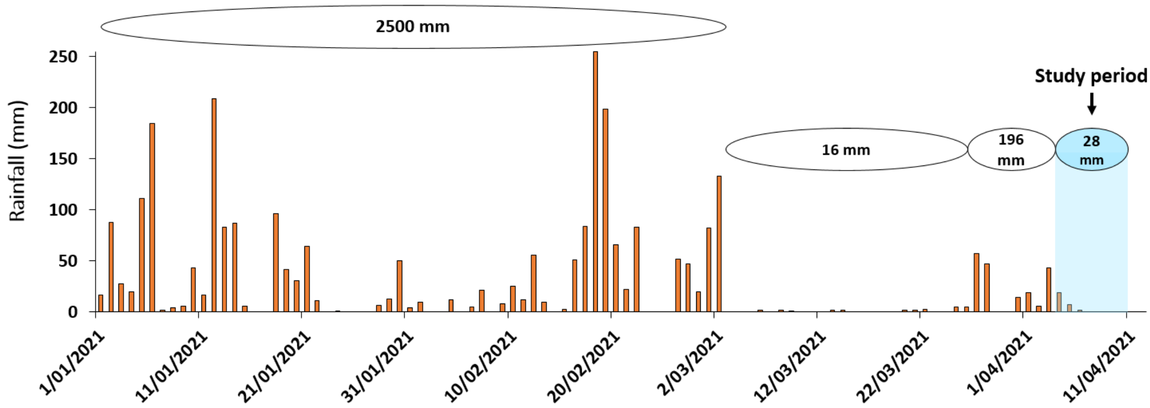

Figure 2. Daily total rainfall (mm), four months prior and during the study period.

2.2 Soil characteristics

We measured within each drain 3-4 points for water temperature, electrical conductivity, and pH with a calibrated water quality meter (ProPlus, YSI meter, OH, USA). Reduction-oxidation (redox) was measured at the sediment-water interface with a redox meter (HQ11d, Hach Pacific, Australia). Surface soil samples (top 5 cm) were collected in triplicate at each site with a plastic 50 mL mini-core. The samples were oven-dried at 60°C for 48 h and weighed, from which bulk density (g cm^{-3}) was calculated. Soil samples were ground and analysed for total nitrogen (N%) and organic carbon (C%) with an elemental analyser (EA-IRMS, Serco System, Griffith University). Before analyses, samples were tested for carbonates with the addition of HCL to the samples to detect bubbling; no reaction was observed. Soil samples were analysed for their grain size by sieving a composing sample of sediment taken from five replicates within each site. The soil was mixed, and a representative subsample of 400 g was dry sieved through four mesh sizes: 150 µm, 250 µm, 500 µm, and 4,000 µm. The soil was separted into six fractions: coarse sand, medium sand, fine sand, very fine sand, silt, and clay (López, 2017).

2.3 Denitrification

We measured denitrification rates for each drain with the isotope pairing technique (Nielsen 1992, Steingruber et al. 2001) adapted to wetlands (Adame et al. 2019). We added enriched ^{15}N-nitrate to the water overlying sediments at a saturating concentration to estimate denitrification rates from the ^{15}N-N_2 gas production. At each site, we collected intact sediment cores of eight cm-depth inside Perspex tubes (4.8 cm internal diameter x 30 cm long). Eight sediment cores were taken at each site. Cores were capped at the bottom, filled with water from each site, placed standing in a rack, and left to equilibrate overnight.

The denitrification experiments were run the next day in large rectangular containers filled with water to maintain a constant temperature (26 and 27°C). The ^{15}N-NO_3^- was added to each core at the beginning of the experiment, topped with water and sealed firmly to maintain low oxygen conditions. A Teflon-coated stirrer bar was connected to the lid, suspended 5 cm above the sediment, and stirred by an external rotating magnet to imitate the natural water movement inside the cores. Three replicates of water samples were obtained before and after enrichment with ^{15}N-NO_3^- to calculate NO_3^--N enrichment during the experiment (ε, Eq. 4).

After 20 minutes of incubation, two cores were sacrificed by adding 1ml of 50% of Zinc Chloride ($ZnCl_2$), which killed the microbes within the sediment. The soil, water and the added $ZnCl_2$ were mixed thoroughly, and three replicates of 10 mL slurries were taken from each core. Samples were transferred to Exetainer vials (Labco, High Wycombe, UK) with 250 µL of 50% w/v $ZnCl_2$. The same procedure was conducted for three cores at two hours and for the other three cores at five hours after the start of the experiment. The gas within each vial was analysed by continuous-flow mass spectrometry for $^{28}N_2$, $^{29}N_2$ and $^{30}N_2$-gas (EA-IRMS, Serco System at Griffith University), their changes in concentration with time were used to estimate denitrification rates (Steingruber et al. 2001; Eq. 1-6). Denitrification rates are reported for ambient light conditions in mg m^{-2} h^{-1} with a detection limit of 0.01 mg N m^{-2} h^{-1}.

1. D_{15}: denitrification from labelled $^{15}NO_3^-$ measured from the production rate of the $^{29}N_2$ and $^{30}N_2$

Equation 1.

$$D_{15} = r_{29} \cdot 2r_{30}$$

Where r_{29} and r_{30} are the production rates of $^{29}N_2$ and $^{30}N_2$, respectively.

2. $D_{14:}$ denitrification from unlabelled $^{14}NO_3$

Equation 2.

$$D_{14} = D_{15} \cdot \frac{r_{29}}{2r_{30}}$$

3. D_t = total denitrification or potential denitrification

Equation 3.

$$D_t = D_{15} + D_{14}$$

4. D_w^{tot}: total denitrification of NO_3^- from the water column

Equation 4.

$$D_w^{tot} = \frac{D_{15}}{\varepsilon}$$

where ε is NO_3^- enrichment during incubation as a result of $^{15}NO_3^-$ additions.

$$\varepsilon = \frac{[NO_3^-]_a - [NO_3^-]_b}{[NO_3^-]_a}$$

where *a* and *b* refer to concentrations after and before $^{15}NO_3^-$ addition.

5. D_w: denitrification from the water column corrected for tracer addition

Equation 5.

$$D_w = D_w^{tot}(1-\varepsilon)$$

6. D_n: Coupled nitrification-denitrification

Equation 6.

$$D_n = D^{tot} - D_w^{tot}$$

2.4 Inflow and outflow water quality and flow measurements

Within each drain, we selected an in and outflow point with the help of Angelo Crema (Fig. 3). For Drain 5, we also selected a second outlet point (hereafter "outlet II"). Water was sampled in duplicates (n = 110 samples) for five occasions at each sampling point during the sampling week. Samples were taken with a new syringe, filtered through a 0.45 µm membrane filter, stored in 25 mL tubes, and frozen before being analysed for nutrients within the next 28 days (colourimetric analyses based on APHA/AWWA/WPCF, 2012; Chemistry Centre, Queensland Department of Environment and Science, Brisbane, Australia). Detection limits (mg L^{-1}) for the nutrient analyses were: 0.002 for NH_4^+-N; 0.001 for NO_x^- -N (NO_3^- + NO_2^-) and 0.001 for PO_4^{3}-P.

