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Key Points:

- Synoptic sampling of DIC, DOC and dissolved C gases along a tropical stream and connected wetlands
- Seasonal wetlands contributed ~15% and 16% of the stream DOC and DIC loads
- The riparian forest and groundwater inflows were likely the main sources of DOC and DIC to the stream, respectively

Supporting Information:

Supporting Information may be found in the online version of this article.

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Seasonal Wetlands Make a Relatively Limited Contribution to the Dissolved Carbon Pool of a Lowland Headwater Tropical Stream

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Abstract

Wetlands process large amounts of carbon (C) that can be exported laterally to streams and rivers. However, our understanding of wetland inputs to streams remains unclear, particularly in tropical systems. Here we estimated the contribution of seasonal wetlands to the C pool of a lowland headwater stream in the Australian tropics. We measured dissolved organic and inorganic C (DOC and DIC) and dissolved gases (carbon dioxide—CO_2, methane—CH_4) during the wet season along the mainstem and in wetland drains connected to the stream. We also recorded hourly measurements of dissolved CO_2 along a 'stream–wetland drain–stream' continuum, and used a hydrological model combined with a simple mass balance approach to assess the water, DIC and DOC sources to the stream. Seasonal wetlands contributed ~15% and ~16% of the DOC and DIC loads during our synoptic sampling, slightly higher than the percent area (~9%) they occupy in the catchment. The riparian forest (75% of the DOC load) and groundwater inflows (58% of the DIC load) were identified as the main sources of stream DOC and DIC. Seasonal wetlands also contributed marginally to stream CO_2 and CH_4. Importantly, the rates of stream CO_2 emission (1.86 g C s^{-1}) and DOC mineralization (0.33 g C s^{-1}) were much lower than the downstream export of DIC (6.39 g C s^{-1}) and DOC (2.66 g g C s^{-1}). This work highlights the need for further research on the role of riparian corridors as producers and conduits of terrestrial C to tropical streams.

Plain Language Summary

Streams and rivers play a vital role in carrying carbon to oceans. This carbon can originate from biological processes in the water or from external sources like rocks, forest and wetland soils. The proportion of carbon from each source depends on factors such as the local geology, climate, and landscape. In this study, we measured how much of the carbon transported by an Australian tropical stream was sourced from the wetlands in the catchment. We found that seasonal wetlands contributed ~15% of the carbon measured in the stream. We conclude that the main sources of carbon to the stream were the riparian forest, and rock-derived carbon carried by groundwater.

1. Introduction

Wetlands play an important role in the global carbon (C) budget, despite only covering between 2% and 6% of the Earth's surface area (Kayranli et al., 2010). The accumulation of organic matter within these freshwater ecosystems (Leibowitz et al., 2018) represents an important pool of C that can either be stored, remineralized and emitted as a greenhouse gas, or exported via streams and rivers depending on the flooding conditions and degree of connectivity with drainage networks (e.g., Eckhardt & Moore, 1990; Mulholland, 1981; Richey et al., 2002). Rates of C gas emissions to the atmosphere and C storage within wetlands have been measured to quantify the source and/or sink strength of wetlands at both local (e.g., Dinsmore et al., 2010; Liu et al., 2022; Saunders et al., 2007) and global scales (e.g., Mitsch et al., 2013; Were et al., 2019). But despite recent calls to better integrate C transfer from wetlands to other aquatic systems (Abril & Borges, 2019), much of the recent literature has ignored the potential influence of wetland inputs on riverine C cycling. This is problematic because failure to account for the wetland–river C flux can lead to inaccuracies in C inventories, particularly the contribution of terrestrial C to inland water systems.

Wetlands can be major sources of organic C to streams and rivers (e.g., Casson et al., 2019; Laudon et al., 2011; Moreira-Turcq et al., 2013; Zhang et al., 2022). The input of dissolved organic carbon (DOC) from wetlands to streams can lead to (a) an increase in the organic C concentrations in streams where it can then be exported downstream, respired or buried, and (b) an indirect increase in the partial pressure of CO_2 (pCO_2) in the stream resulting from metabolic activity where the dissolved organic matter is mineralized into inorganic C (Abril & Borges, 2019; Richey et al., 1990). However, precipitation events may also dilute DOC in wetland waters before they reach the stream, leading to a decrease of the stream DOC concentration. This dilution pattern has been evidenced in boreal and temperate streams (e.g., Eimers et al., 2008; Laudon et al., 2011).

Wetlands can also transfer dissolved inorganic C (DIC) to streams in significant amounts (e.g., Abril et al., 2014; Dinsmore et al., 2010; Schneider et al., 2020). Wetland DIC can be derived from the weathering of carbonate minerals in the soil, root respiration and/or organic matter decomposition. The patterns of export of DIC are complex due to parameters such as pH and light availability that control its speciation (i.e., the amount of DIC found as CO_2, HCO_3^- or CO_3^{-2}) and affect its concentration through photosynthesis and respiration. Overall, CO_2-enriched wetland waters can increase the DIC concentration and downstream export in streams (e.g., Borges et al., 2019; Geeraert et al., 2017), but in systems where DIC primarily originates from mineral weathering, the connectivity between wetlands and streams can cause a dilution of this geogenic DIC in the stream (e.g., Duvert, Hutley, Beringer, et al., 2020). A temporary DIC dilution can also occur in streams where respired-CO_2 is dominant. High flows can exhaust the CO_2-respired sources, resulting in a concomitant decrease in pCO_2 in wetlands and streams (e.g., Hope et al., 2004)—yet despite this DIC dilution effect, streams often remain oversaturated in CO_2 at high flow conditions.

While the influence of wetlands on stream C cycling is likely to be important across different regions and climates, the bulk of literature on wetland–stream C linkages has been conducted in northern latitudes (e.g., Aho & Raymond, 2019; Eimers et al., 2008; Hope et al., 1997). In the tropics, recent research has mostly focused on large wetland–river systems, leading to an increased recognition that tropical wetlands may play a significant role sourcing C to large rivers (Abril & Borges, 2019; Abril et al., 2014; Borges et al., 2019). Because of their generally high rates of primary productivity (Davies et al., 2008), tropical wetlands are likely to hold high amounts of C that can be rapidly transferred to adjacent rivers due to high landscape connectivity in wet periods (Boulton et al., 2008). Lateral export of C from wetlands to rivers have been studied mostly in the Amazon River, where wetlands can export up to 50% of their gross primary productivity (GPP), resulting in an important downstream increase in the dissolved C pool, which in turn can fuel CO_2 emissions from the river (Abril et al., 2014; Mayorga et al., 2005; Moreira-Turcq et al., 2013; Richey et al., 2002). Studies have also been undertaken in large tropical African wetland–river systems, where seasonal and perennial flooded land also exports large amounts of DOC and DIC into the Congo, Tana and Zambezi Rivers (Borges et al., 2015, 2019; Geeraert et al., 2017; Teodoru et al., 2015). But to date, few studies have explored the exchange of C between small wetlands and low-order streams in headwater catchments of the tropics (but see Birkel et al., 2020; Moustapha et al., 2022; Schneider et al., 2020).