The flow rate in each drain was measured at the beginning and the end of the sampling period with an open stream current velocity meter (2100-STDX, Swoffer Instruments, Inc. WA, USA). The device has a minimum velocity detection of 0.003 m s^{-1}. At each sampling point, a transect was selected perpendicular to the drain flow. Each cross-section was divided into vertical subsections, whose area was calculated from its width and depth (Fig. 3). The flow velocity was measured at each subsection (Fig. 4). The volume of water moving down each subsection (discharge; m^3 s^{-1}) was calculated from its area (m^2) times flow velocity (m s^{-1}; Eq. 7). The total discharge of the drain was obtained by summing the subsections (Eq. 8). In cases where the drain was too wide to sample with the velocity meter, water depth was measured for half of the drain's width, and the cross-section was mirrored for the other half.

Equation 7.

$$Discharge = Area \times Velocity$$

Equation 8.

$$Total\ discharge = (Area_1 \times Velocity_1) + (Area_1 \times Velocity_1) + \cdots + (Area_n \times Velocity_n)$$

where *n* is the number of subsections.

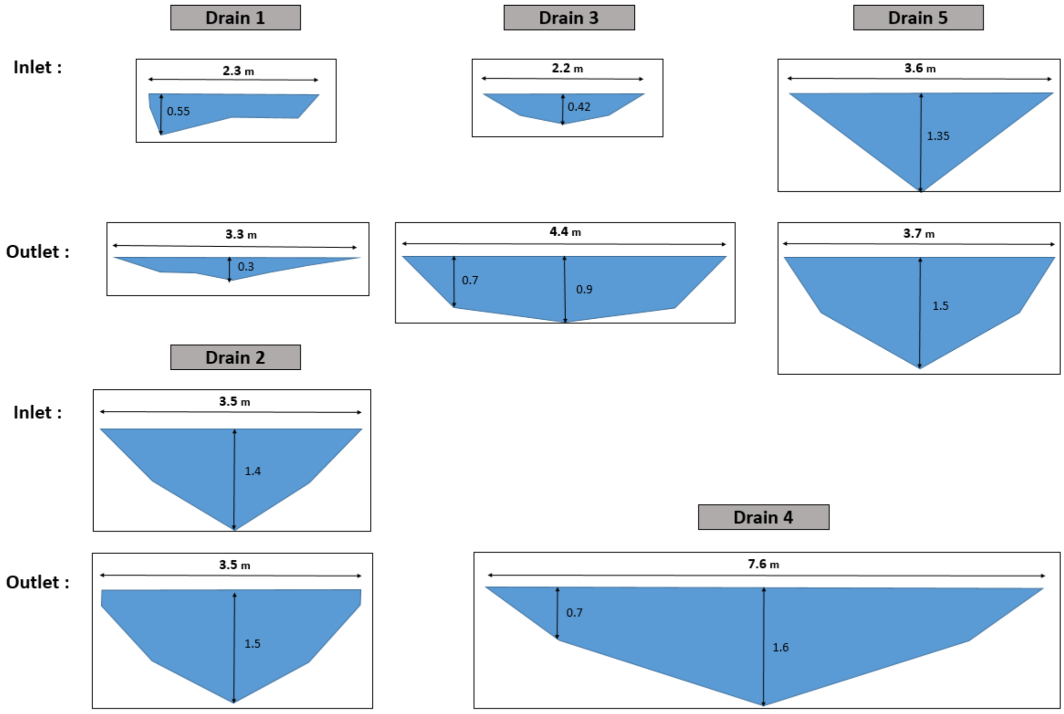

Figure 3. Cross-section area of drains at the inlet and outlet monitoring points. Dimensions are presented in the meter.

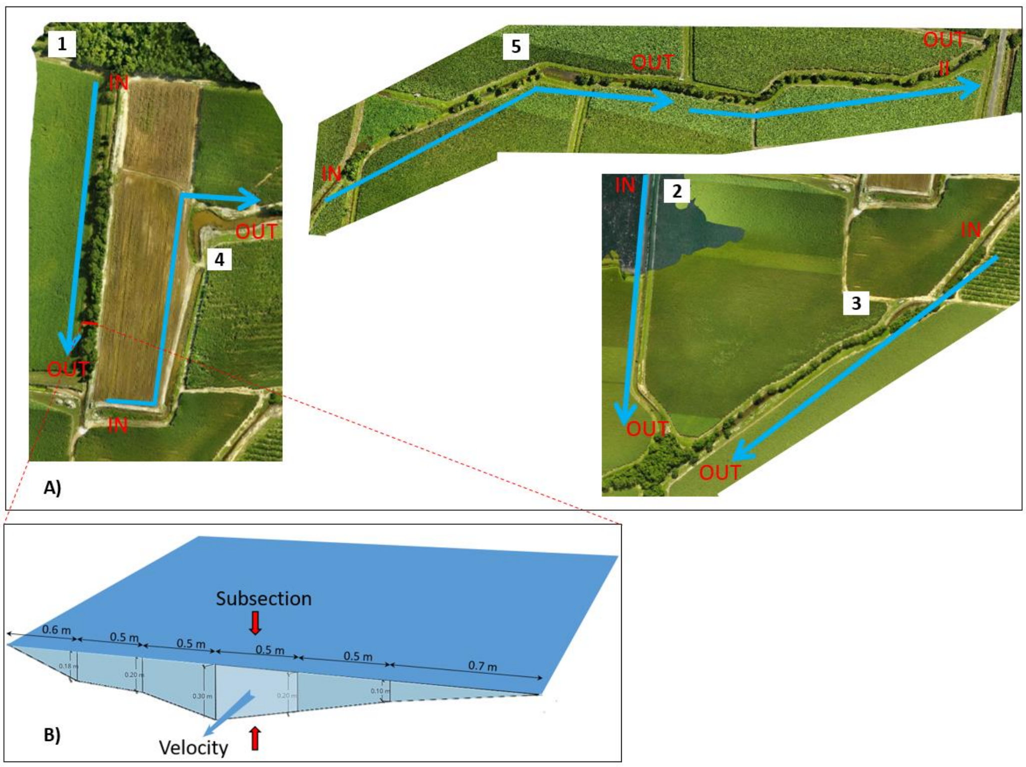

Figure 4. A) Direction of flow and “in” and “out” sampling locations for five drains in this study; B) details of the vertical subsections of Drain 1 to estimate flow discharge.

The removal rate of nutrients in the drain was calculated as changes in concentration (Δ mg L^{-1}): inflow – outflow. The nutrient load reduction per drain area (ha) was calculated from the change in nutrient concentration (mg L^{-1}) and average flow discharge of the drain (m^3 s^{-1}; Eq. 9). The nutrient load reduction was projected over a year (365 days) to provide the hypothetical scenario if a wetland was managed to maintain similar nutrient concentrations and flows throughout the year. Annual nutrient load reductions are shown as kg removal per year per hectare of drain.

Equation 9.

$$Nutrient\ load\ reduction = \frac{Removlal\ rate\ \times\ Flow\ discharge}{Drain\ area}$$

2.5 Groundwater/rainfall vs runoff/evaporation

We took samples in duplicate from the in- and outlet points for each drain to determine the origin of the water moving down the drains. Sampling was done at the beginning and end of the sampling period. The samples were collected in plastic containers that were sealed with minimum airspace and kept refrigerated. Samples were analysed for hydrogen and oxygen isotopes (δD and $\delta^{18}O$) which are useful tracers of different water sources (Adame et al. 2013). The water samples were analysed using a laser-based isotope analyser (LGR, Los Gatos Research, CA, USA). Measurements were calibrated with international standards Vienna Standard Mean Ocean Water (VSMOW), Greenland Ice Sheet Precipitation (GISP), and Standard Light Antarctic Precipitation (SLAP), as well as five manufacturer-supplied standards LGR1-LGR5 (Ahmad et al., 2012). The isotope values had errors (SD) < 2.0‰ for δD and < 0.3‰ for $\delta^{18}O$. We compared the δD and $\delta^{18}O$ with four years precipitation database obtained for the region, with the lowest δD and $\delta^{18}O$ values (< -20‰ δD and ≤ -4‰ $\delta^{18}O$) measured between January and May, and highest values (> -10‰ δD and > -2‰ $\delta^{18}O$) between June to December (Bowen et al., 2019, Munsksgaard et al. 2019). Rainfall has low isotope values and, in general, the stronger the rainfall event, the lower the value (Adame et al., 2013). If rainfall falls and percolates into the groundwater system, it will retain the value of precipitation. However, if water is transported as runoff, evaporation favours the loss of "light" isotopes, (δH and $\delta^{16}O$) which results in the enrichment of heavier isotopes (δD and $\delta^{18}O$), resulting in higher isotopic values in runoff (Fry 2008).

2.6 Data analyses

We tested for significant differences between inflow and outflow concentrations with the non-parametric Mann-Whitney U Test. This test was selected as drains were analysed separately, leading to a relatively low number of data points (n = 5 events: n = 5 drains, n =110 samples). The significance level was set at 0.05, and data are reported as mean ± standard error. Statistical analyses were performed with SPSS (v24, IBM, New York, USA).

3 Results

3.1 Environmental characteristics

The soil was predominately sand (150-500 µm) in all the drains, with the highest contribution (> 70%) in D1, D3 and D5 (Table 1). Site D1 had the highest contribution of gravel (> 4,000 µm) with 7.7 %, while D2 and D4 had the highest contribution of clay and silt (< 150 µm) with 30.8 % and 40.4 %, respectively.

Table 1. Soil grain size (%) of five drains in a sugarcane farm in the Tully River catchment, Wet Tropics.

	< 150 µm	150 – 250 µm	250 – 500 µm	500 – 4,000 µm	> 4,000 µm
Site	Silt and clay & very fine sand	Fine sand	Medium sand	Very coarse sand	gravel
D1	11.8	9.5	12.9	58.1	7.7
D2	26.1	13.2	16.4	43.3	0.9
D3	18.8	12.4	17.9	49.7	1.2
D4	35.9	22.2	12.5	28.7	0.6
D5	6.9	8.6	16.2	63.8	4.5

Soil organic C and N content ranged from 0.1 – 10.7 % and 0.01 – 0.4 %. Sites D1, D4 and D5 drains had lower C (< 1.7 %) and N (< 0.08 %), compared to D2 and D3 with 3.6 ± 0.3 % C and 0.2 ± 0.0 % N and 10.7 ± 1.9 % C and 0.4 ± 0.0 % N, for D2 and D3, respectively (Table 2, Fig. 5). The C:N (molar ratio) ranged between 17 and 24, with the highest values in D1 and D4, and lowest in D2 and D3.

Water temperature during sampling ranged from 23.6 ± 0.1 °C in D1 to 28.4 ± 0.9 °C in D3. Site D1 had the highest pH with 6.9 (neutral), while D3 and D5 had the lowest (4.9). The redox potential was highly variable among drains, ranging from -30.0 mV in D4 (anoxic) to 190.0 mV in D2 (aerobic conditions). Electrical conductivity and water salinity were highest in

D3 and D4 (> 65 mS cm^{-1} and ≥ 0.05 ppt) and lowest in D1 and D2 (≤ 36 mS cm^{-1} and 0.01 ppt; Table 2, Fig. 5).

Table 2. Physicochemical characteristics of five drains in a sugarcane farm in the Tully River catchment, Wet Tropics. C = carbon, N = nitrogen. Conductivity, pH, salinity was measured from the water; redox was measured from the soil-water interface; and C, N and bulk density were measured from the soil. Values are mean ± standard errors for 3- 5 samples for each drain. Values are mean ± standard error.

Site	Conductivity (mS cm^{-1})	Water pH	Salinity (ppt)	Soil-water Redox (mV)	C (%)	N (%)	Bulk density (g cm^{-3})
D1	33.0 ± 0.1	6.9 ± 0.3	0.01 ± 0.0	103.8 ± 4.2	0.1 ± 0.00	0.01 ± 0.0	1.25 ± 0.04
D2	36.1 ± 1.5	5.5 ± 0.4	0.01 ± 0.0	190.0 ± 7.8	3.6 ± 0.3	0.2 ± 0.01	0.84 ± 0.09
D3	65.5 ± 7.0	4.9 ± 0.1	0.05 ± 0.01	136.9 ± 5.5	10.7 ± 1.9	0.4 ± 0.04	0.55 ± 0.08
D4	69.6 ± 5.7	5.3 ± 0.1	0.06 ± 0.01	-30.0 ± 7.4	1.7 ± 0.2	0.07 ± 0.01	0.88 ± 0.04
D5	57.8 ± 2.5	4.9 ± 0.1	0.03 ± 0.0	35.5 ± 19.4	0.9 ± 0.3	0.8 ± 0.03	1.31 ± 0.08

3.2 Groundwater/rainfall vs runoff/evaporation

Water $\delta^{18}O$ and δD values were similar between the in- and outflow of all drains (Fig. 6), and they were all above the local precipitation data, suggesting that most of the water sampled had gone through some level of evaporation (2014-2017; Munksgaard et al. 2019; -9.4 ‰ to -8.4 ‰ and -42.6 ‰ to -36.9 ‰ for $\delta^{18}O$ and δD, respectively). The $\delta^{18}O$ and δD values from D1, D2 were the lowest, D3 and D4 were the highest, and D5 had isotopic values in between. This suggests that drains D1 and D2 had the highest contributions of rainfall/groundwater and that D3 and D4 had the highest contributions of either runoff or that had higher evaporation rates (Huddart et al. 1999). Values for D5 suggest mixing of all sources, consistent with the fact that this is the main drain of the property.