Northern Australia is characterized by a highly seasonal tropical climate, with most of the rainfall (~90%) occurring between December and April (Petheram et al., 2008). The concentrated rain events (i.e., few rain days and high mean daily intensities (Jackson, 1988)) and mostly flat topography result in the flooding of up to ~25% of the Australian wet-dry tropics (Warfe et al., 2011). Together, these environmental conditions suggest that seasonal wetlands may contribute substantially to the C pool of streams and rivers during the wet season. Studies conducted in a peri-urban catchment in the Australian wet-dry tropics found that seasonal wetlands play a major role in fuelling catchment C export during the wetter periods (e.g., Birkel et al., 2020; Duvert, Hutley, Birkel, et al., 2020). The annual DOC load in the stream was estimated at 5.9 g C m^{-2} yr^{-1} (Duvert, Hutley, Beringer, et al., 2020), from which almost 90% was sourced by seasonal wetlands and the riparian forest (Birkel et al., 2020). While DIC export from wetlands to streams in the Australian wet-dry tropics has not been quantified yet, a DIC source partitioning model showed that seasonal wetlands may be particularly important contributors to the stream DIC pool during the wet season and recession period (Duvert, Hutley, Birkel, et al., 2020).

Duvert, Hutley, Beringer, et al. (2020) described the uncertainties in quantifying fluvial fluxes in these highly seasonal systems and the likely importance of riverine C sourced from seasonal wetlands. As such, we build on their work in this study by quantifying the contribution of wetlands to the dissolved C pool in Manton Creek, a headwater lowland stream in northern Australia with relatively little anthropogenic intervention. We hypothesize that:

(H1) seasonal wetlands are major contributors of DOC to the stream as organic matter accumulates in the wetlands and is then transported to the stream during the wet season; and
(H2) due to the likely importance of carbonate-rock derived groundwater DIC inputs to the stream, wetlands are not the dominant contributors of DIC to the stream C pool.

To test these hypotheses, we conducted a wet-season synoptic sampling of DOC, DIC, pCO_2 and pCH_4 longitudinally along the main stream and in several wetland drains directly connected to the stream, and monitored pCO_2 in the water along a "stream–wetland drain–stream" continuum. Additionally, we used a tracer-aided, conceptual rainfall-runoff model (SAVTAM) developed by Birkel et al. (2020) to perform a hydrograph separation and simulate year-round DOC fluxes. Lastly, we used the results of our synoptic campaign and modeling to assess the contributions of different hydrological units (seasonal wetlands, riparian forest, deeper groundwater) to the catchment DOC and DIC loads.

2. Methods

2.1. Study Site

This study was undertaken in Manton Creek, a low relief headwater catchment located in the Australian wet-dry tropics, 75 km south of Darwin (12.8804S, 131.1301E; Figures 1a and 1b). The climate in the region is highly seasonal with two main seasons; a hot wet season characterized by intense monsoonal events from December to April and a cooler dry season from May to November. The mean annual temperature in the study region is 27.5°C (Bureau of Meteorology, 1992-2003). The Manton Creek catchment has a mean altitude of 70 masl and is underlain by an extensive dolostone aquifer. The soils are sandy loam, characterized by 10%–20% of clay within the first 30 cm (R. A. Viscarra Rossel, Webster, et al., 2014).

The catchment drains a 34 km^2 area. It is covered by savanna woodlands (31 km^2; 91% of the catchment area), seasonal wetlands (3 km^2 of flooded area at the peak of the wet season; 9% of the catchment area) and a riparian forest along the stream and around the main spring area (<0.1 km^2; <1% of the catchment area; Figure 1b). The dominant vegetation consists of perennial grasses and Eucalyptus savanna woodland, with riparian forest dominated by *Melaleuca* spp., *Lophostemon* spp., and other vine forest species along the stream channel.

The seasonal wetlands consist of small depressions in the landscape surrounded by upland tropical savanna. These non-floodplain wetlands are connected to the stream through unidirectional hydrological flows (as per Leibowitz et al., 2018) via natural drainage channels (from now on termed "wetland drains", Figure 1c). Because they are located higher up in the landscape than the main stream, these wetlands are likely to receive water from (a) direct runoff from upland areas and/or (b) saturation excess overland flow. The wetland soils typically become saturated at the start of the wet season (December–January), due to runoff from the highlands and a rise of the local water table. The flooding lasts for 1–4 months depending on the magnitude of wet-season rainfall; the wetlands then dry out between April and June. The total flooded area fluctuates seasonally depending on the magnitude of the wet season, covering up to 3 km^2 (i.e., 9% of the catchment) at the peak of the wet season.

Manton Creek is a third-order stream during the wet season and transitions to a first-order stream during the dry season. In its upstream part, water originating from extensive seasonal wetlands flows into the mainstream through two wetland drains (sites 6 and 7, Figure S1 in Supporting Information S1). The contribution from these upstream seasonal wetlands accounted for less than 20% of the streamflow measured at the outlet of the catchment during our synoptic campaign.

2.2. Field Measurements and Sampling

Field measurements and sampling were conducted during the wet season 2021 (from January to May) to assess the influence of the wetland inputs on the stream C pool. The rainfall during this period was 1,255 mm, while the entire wet season rainfall for the 2020–2021 hydrological year amounted to 1,797 mm (Northern Territory Government, 2023).

A synoptic sampling campaign was undertaken during flow recession, ~10 days after a monsoonal event in March 2021 (174.5 mm of rain in 6 days). Samples for DOC and DIC concentrations were collected longitudinally at 10