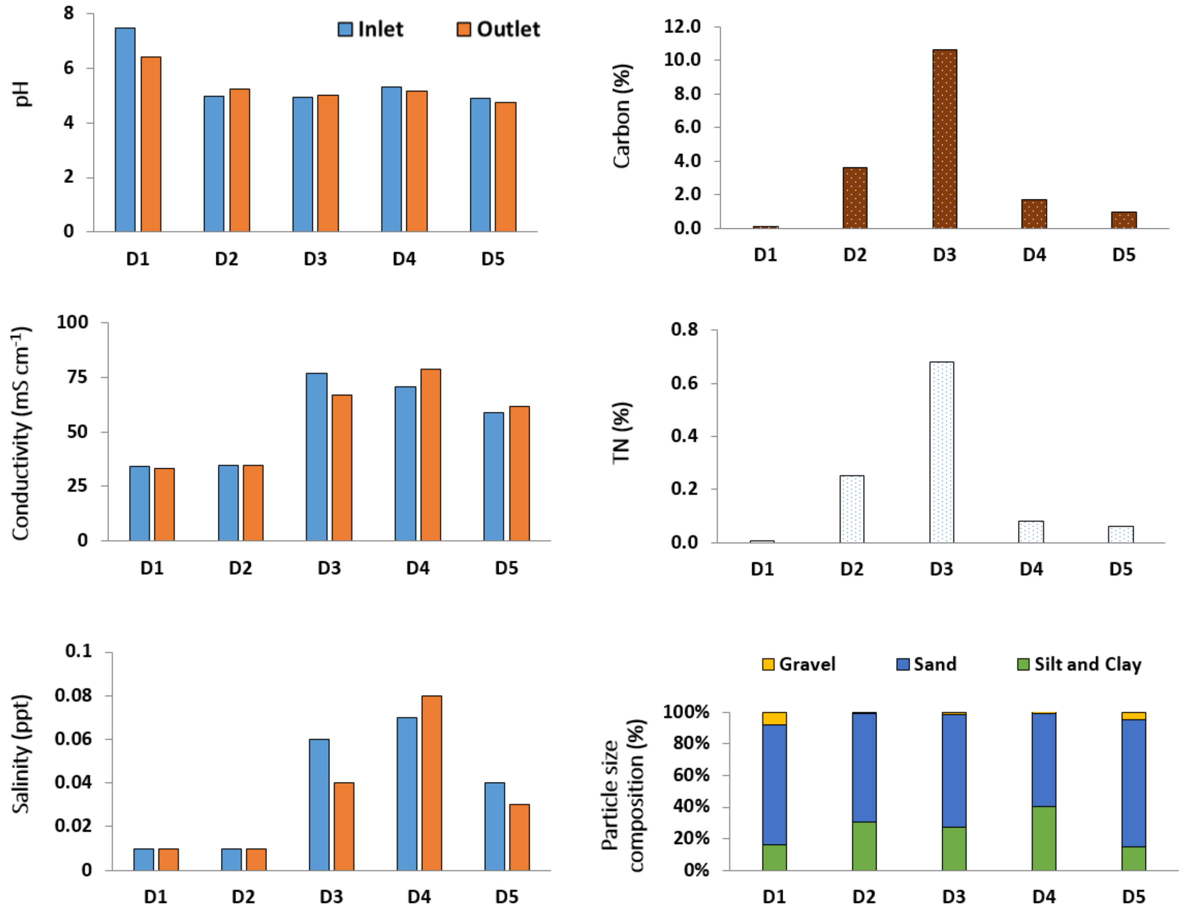

Figure 5. Physicochemical characteristics of surface water from the inflow (blue), outflow (orange) and soil (carbon, C%, total nitrogen, total nitrogen, TN% , and grain size) for five drains.

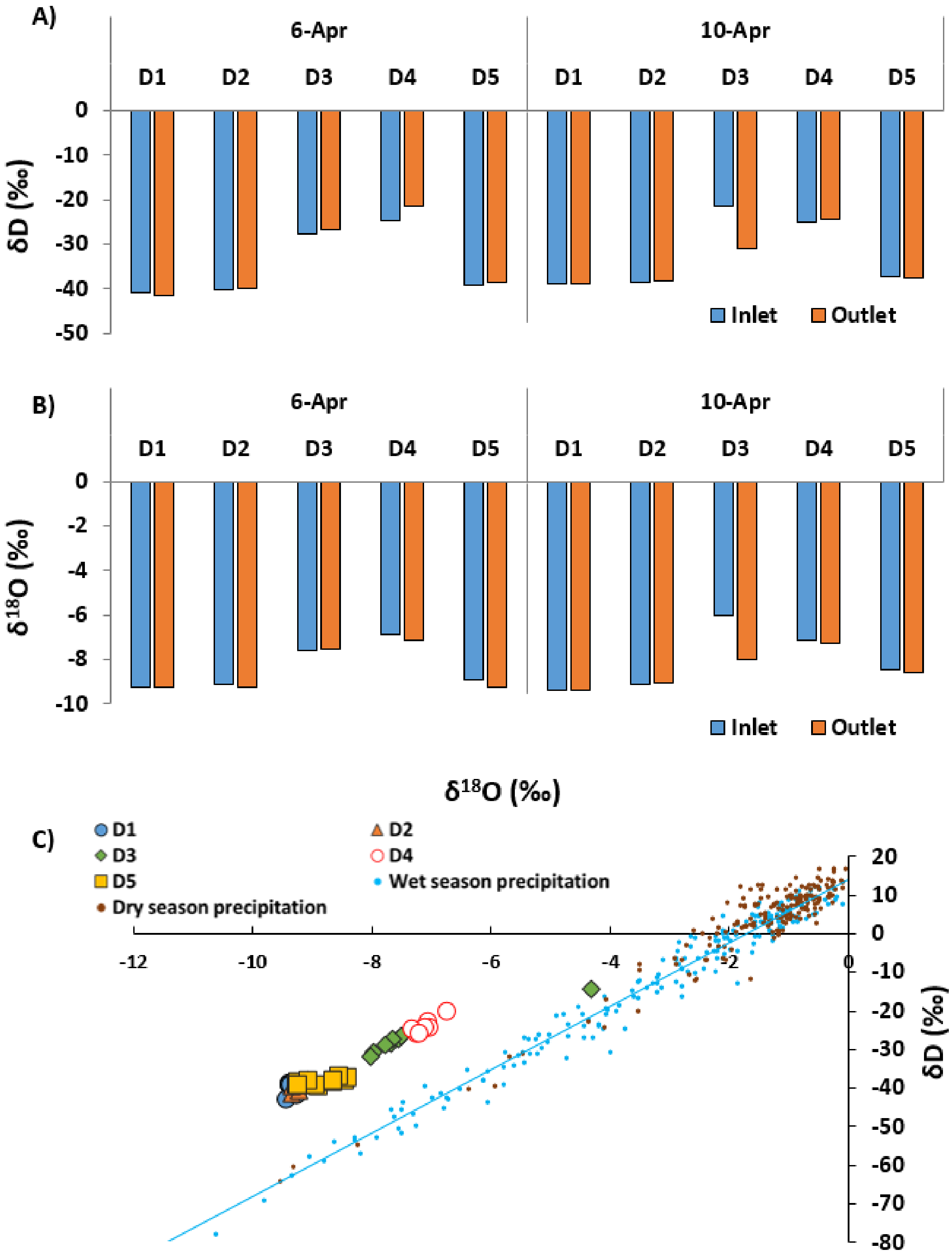

Figure 6. (A) δD and $\delta^{18}O$ values from in and outflow of five drains (B), and comparison of measured values with long term δD and $\delta^{18}O$ precipitation data (Munksgaard et al., 2019). The deviation of water of the drain (higher values) compared to those of precipitation suggest evaporation.

3.3 Denitrification

The water redox was highly variable during the denitrification experiments, ranging from -46 in D3 to 234 mV in D2 (Table 3). The experimental NO_3^--N ranged from 0.06 to 0.2

mg L^{-1} in D5 and D2, respectively, while NH_4^+-N concentrations were lowest in D1 with 0.01 mg L^{-1} and highest in D3 with 0.35 mg L^{-1}.

All the drains showed potential for denitrification with an average rate of 7.8 ± 2.4 mg m^{-2} h^{-1}, with minimum values in D4 (1.8 mg m^{-2} h^{-1}) and maximum values in D2 (15.4 mg m^{-2} h^{-1}). Coupled nitrification-denitrification (D_n) accounted for most of the denitrification (> 80%) of all drains except in D5, where D_n accounted for 60%. Site D5 was the only site where a considerable amount of denitrification (40 %) was fuelled directly from NO_3^--N in the water column (D_w^{tot}).

Table 3. Denitrification rates of five drains (D1-D5) and experimental conditions of redox, NO_x^--N, and NH_4^+-N concentrations in the water column.

Dt = total denitrification; Dwtot = denitrification of nitrate from the water column; Dn = coupled nitrification-denitrification; Dw = denitrification corrected for nutrient additions during experiments; ε = NO_x^--N enrichment.

	D1	D2	D3	D4	D5
D_t	9.6 ± 4.9	15.4 ± 6.8	3.2 ± 0.4	1.8 ± 1.8	9.1 ± 4.6
D_w^{tot}	1.2 ± 1.0	1.3 ± 0.6	0.7 ± 0.6	0.3 ± 0.3	3.7 ± 1.9
D_n	6.8 ± 5.6	14.1 ± 6.2	2.5 ± 2	1.5 ± 1.5	5.5 ± 2.8
D_w	1.0 ± 0.8	1.2 ± 0.5	0.5 ± 0.4	0.3 ± 0.3	2.2 ± 1.1
ε	0.15	0.08	0.22	0.18	0.4
Experiment conditions					
Redox -EXP (mV)	189 ± 34	233.5 ± 17.7	-45.7 ± 13.1	163.1 ± 11.2	-32.7 ± 18.7
NO_X^--N -EXP (mg L^{-1})	0.18 ± 0.0	0.2 ± 0.01	0.14 ± 0.0	0.13 ± 0.0	0.06 ± 0.02
NH_4^+-N -EXP (mg L^{-1})	0.01 ± 0.003	0.04 ± 0.02	0.35 ± 0.03	0.02 ± 0.006	0.05 ± 0.04

3.4 In and out water flows

Flow velocity was highest at D1 with 0.64 m s^{-1}, followed by D2 with 0.12 m s^{-1}, and D3 and D5 with 0.04 m s^{-1}. In D4, water was almost stagnant with water velocity approaching zero (0.003 m s^{-1}, Fig. 7). Flow discharge was highest at D1 and D2, with values around 0.36 m^3 s^{-1}, followed by D5 with 0.09 m^3 s^{-1}. The lowest discharge rates were measured for D3 and D4 at 0.02 m^3 s^{-1} (Table 4, Fig. 7). Flow discharge was highest at the beginning of the sampling period, reflecting the 3-day antecedent rainfall of 68 mm (Fig. 2).

Table 4. Flow velocity (m s^{-1}), discharge (m^3 s^{-1}) and residence time (min) of five drains during April 2021. Values are means of measurement within the inflow and outflow of the drains.

	Water velocity (m s^{-1})	Discharge (m^3 s^{-1})	Residence time (min)
D1	0.64 ± 0.15	0.36 ± 0.02	9
D2	0.12 ± 0.02	0.37 ± 0.01	46
D3	0.04 ± 0.02	0.02± 0.01	450
D4	0.003 ± 0.0	0.02 ± 0.0	2000
D5	0.04 ± 0.0	0.09 ± 0.01	456

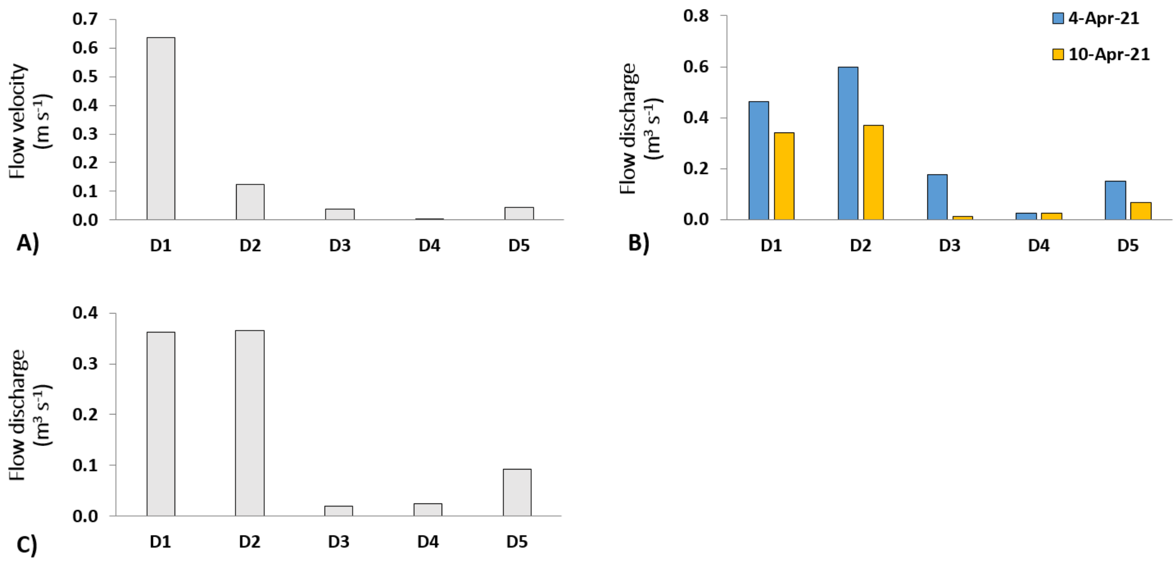

Figure 7. (A) Flow velocity (m s^{-1}), (B) flow discharge (m^3 s^{-1}), and (C) mean flow discharge for five drains during April 2021.