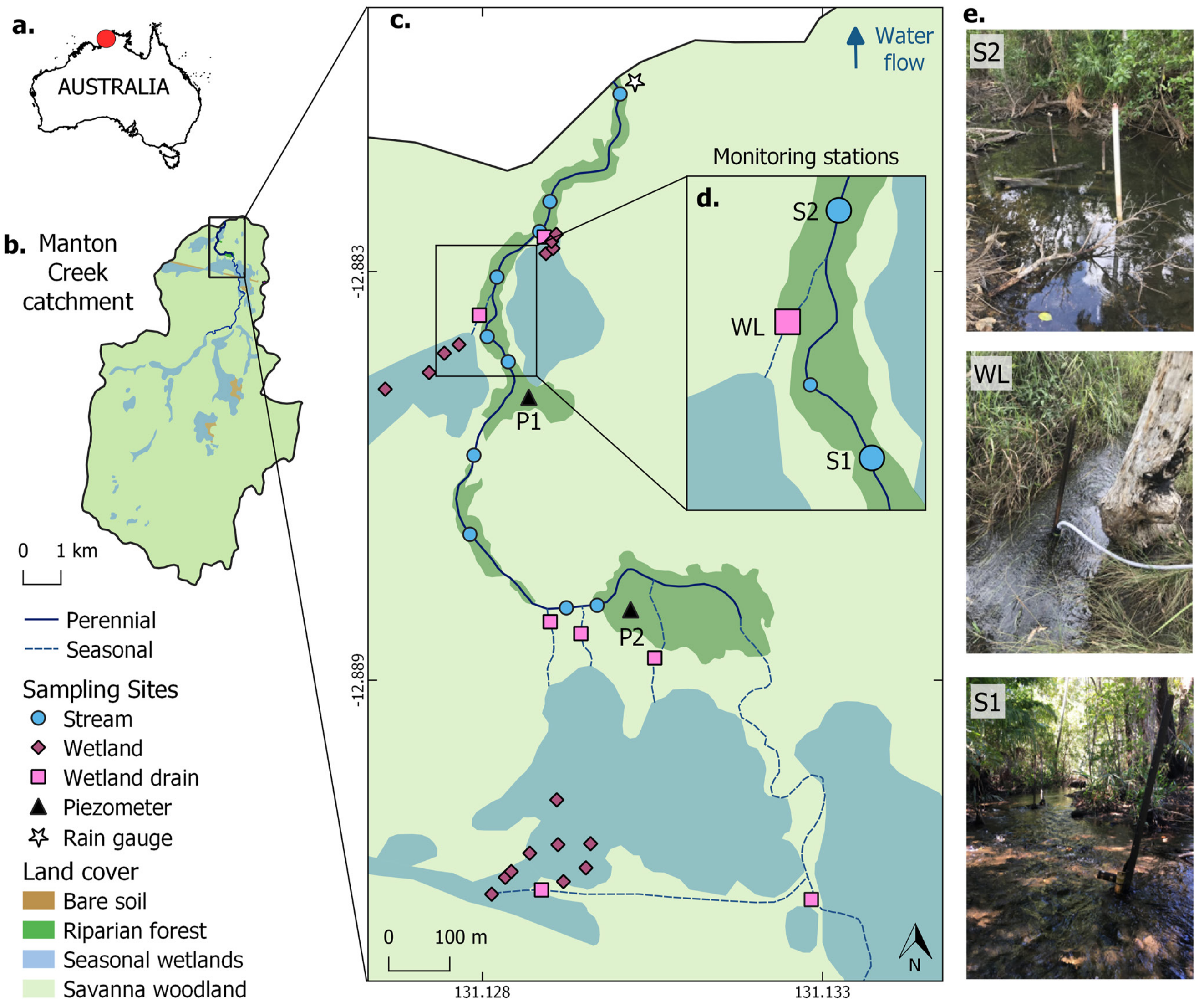

Figure 1. Location of the main study site in (a) Australia and (b) the Manton Creek catchment. The detailed map in (c) shows the location of the sampling points along the stream and in the seasonal wetland drains. The three automated monitoring stations (i.e., our "stream–wetland drain–stream" continuum) are shown in inset (d) with site pictures in (e). The land cover areas are based on Sentinel-2 satellite imagery (European Space Agency, 2017).

sites along the stream, from an upstream location near a permanent dolostone groundwater inflow (main spring area), to the catchment outlet ~1.5 km downstream (Figure 1c). We also sampled seven seasonal wetland drains that flow into the stream. These wetland drains were sampled as close as possible to their confluence with the main stream.

To characterize the riparian C source, we collected soil water from a shallow PVC piezometer (0.5 m long slotted screen section; ~1 m depth) located within a riparian vine forest thicket (P1; Figure 1c). The piezometer intersects a shallow subsurface flow path that drains the forested area. This flow path was directly connected to the stream and remained active late into the dry season, as suggested by the high soil moisture observed throughout the year (water table <0.8 m below ground at the end of the dry season). To obtain a more robust characterization of this riparian endmember, we considered seven DOC and two DIC measurements obtained from P1 at different dates (see Table S2 in Supporting Information S1).

To characterize the dolostone groundwater C source, we collected water from one piezometer and one NTG observation bore. The piezometer (0.5 m screen) intersects the groundwater inflow at ~1 m depth (P2; Figure 1c), while the bore (RN025962) is located ~1 km north of the catchment (not shown in Figure 1) and intersects the same dolostone aquifer at a depth >30 m. Before collecting water samples, we purged the piezometers and bore three times using submersible pumps (GP1352, Whale, UK; and Monsoon, Proactive Environmental Products, US).

At each sampling site, water quality parameters (water temperature, electrical conductivity, and pH) and discharge were measured using a Hanna Scientific handheld multiparameter sensor (HI98194) and an OTT MF-Pro Flowmeter, respectively. Additionally, high frequency (10-min) discharge data at the outlet of the catchment were retrieved for the study period from the Manton Creek gauging station (Northern Territory Government, 2023).

Emissions of CO_2 and CH_4 along the stream, in the wetland drains and in standing water within the wetlands were also measured in March 2021 to compare the significance of wetland emissions relative to their downstream export. We used a customized floating chamber connected to a calibrated Licor LI-7810 gas analyzer (LI-COR Biosciences, USA). Deployment sites along the stream were chosen based on crocodile risk; while in the wetlands, sites were chosen based on vegetation and water level to ensure representativeness (Figure 1c). The partial pressure of CO_2 (pCO_2) in the water column was measured using a CM-0056 m (CO_2 Meter, USA) connected to a RAD-AQUA degassing system (Durridge, USA). Pumped water was sprayed into the RAD-AQUA chamber, allowing the dissolved CO_2 to reach equilibrium with the headspace air in the chamber. The equilibrated air was drawn into the CO_2 analyzer through a closed gas loop. Both pCO_2 and the CO_2 emission flux were used to calculate the gas transfer velocity for CO_2 according to Fick's law of gas diffusion (Raymond et al., 2012), which was then used to estimate the gas transfer velocity for CH_4 based on Wanninkhof (1992), using the Schmidt coefficients provided by Raymond et al. (2012). We then paired the measured CH_4 emission flux and its gas transfer velocity to obtain the partial pressure of CH_4 (pCH_4) values at each sampling site.

Additionally, three automated stations were installed along a stream–wetland drain–stream continuum (S1, WL, and S2, Figures 1d and 1e) for 54 days (from late March to late May 2021) to measure pCO_2 and assess the contribution of the wetland drain to the stream as well as its variations over time. Measurements in the wetland drain stopped after 13 days when the water level was <5 cm. Each monitoring station consisted of a submerged pCO_2 sensor (eosGP, Eosense, Canada) connected to a CR800 logger (Campbell Scientific, USA). Measurements were recorded every 10 min and averaged to obtain hourly pCO_2 values.

In addition to the March 2021 intensive sampling campaign, water sampling was conducted from November 2017 to December 2021 to capture concentrations of DOC, DIC and the hydrogen isotopic ratio in water (δ^2H). This sampling effort is part of a larger program that included daily rain samples collected in Darwin (Munksgaard et al., 2019) and stream water samples collected at the outlet of the Manton Creek catchment (Site A, Figure S1 in Supporting Information S1) during regular field visits. Data from this long time series are used for partitioning DOC fluxes as described below. We note that all DOC samples collected during the 2019–2020 water year were lost.

2.3. Water Analyses

All collected water samples were analyzed for water stable isotopes (including δ^2H) at Charles Darwin University (Environmental Chemistry and Microbiology Unit—ECMU) using a Picarro L2130-i cavity ringdown spectrometer. Samples for DOC analyses were filtered (0.45 μm) in the field and collected in pre-acidified (98% H_2SO_4) 40 mL borosilicate amber vials. DOC was measured at Southern Cross University using a total organic C analyzer (Shimadzu TOC-VCPH) with a precision of <2%. Samples for DIC concentrations were also filtered in the field (0.45 μm) before collection in 12 mL gas-tight glass vials (Exetainers, Labco). These samples were later analyzed at the Australian Nuclear Science and Technology Organization (ANSTO) laboratory using an equilibration method on a Gas Bench II coupled to continuous-flow Delta V Advantage isotope ratio mass spectrometer based on Assayag et al. (2006). The DIC concentrations have a precision of 20%–40% resulting from the analytical method used which was specifically developed for higher DIC concentrations in groundwater.

2.4. Statistical Analyses

All the analyses were performed using R (R Core Team, 2023). The Shapiro Wilk test was used to assess whether data were normally distributed. Given the assumption of normality was not met, the Kruskal-Wallis test was used to determine whether the water quality parameters, dissolved C concentrations and emission rates from the wetlands, wetland drains and the stream were statistically different.

2.5. Tracer-Aided Hydrological Model

To enhance our understanding of the temporal variations in wetland contribution to the stream C load, long term partitioning of DOC fluxes was modeled using SAVTAM (SAVanna Tracer-Aided Model). This semi-distributed model was developed by Birkel et al. (2020) to analyze the DOC dynamics in the nearby (~50 km distance) but larger ~120 km^2 Howard River catchment, a similar system to Manton Creek in terms of topography, geology, land cover and climate. The model is based on the conceptual understanding of the catchment functioning of Duvert, Hutley, Birkel, et al. (2020) and Cook et al. (1998). It considers the extreme climatic seasonality of the wet-dry tropics, the dominant landscape units (i.e., the spatial extent of the seasonal wetlands and savanna woodlands and its variations through the seasons) and connected hydrological processes occurring in the catchment such as the runoff generation, infiltration and recharge processes.

The model performs a hydrograph separation based on daily hydrometeorological and isotopic data and simulates DOC fluxes from two different water sources, that is, groundwater and seasonal wetlands/riparian zone. The deuterium isotopic ratio (δ^2H) in the stream represents the isotopic integration of different water sources. Since this tracer has a conservative behavior upon mixing, it can be used to estimate the contribution of different water sources based on their specific δ^2H signature.

Because the structure of the model combines the riparian forest and seasonal wetlands into the same module (Birkel et al., 2020), in our work the DOC contributions from both the seasonal wetlands and the riparian forest were summed up and given as one sole output—a limitation that we detail further in the Discussion.

We set the model to run from November 2017 to December 2021 given available data as described above. The hydrometeorological data were obtained from different sources: (a) daily relative humidity and temperature data were obtained from a weather station located 22 km southwest of the monitoring station (Bureau of Meteorology, 2023); (b) rainfall data were obtained from two rain gauges depending on data availability. From November 2017 to December 2019, we used data from the Batchelor weather station (Bureau of Meteorology, 2023) and from January 2020 to December 2021 we used data from the Manton Creek rain gauge (Northern Territory Government, 2023). While the two rain gauges are relatively close to each other, we acknowledge that very localized rainfall events could lead to uncertainties in the modeling; (c) potential evapotranspiration (PET) was calculated based on temperature according to Hargreaves and Samani (1985); and (d) discharge data were retrieved from the Manton Creek gauging station (Northern Territory Government, 2023) located at the outlet of the catchment.

The model inputs consisted of rainfall, PET and rainfall δ^2H, while discharge, streamflow δ^2H and observed DOC data (all collected at the outlet of the Manton Creek catchment, that is, Site A in Figure S1 in Supporting Information S1) were used to calibrate the model using a multi-objective optimization procedure. We used the Non-Sorted dominated Genetic Algorithm (NSGA2) from Deb et al. (2002) and the associated R package with the default setup. Four calibration objectives were used; the modified Kling-Gupta Efficiency (KGE) criterion (Kling et al., 2012) for discharge, streamflow δ^2H and stream DOC concentrations and additionally, a logarithmic version of the KGE to calibrate lower stream flows. The model uses a total of 19 calibrated parameters, eight parameters representing the rainfall-runoff simulations, four parameters for stable isotope mixing, and seven parameters for the DOC mass balance. The warm-up period involved running 500 parameter populations over 1,000 generations, resulting in a total of 500,000 iterations of model parameter sets. The best 500 parameter combinations were retained for simulation, and used as indicators of uncertainty, hydrograph separation, water and tracer budgets with the respective 5th/95th percentiles. For a full description of the model, we refer to Birkel et al. (2020).

2.6. Water and C Flux Partitioning

To obtain approximate estimates of the contribution of different landscape inputs to the stream C pool, we estimated the DIC and DOC loads based on the data obtained across the catchment during our synoptic campaign, in March 2021. We combined the flow partitions obtained from SAVTAM (Q_{outlet} and $Q_{\text{groundwater}}$) and our flow measurements in the wetland drains ($\Sigma Q_{\text{wetland}}$) to estimate the flow contribution from the riparian corridor (Q_{riparian}):

$$Q_{\text{riparian}} = Q_{\text{outlet}} - Q_{\text{groundwater}} - \sum Q_{\text{wetland}} \tag{1}$$

where $\Sigma Q_{\text{wetland}}$ was scaled to account for the difference between the measured and modeled streamflow at the outlet of the catchment. In this formulation, the riparian contribution can be thought of as the sum of interflow and subsurface stormflow pathways discharging into the stream. We multiplied each flow estimate to its corresponding DIC and DOC concentration (assuming stable concentrations for the duration of the synoptic campaign) to calculate average "input" loads for each landscape unit. We then compared these DIC and DOC inputs to the catchment DIC and DOC "outputs" estimated at the outlet (site A, Figure S1 in Supporting Information S1). For DOC, we used the mean of four values measured at the outlet during the wet season 2021 (Table S2 in Supporting Information S1) as a more robust estimate of catchment DOC export. For DIC, we used the only available measurement taken at the outlet during the synoptic sampling campaign.

The difference between DOC input and output loads can be seen as a rough estimate of the rate of DOC mineralization occurring along the stream. We compared this estimate to the mean wet-season in-stream respiration rate reported by Solano et al. (2023) for Manton Creek.

Additionally, we estimated an average CO_2 emission rate along the stream using our flux chamber measurements and compared this estimate to that based on the stream water residence time and average reaeration coefficient. The reaeration coefficient was estimated using the sixth empirical model of Raymond et al. (2012) and an average stream depth (see Solano et al. (2023)). Our ultimate goal here was to compare the estimated CO_2 emission rate to the downstream DIC load.

3. Results

3.1. Dissolved Carbon Concentrations

During the synoptic campaign in March 2021, the stream had a mean daily discharge of 0.28 $m^3\ s^{-1}$ at its outlet (range 0.22–0.60 $m^3\ s^{-1}$), which is much higher than the annual median daily discharge (0.02 $m^3\ s^{-1}$, range 0.001–19.49 $m^3\ s^{-1}$) for 2021. Meanwhile, the wetland drains had discharges ranging from 0.001 $m^3\ s^{-1}$ to 0.04 $m^3\ s^{-1}$. All water quality parameters exhibited significant differences between the stream and the wetland drains ($p < 0.001$, Table S1). Overall, electrical conductivity (EC) was higher in the stream (range 183–222 $\mu S\ cm^{-1}$, mean 202.4 $\mu S\ cm^{-1}$, Figure 2a) than in the wetland drains (11–122 $\mu S\ cm^{-1}$, mean 48.7 $\mu S\ cm^{-1}$, Figure 2a). The pH was slightly higher in the stream (6.6–7.2, mean 7) than in the wetland drains (5.8–6.9, mean 6.4), while temperature was higher in the wetland drains (mean 33.3°C) than in the stream (mean 30.3°C).