3.5 Inflow versus outflow water quality

3.5.1 D1 and D2

The NO_x^--N concentrations of in D1 and D2 ranged between 0.05 to 0.07 mg L^{-1} and were similar between inflow and outflow, suggesting no net removal ($p > 0.05$, Fig. 8). These drains had low NH_4^+-N and PO_4^--P concentrations at around 0.01 mg L^{-1} and 0.005 mg L^{-1}, respectively, similar in the in- and outflow ($p > 0.05$, Fig. 9,10). D2 is the continuation of D1 (see Fig. 1); thus, considering the whole drain as one, during the last three days of sampling, there was a slight NO_x^--N removal and NH_4^+-N release ($U = 11$, $Z = -2.959$, $p < 0.01$).

3.5.2 D3

The NO_x^--N concentrations in the inflow of this drain were the highest of all sites, with values reaching 0.12 mg L^{-1} on the first day of sampling and decreasing to 0.05 mg L^{-1} by the last day (Fig. 8). The NO_x^--N and NH_4^+-N fluxes were variable (Fig. 8, 9) and PO_4^--P, concentrations were similar for in and outflow, suggesting no net change ($p > 0.05$, Fig. 10).

3.5.3 D4

The NO_x^--N concentrations in this drain were much lower than the other sites with values ~0.01 mg L^{-1}. The difference in NO_x^--N concentration between the inflow and outflow suggested a small release of NO_x^--N by 0.006 ± 0.0 mg L^{-1} ($U = 9.5$, $Z = -3.085$, $p < 0.01$, Fig. 8). This drain had comparatively high NH_4^+-N concentrations with 0.12 ± 0.02 mg L^{-1}, and acted as a sink, reducing NH_4^+-N by 0.11 ± 0.02 mg L^{-1} (84 % reduction; $U = 0$, Z = -3.788, $p < 0.001$, Fig. 9). D4 also reduced PO_4^--P by 0.001 ± 0.0 mg L^{-1} ($U = 0$, $Z = -3.785$, $p < 0.001$, Fig. 10).

3.5.4 D5

This drain had the highest removal potential for both NO_x^--N and NH_4^+-N, although these were in relatively low concentrations (< 0.04 mg L^{-1}). The D5 reduced on average concentrations by 0.02 ± 0.005 mg L^{-1} (73 % reduction) and 0.03 ± 0.01 mg L^{-1} (31 % reduction), respectively (Fig. 8, 9), with maximum reductions by 0.03 mg L^{-1} and 0.07 mg L^{-1}, respectively. Overall DIN was reduced in D5 by 0.05 ± 0.02 mg L^{-1}. By including the second outflow sampling site, 400 m below (outlet II), DIN concentration was reduced by 0.07 ± 0.02 mg L^{-1} over 850 m. This drain also reduced PO_4^--P concentrations by 0.05 ± 0.02 mg L^{-1} (Fig. 10).

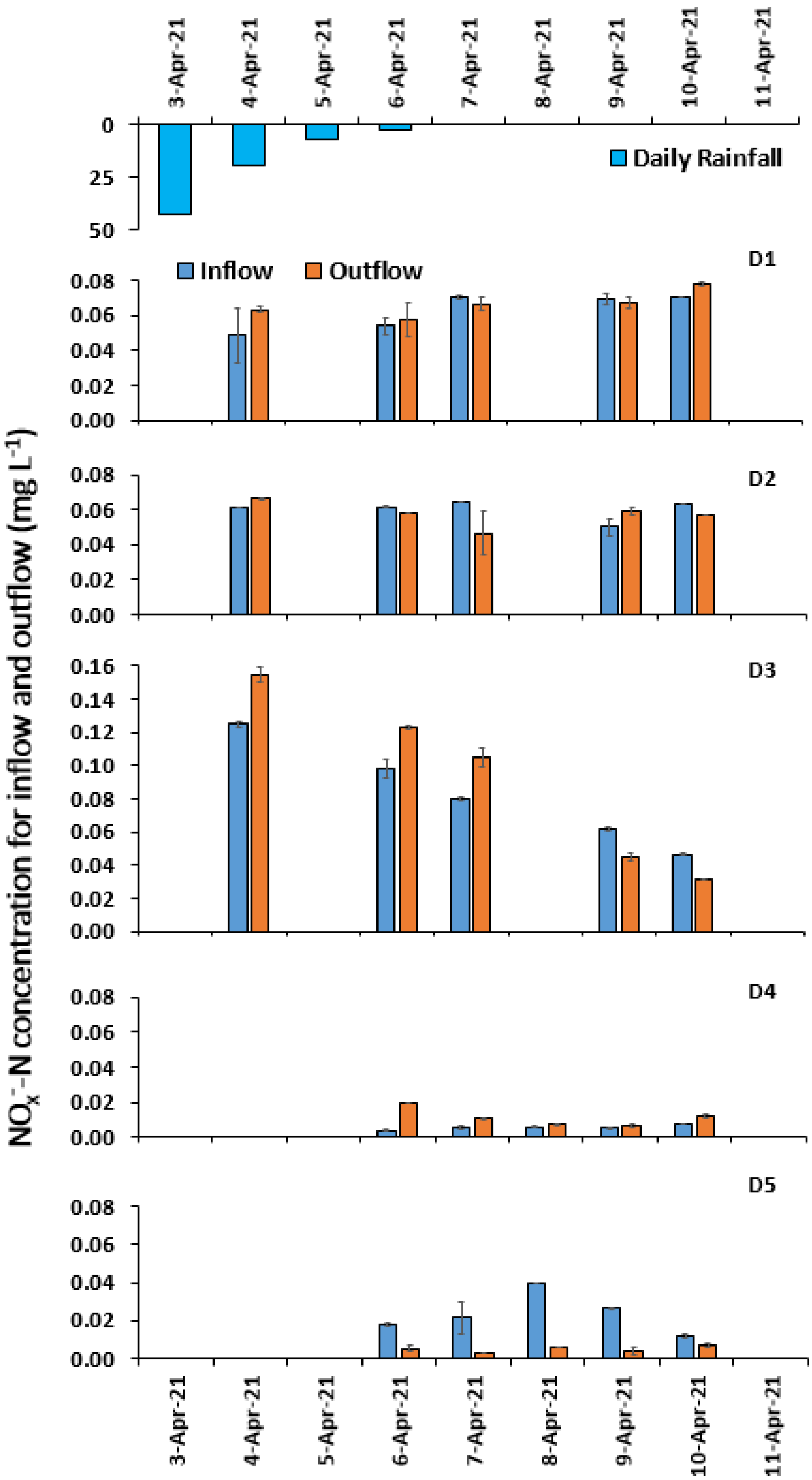

Figure 8. Rainfall (mm) and NO_x^--N concentration (mg L^{-1}) in the inflow (blue) and outflow (orange) for five drains (D1-D5)