Dissolved CO_2 concentrations were significantly higher in the stream, with a mean of 3.5 g C m^{-3} (conversion factor to mg C L^{-1} is 1) compared to a mean of 2.0 g C m^{-3} in the wetland drains ($p < 0.01$, Figure 2b). Dissolved CH_4 concentrations, on the other hand, were slightly higher in the stream (mean 0.022 g C m^{-3}) than in the wetland drains (mean 0.019 g C m^{-3}) but there was no significant difference between both groups (Figure 2c).

Both DOC and DIC spanned a wide range across the wetland drains (Figures 2d and 2e), with DOC concentrations varying from 1.8 g C m^{-3} to 17.8 g C m^{-3} (mean 5.5 g C m^{-3}) and DIC ranging from 3.6 g C m^{-3} to 42.0 g C m^{-3} (mean 14.6 g C m^{-3}). The highest DIC concentrations occurred at the two upper most drains (sites 6 and 7 in Figure S1 in Supporting Information S1), where DOC and EC were lowest; these two sites drain the large seasonal wetlands that cover the upstream area of the catchment.

The DOC concentrations obtained along the stream during the synoptic campaign were lower and less variable than in the wetland drains, ranging from 1.5 g C m^{-3} to 7.5 g C m^{-3}, while the wet-season stream DOC concentrations obtained at the outlet were generally higher (range 2.5–17.7 g C m^{-3}, mean 7.5 g C m^{-3}, $p < 0.05$, Figure 2d). DIC was higher in the stream when compared to wetland drains, with a mean of 19.9 g C m^{-3} and values ranging from 14.4 g C m^{-3} to 30.0 g C m^{-3} (Figure 2e). The highest mean DOC concentration was found in the riparian source (P1; 11.9 g C m^{-3}), while the highest mean DIC concentration was found in the dolostone groundwater source (P2; 54.6 g C m^{-3}).

3.2. Emission Fluxes

Emission fluxes were consistent among replicate chamber measurements at each sampling site. The mean emission rates and their corresponding standard deviations are reported in Table S1. Overall, CO_2 emission rates were up to two times higher in the stream (mean 16.0 g C $m^{-2}\ d^{-1}$) than in the wetland drains (mean 10.2 g C $m^{-2}\ d^{-1}$), and almost five times higher than in the wetlands themselves (mean 3.4 g C $m^{-2}\ d^{-1}$, Figure 3). The wetland drains had the highest CH_4 emission rates (mean 0.10 g C $m^{-2}\ d^{-1}$) followed by the stream (mean 0.08 g C $m^{-2}\ d^{-1}$) and

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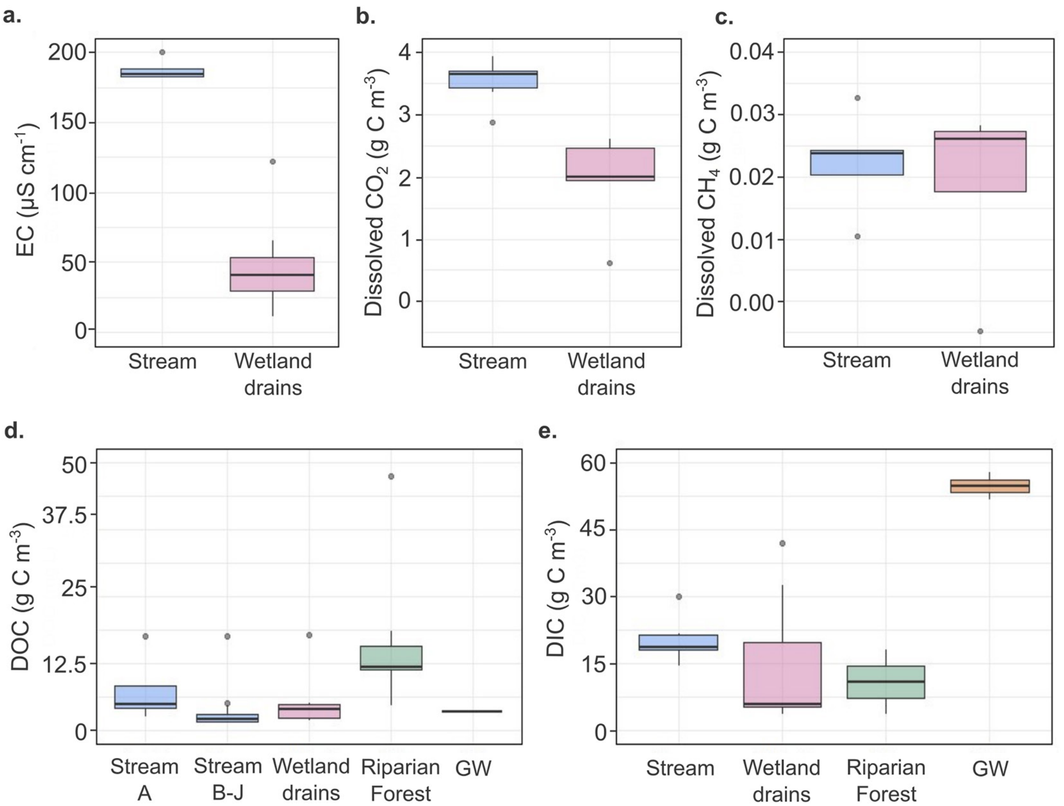

Figure 2. Electrical conductivity (EC) and dissolved C concentrations measured in the stream, wetland drains and piezometers across the catchment. For stream DOC, "stream A" corresponds to measurements at the catchment outlet during the wet season 2021 while "stream B-J" corresponds to measurements along the stream during the synoptic campaign. "GW" stands for groundwater. Details for each sampling point can be found in the Supplementary Information (Figure S1 in Supporting Information S1, Table S1, and Table S2 in Supporting Information S1).

wetlands (mean 0.04 g C m^{-2} d^{-1}). The differences in CO_2 and CH_4 emission rates between landscape units were only significant between the stream and the wetlands ($p < 0.05$ and $p < 0.01$, respectively). Lower fluxes in the wetlands when compared to the stream and wetland drains were likely a result of the much lower turbulence in the wetlands.