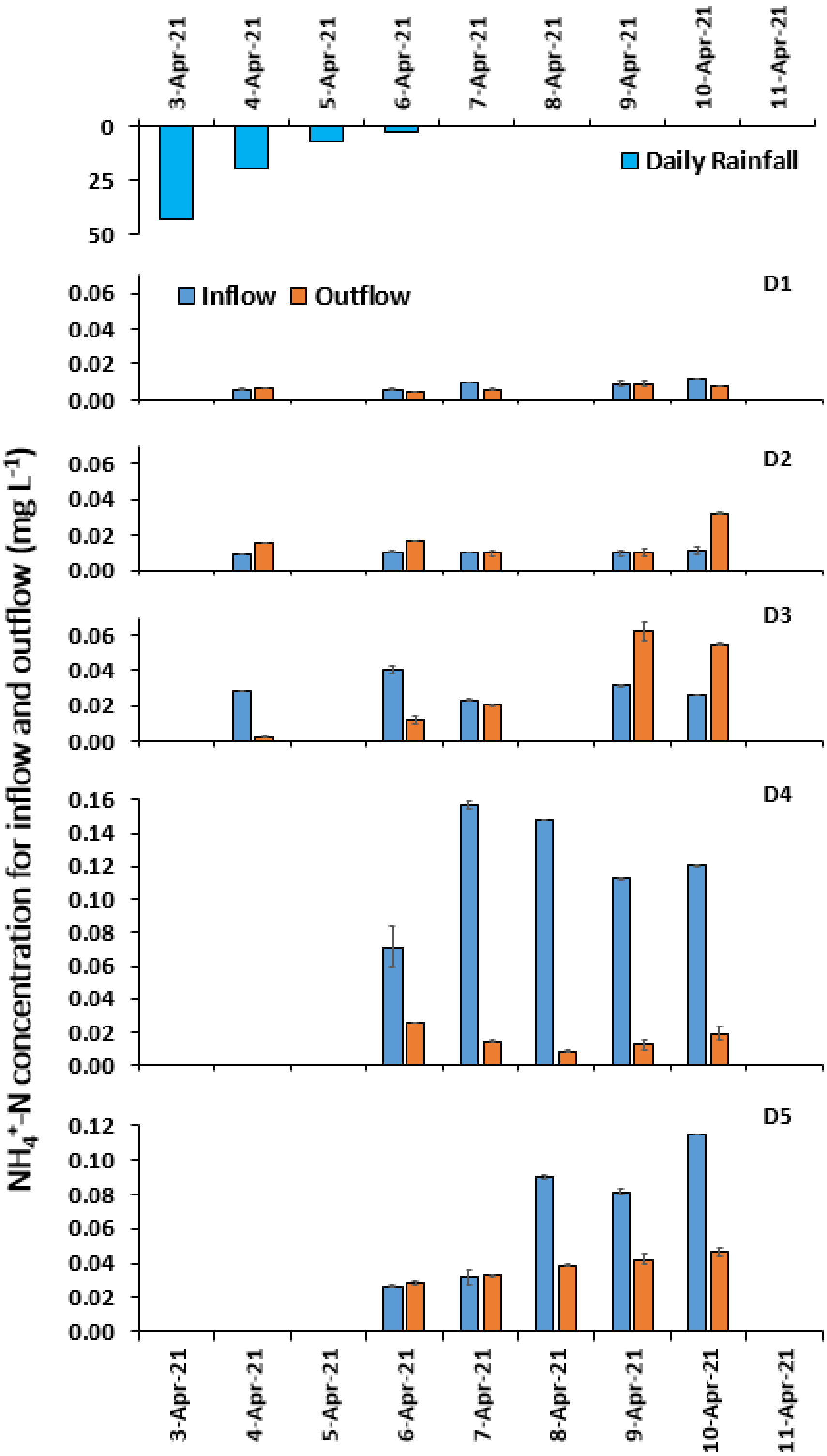

Figure 9. Rainfall (mm) and NH_4^+-N concentration (mg L^{-1}) in the inflow (blue) and outflow (orange) for five drains (D1-D5)

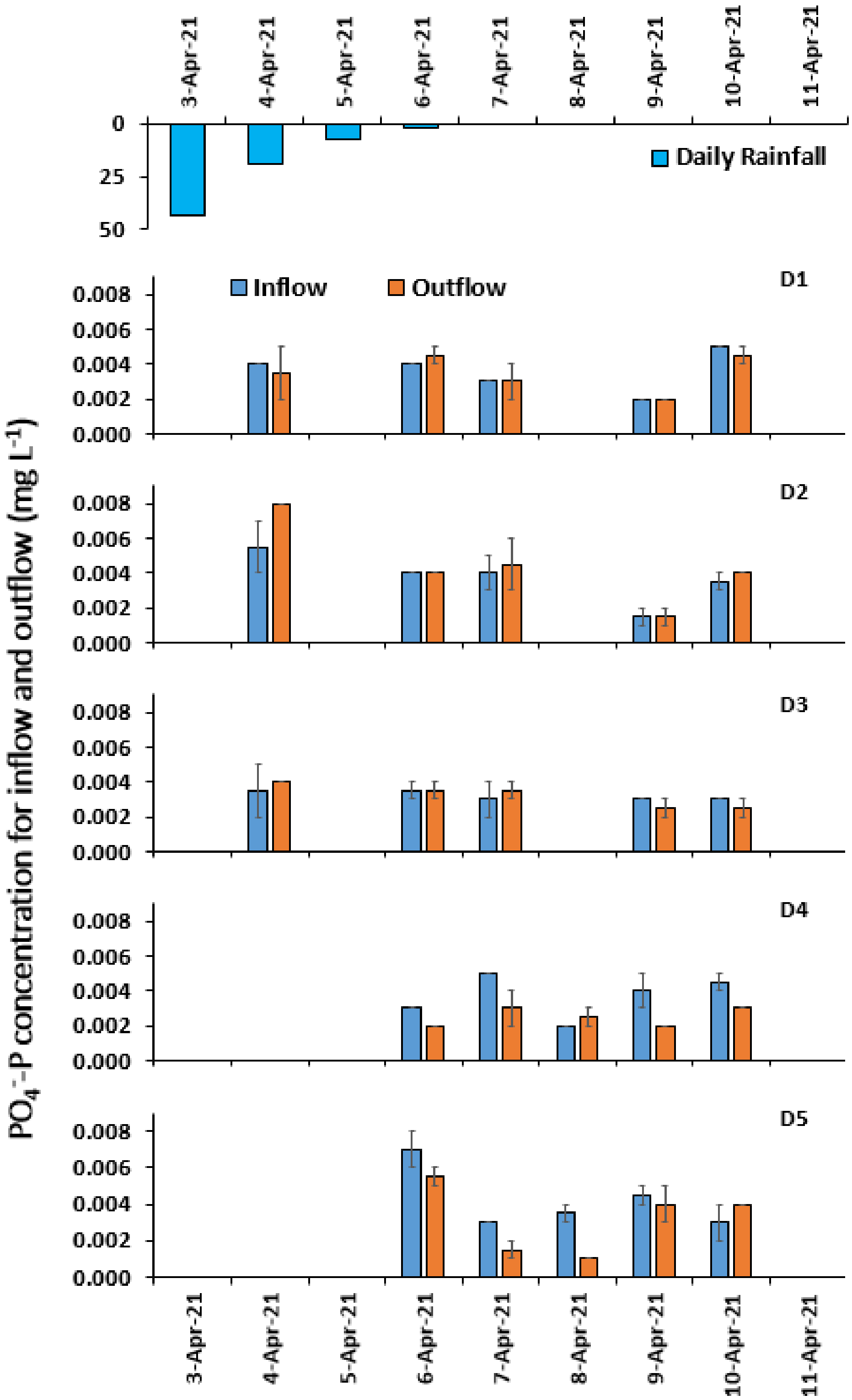

Figure 10. Rainfall (mm) and PO_4^--P concentration (mg L^{-1}) in the inflow (blue) and outflow (orange) for five drains (D1-D5)