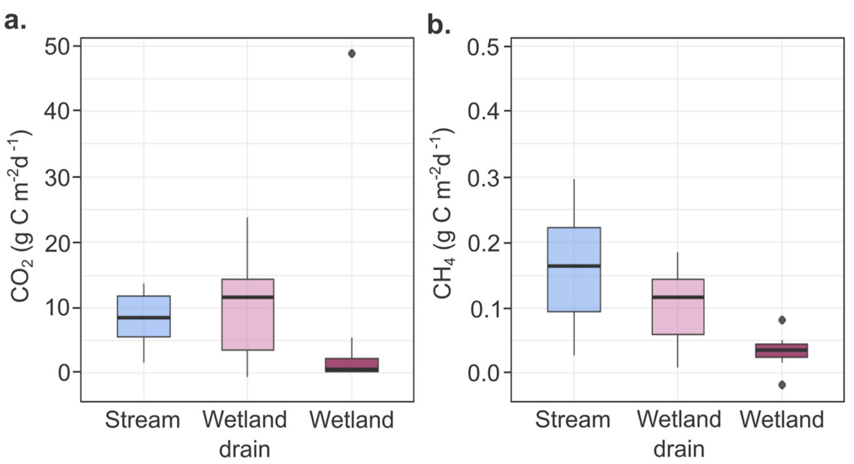

Figure 3. CO_2 and CH_4 emission fluxes measured at different sites along the stream, the wetland drains and the wetlands.

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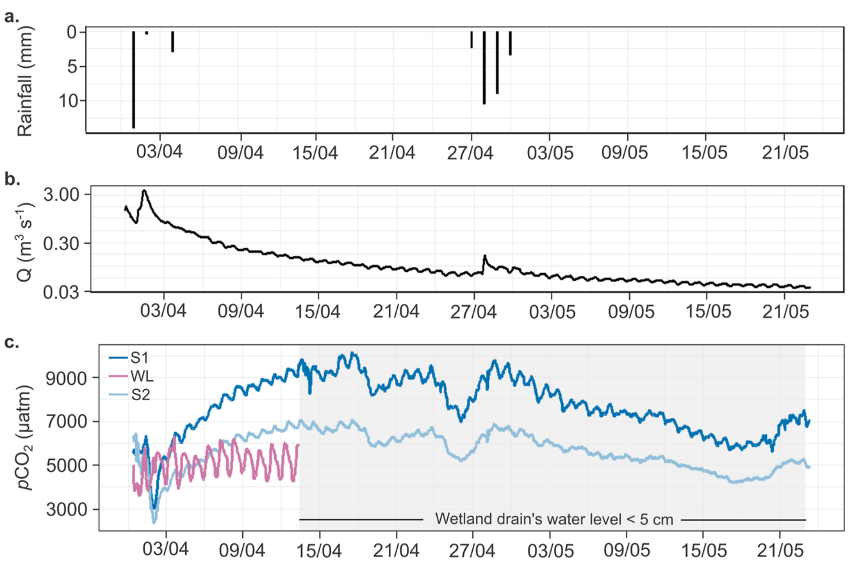

Figure 4. Daily rainfall (a) and hourly mean discharge at the outlet of the catchment (b), and CO_2 partial pressure (c) along a stream–wetland drain–stream continuum at Manton Creek. The CO_2 partial pressure was measured at two locations along the stream; upstream (S1) and downstream (S2) of a wetland drain and in the wetland drain itself (WL). Locations and pictures of the three sites are shown in Figures 1d and 1e. The $p\mathrm{CO}_2$ values are included in (Table S3).

3.3. Temporal Changes in $p\mathrm{CO}_2$ Along a Stream–Wetland Drain–Stream Continuum

During the 54 days of hourly $p\mathrm{CO}_2$ measurements, two important rainfall events occurred in the catchment, with 17.5 mm falling in 3 days in early April and 25.5 mm falling in 4 days in late April–early May (Figure 4a). These events resulted in an increase in streamflow (Figure 4b) but had a negligible effect on the wetland drain discharge. The stream $p\mathrm{CO}_2$ was consistently higher upstream of the wetland inflow (S1, mean 7,747 µatm) than immediately downstream of it (S2, mean 5,652 µatm, Figure 4c), with similar temporal patterns of variation at the two sites. The lowest daily $p\mathrm{CO}_2$ values occurred after the first rainfall event (S1; 4,222 µatm, S2; 3,498 µatm), while the highest daily values occurred during the second rainfall event (S1; 9746 µatm, S2; 6831 µatm, Figure 4c).

The wetland drain had lower and more stable $p\mathrm{CO}_2$ than the two stream sites, with a daily mean of 5,071 µatm (range 3,612–6,287 µatm, Figure 4c) except for 4 days after the first rainfall event (late March) when the discharge was high. Furthermore, the difference in $p\mathrm{CO}_2$ between the upstream (S1) and downstream (S2) stream sites was similar between the periods with and without wetland inflow.

3.4. Modeled Hydrograph Separation and DOC Sources

The SAVTAM model performed reasonably well, with best-fit KGE values of 0.70 for simulated streamflow, 0.74 for stream δ^2H, and 0.65 for simulated DOC concentrations (Figures S2 and S3, Table S5 in Supporting Information S1). Simulations were able to reproduce the measured DOC loads and their variations over time, with wet-season peaks and dry-season lows (Figure 5). However, for the 2018–2019 period the model tended to overestimate the DOC loads (Figure S4 in Supporting Information S1), particularly in the dry season, which might be related to the higher uncertainties with discharge measurements at low flow.

Based on the model outputs, the DOC load exported annually from Manton Creek catchment was 4.62 g C m^{-2} yr^{-1}, 96.5% of which was exported during the wet season and 3.5% during the dry season (Figure 5). We found that the bulk of DOC originated from the undifferentiated area that encompasses both the riparian forest and seasonal wetlands (~92%), with less than 8% sourced from groundwater inflows.

3.5. Water and C Flux Partitioning

According to SAVTAM's hydrograph separation and Equation 1, during our synoptic sampling ~24% of the catchment streamflow originated from the seasonal wetlands, ~53% from the riparian compartment, and 23%

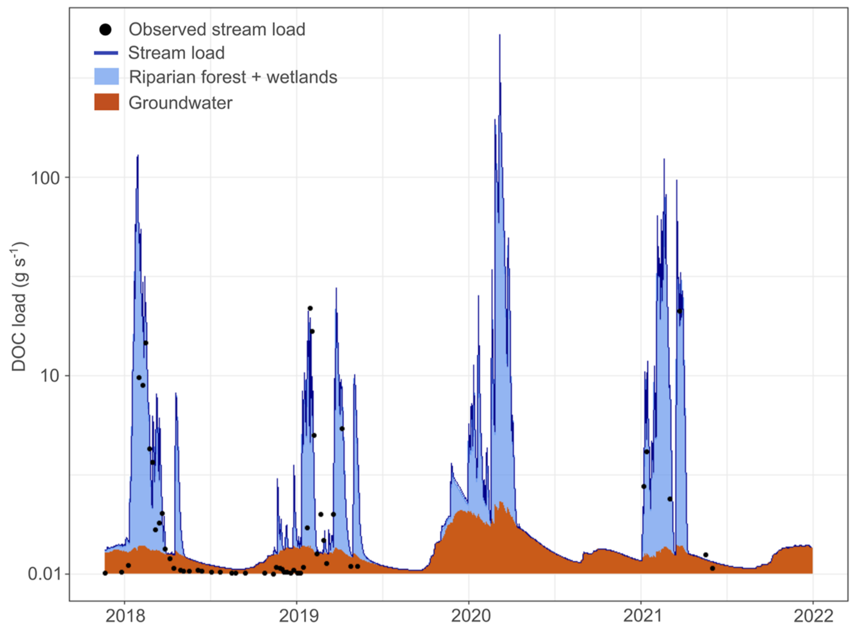

Figure 5. Simulated daily DOC loads (log-scale) from different sources at Manton Creek compared to observed DOC loads in the main stream. Note that all DOC samples taken in the 2019–2020 water year were lost.

from the deeper groundwater (Figure 6). It is worth noting, though, that the riparian contribution comprised subsurface inputs that may have transited through other landscape units (e.g., savanna soils) before reaching the riparian thicket.