4 Discussion and management recommendations

Every drain had the potential for NO_3^--N removal through denitrification, as shown in our experiments, suggesting that they mostly fulfilled the requirements for this process to occur. Denitrification requires four conditions: (1) relatively high NO_3^--N concentrations (> 0.02 mg L^{-1}); (2) high soil organic carbon (> 2%) and nitrogen (> 0.2%); (3) mildly anoxic conditions (-100 to 300 mV) and 4) an established microbial community (Adame et al. 2019, 2021, Mitsch and Gosselink 2007). Only site D1 had much lower C and N (0.1 and 0.01%) for what is optimal for denitrification (Table 2). Despite all the sites having the potential to denitrify, only D5 ("main drain") had the adequate residence time for this potential to be realised, which occurred at water velocities around 0.04 m s^{-1}. Other studies had shown similar results, with N removal in managed agricultural drains enhanced by one order of magnitude when flow velocity increased from zero to 0.06 m s^{-1} (Castaldelli et al., 2018). Importantly, D5 passed through a large patch of densely vegetated Paragrass, which is likely to have enhanced denitrification and plant uptake (Fig. 11).

As the water moved through D5, DIN was reduced by 0.05 mg L^{-1} over the 4550 m^2 of the drain between the inlet and outlet. If the conditions during our sampling could be maintained throughout the year, this would result in the removal of 320 kg DIN yr^{-1} per hectare of drain or 120 kg of NO_x^--N ha^{-1} yr^{-1}. This estimation from the in-out flow is lower compare to the measured NO_x^--N loss estimated from denitrification (790 kg of NO_x^--N ha^{-1} yr^{-1}). The DIN removed by D5 orginitaes from a mixture of groundwater/rainfall and runoff.

Contrarily, D1 and D2, despite having potential for denitrification, did not show any significant or consistent nutrient removal. These drains had the highest flow discharge at 0.36 m^3 s^{-1}, which were slightly reduced as water moved from D1 to a more densely vegetated D2. In this fast-flowing water, anaerobic conditions were likely to dominate. And without adequate residence time, the potential for denitrification achieved at the steady conditions of the experiment did not occur.

Site D3 did not show any significant nutrient removal; however, D4 had high NH_4^+-N removal, contributing to DIN losses. These two drains had very low flow discharge at around 0.02 m^3 s^{-1}, and at D4, water was almost stagnant. These conditions are likely to favour the transformation of organic N to NH_4^+-N, which was likely to be consumed by plants or algae, resulting in a temporary DIN removal. However, because denitrification is the only process that can permanently remove N from the water, the performance of the drains under this stagnant condition was not optimal.

Overall, the hydrological and physicochemical conditions of D5 can be used as an example of "optimal" conditions that could be achieved through the management of drains. Management activities could include reducing water flows by modifying the drain slope, allowing vegetation to grow within "nodes" of the drains simulating natural wetland conditions, increasing retention times, and avoiding stagnation. Our results also show that managing the "main drains" could provide a better approach than managing multiple lateral drains. Figure 11 provides a schematic diagram of the ideal conditions for N removal that could also be used to communicate the result of this study with landholders.

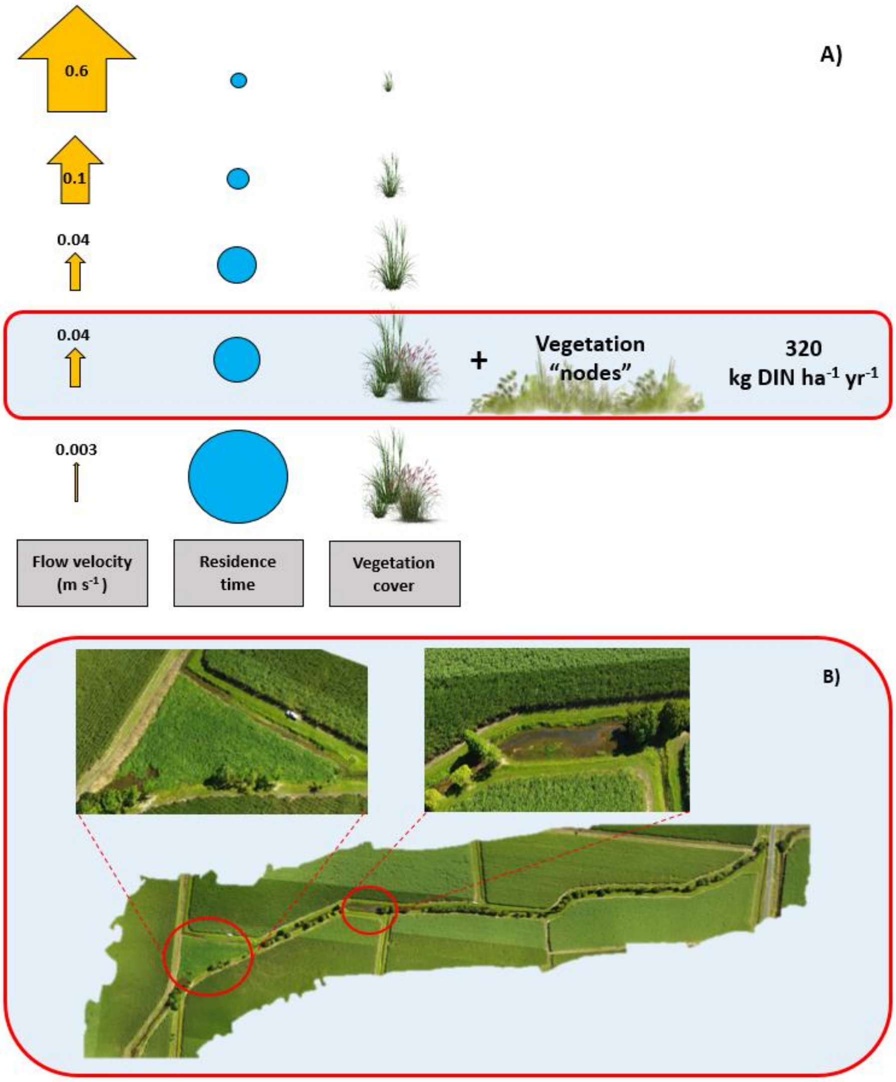

Figure 11. Schematic diagram of the optimal condition (flow velocity, residence time and vegetation cover) for DIN removal in drains. The size of the arrows and circles represent the magnitude of the value, (B) vegetation "nodes" within the main drain, which increase residence time while maintaining flow velocity.

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