Regarding catchment DOC loads, the contribution from the riparian source (~75%) largely exceeded that from the seasonal wetlands (~15%), while the dolostone groundwater contributed the remaining 10% (Figure 6, Table S6 in Supporting Information S1). Regarding catchment DIC loads, groundwater contributed the bulk of DIC to Manton Creek (~58%), while the riparian forest contributed ~26% and the seasonal wetlands contributed only ~16% (Figure 6, Table S6 in Supporting Information S1).

The DIC and DOC contributions from the seasonal wetlands are consistent with and even slightly higher than the area they cover (~9% of the catchment area). According to our estimates, the wetlands contributed 0.15 g C km^{-2} s^{-1} of DOC to the stream, which is similar to the average catchment load (0.08 g C km^{-2} s^{-1}). Likewise, the DIC area-weighted contribution of the seasonal wetlands (0.17 g C km^{-2} s^{-1}) was similar to the average catchment DIC load (0.23 g C km^{-2} s^{-1}). By contrast, the DOC and DIC contributions of the riparian forest were much higher (22.43 and 1.88 g C km^{-2} s^{-1}, respectively) than the average catchment loads.

The sum of the C inputs to the stream resulted in a DIC load of 7.75 g C s^{-1} and a DOC load of 2.99 g C s^{-1}. These loads were higher but comparable to those obtained using the modeled streamflow and the measured C

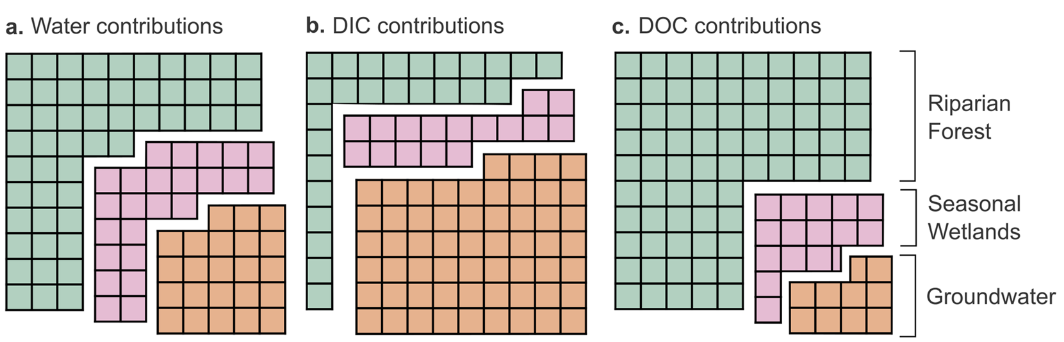

Figure 6. Relative contributions of (a) water, (b) DIC and (c) DOC inputs from each landscape unit to the stream. Further information about the data in this figure can be found in Supporting Information S1 (Table S6 in Supporting Information S1).

concentrations at the stream outlet ("C outputs"), which amounted to a DIC load of 6.39 g C s^{-1} and a DOC load of 2.66 g C s^{-1} (Table S6 in Supporting Information S1). The difference between the DOC inputs and outputs amounted to 0.33 g C s^{-1} and can be seen as an indirect estimation of the stream respiration rate (given that internal primary productivity is negligible; Solano et al., 2023). This value was relatively similar to that obtained using the respiration rates reported by Solano et al. (2023) for the same stream (0.19 g C s^{-1} during the wet season).

Based on flow velocity and river mapping, we estimated that the stream residence time was about 1.3 days from the spring to the catchment outlet (site J to site A, Figure S1 in Supporting Information S1). Considering a mean daily reaeration coefficient during our synoptic campaign of 0.018 $days^{-1}$, we estimated a CO_2 emission rate of 1.86 g C s^{-1}, which is similar but somewhat higher than the mean CO_2 emission rate resulting from the chamber measurements (1.36 g C s^{-1}, Table S1). Importantly, our estimate of CO_2 emission was over three times lower than the downstream DIC export at the stream outlet.

4. Discussion

Tropical headwater catchments are often thought of as hotpots of CO_2 emissions, yet our understanding of the processes governing C cycling in these systems remains limited. We quantified the contribution of seasonal wetlands to the dissolved C pool in a lowland headwater stream in the Australian wet-dry tropics, a region where monsoonal rainfall events can lead to flooding of a large portion of the landscape (Warfe et al., 2011). During our synoptic sampling campaign conducted under high flow conditions, the seasonal wetlands contributed ∼15% and ∼16% of the DOC and DIC loads in Manton Creek, respectively. While substantial and higher than the proportion of the catchment area covered by wetlands (∼9% during the wet season), these values suggest that the wetlands were not the primary source of either DOC or DIC to the stream. In the case of DOC, this outcome contradicts our initial expectations. Our results suggest that the more productive riparian forest was the main source of DOC, and that carbonate-rich groundwater inflows were the main source of DIC to the stream. Overall, this work highlights how the knowledge acquired in large river–floodplain systems does not necessarily translate to small headwater stream–wetland systems.

4.1. Wetland Contribution to Riverine C Across the Tropics

In the tropics, an increase in dissolved C concentrations in streams and rivers has been linked to their connectivity to floodplains and wetlands (e.g., Coynel et al., 2005; Melack & Engle, 2009; Teodoru et al., 2015), suggesting that wetlands can play an important role as C sources to streams and rivers. The contribution of wetlands to riverine C seems to be particularly high in large river–wetland systems of lowland tropical regions. Studies conducted in the Amazon, Tana, Zambezi and Congo Rivers have shown that river DIC and DOC are largely sourced from the primary productivity that occurs in their associated wetlands (Abril et al., 2014; Borges et al., 2019; Geeraert et al., 2017; Teodoru et al., 2015). These wetland C inputs are then likely emitted to the atmosphere further downstream, contributing significantly to riverine CO_2 emissions at a regional level (Abril et al., 2014; Borges et al., 2015; Richey et al., 2002). However, there are exceptions to this pattern of high wetland C inputs to tropical rivers. For instance, Lewis et al. (1990) found that the Orinoco floodplain behaved like a closed system, with no net export of nutrients and organic C even during high flow events.

Only more recently have the linkages between wetlands and lower order streams received some attention, with no emergent pattern across studies so far. Moustapha et al. (2022) found that wetlands sourced ∼27% of stream DOC in the Mengong catchment, which is a first order tributary of the Nyong River (0.6 km^2 catchment area, 20% wetland coverage). In contrast, Birkel et al. (2020) showed that seasonal wetlands and riparian forests contributed 90% of the annual stream DOC export to the Howard River (126 km^2 catchment area, 25%–30% wetland coverage). In both studies, the DOC export occurred mostly during high rainfall events, when hydrological connectivity and flow velocity were higher, in line with the pulse-shunt concept (Raymond et al., 2016). Similar to DOC, the transfer of DIC from wetlands to tropical streams remains understudied. Two studies have shown that wetlands can play an important role in contributing DIC and/or CO_2 to streams when hydrological connectivity is highest (Duvert, Hutley, Birkel, et al., 2020; Schneider et al., 2020). One of the few attempts to quantify this flux suggests that wetlands contributed <25% of the stream DIC load (Moustapha et al., 2022).

Adding to this body of knowledge, our results indicate that under high flow conditions, seasonal wetlands contributed ∼15% and ∼16% of the DOC and DIC loads in Manton Creek. These percent contributions are relatively similar to those reported by Moustapha et al. (2022) despite the lower wetland coverage in our study catchment (20% vs. 9%). When aggregating inputs from wetlands and the riparian forest, our percent DOC contribution reached 90%, very similar to the estimate from Birkel et al. (2020) despite a higher relative wetland area in their catchment.

4.2. Wetland DOC Contribution to Manton Creek

Based on our modeling results, Manton Creek catchment had an average annual DOC of 4.62 g C m^{-2} yr^{-1} over the 4-year study period. This value is slightly smaller but comparable to those estimated in other tropical streams of around 6 g C m^{-2} yr^{-1} (Duvert, Hutley, Beringer, et al., 2020; Moustapha et al., 2022). Approximately 15% of the stream DOC load was sourced from the seasonal wetlands during our synoptic campaign in the 2021 wet season, a substantial yet relatively limited contribution relative to the input from the riparian zone. This result allowed us to reject our first hypothesis (H1) that seasonal wetlands are the main contributors to stream DOC. This is likely a result of the low wetland primary productivity, short residence times and relatively small soil organic C pool in these wetlands, as discussed below.

Tropical floodplains and wetlands are considered one of the most productive systems on Earth (Davies et al., 2008), but the flooded systems of the seasonal tropics of Australia are typically in the lower range of primary productivity estimates (e.g., Adame et al., 2017; Beringer et al., 2013; Duvert, Hutley, Beringer, et al., 2020). The soils of the wet-dry tropics are in general ancient, highly leached and nutrient poor (Fellman et al., 2013). These factors, in addition to the annual water limitation and high fire activity of the dry season, limit vegetation productivity (Finlayson, 1991; Spessa et al., 2005), resulting in overall low C stocks in the seasonal wetlands of northern Australia. Moreover, the soil organic C stock in Manton Creek is even lower than that of other wetland systems in tropical Australia (R.A. Viscarra Rossel, Webster, et al., 2014). This low wetland organic C stock could further explain the low DOC concentrations measured in the wetland drains in Manton Creek compared to those measured in other wetlands in the Australian wet-dry tropics (e.g., Bass et al., 2014; Pettit et al., 2011).

Some of the DOC that transits through the seasonal wetlands may be converted to CO_2 through internal respiration before reaching the stream. In-stream respiration rates were found to be higher in Manton Creek during the wet season when DOC was readily available and water temperature was high (Solano et al., 2023). However, the higher DOC concentrations measured in the wetland drains compared to those measured in the stream (Figure 2d), coupled with short wetland residence times, support the notion that DOC mineralization may not have been an important process within the wetlands. The high $p\mathrm{CO_2}$ diel cycles measured in the wetland drain (Figure 4c), may therefore be primarily driven by photosynthesis.

4.3. Wetland DIC and Dissolved Gas Contribution to Manton Creek

Our results indicate that seasonal wetlands contributed around 16% of the DIC load in Manton Creek during our wet-season sampling campaign, a substantial contribution given their percent area cover (~9%). While non-negligible, this wetland contribution was eclipsed by the DIC inputs of carbonate-rich groundwater from the dolostone aquifer. This result and the negligible influence of wetland drain $p\mathrm{CO_2}$ on stream $p\mathrm{CO_2}$ (Figure 4) support our second hypothesis (H2) that seasonal wetlands are not the main contributors to the stream inorganic C pool.

In northern Australia, high soil temperatures and water content during the wet season can lead to high root respiration rates and organic matter decomposition in the soil (Chen et al., 2002, 2003). However, wetland soils may generate lower rates of CO_2 than savanna soils, due to the likely anoxic conditions at the peak of the wet season that may limit organic matter respiration. Electrical conductivity data (Figure 2a; Table S1) indicate that wetland waters in Manton Creek were mostly recent rainwater with relatively short residence times, which may have prevented the water to fully equilibrate with CO_2 in the wetland soils, and in turn limited their DIC contribution to the stream. This interpretation is corroborated by the lower CO_2 concentrations we measured in the wetland drains relative to those in the stream (Table S1).

Our hourly $p\mathrm{CO_2}$ measurements along the stream–wetland drain–stream continuum confirm that the wetland drain had a minor effect on the stream CO_2 pool, even during and immediately after rainfall events (Figure 4), which can be attributed to the comparably low concentrations and flow. The limited contribution of CO_2 from the wetlands to the stream is not completely unexpected when considering that Australian savannas are not organic-rich environments. Studies undertaken in mineral soil environments and similar systems to Manton Creek have shown that wetland waters can dilute the stream CO_2 pool (e.g., Duvert, Hutley, Beringer, et al., 2020; Hope et al., 2004). We also observed strong daily cycles in pCO_2 measurements in the stream and wetland drain, likely resulting from internal metabolism. While studying these sub-daily variations is beyond the scope of this work, their significance should not be overlooked as they can affect catchment C budgets (Attermeyer et al., 2021; Gómez-Gener et al., 2021).

We also found that the seasonal wetlands were unlikely to be a major source of dissolved CH_4 to the stream. Higher temperatures, higher availability of organic C and the anoxic conditions in the wetlands are expected to favor higher CH_4 concentrations in this landscape unit when compared to the stream (Stanley et al., 2016; Zhu et al., 2020). However, high CH_4 emission rates within the wetland drains (Figure 3) likely reduced the dissolved CH_4 flux before the wetland waters reached the stream.

The highest pCO_2 values at the outlet of Manton Creek occurred during the dry season (Solano et al., 2023), which further suggests that stream CO_2 cannot be sourced mainly from the seasonal wetlands, dry and disconnected during that period. During our sampling campaign, the DIC input from the dolostone groundwater (4.48 g C s^{-1}) was much higher than the sum of inputs from the wetland drains (1.24 g C s^{-1}), indicating that groundwater inputs were the main source of DIC to the stream. The contribution of internal stream metabolism to the stream DIC pool was comparatively lower (0.33 g C s^{-1}). The proportion of the groundwater DIC input that originates from geogenic versus biogenic sources remains unknown, though. Future studies in Manton Creek should investigate the isotopic signatures and age of different C sources to identify the major contributors of DIC to the stream and close the catchment C budget.

4.4. Riparian Corridors as an Overlooked C Source

Modeled DOC loads using SAVTAM showed a marked seasonal behavior (Figure 5), with higher DOC fluxes related to the wet season and mostly originating from the wetlands and riparian forest. Similar DOC dynamics were found in the Howard River (Birkel et al., 2020). However, an important limitation of our modeling framework is the inability to explicitly separate between wetland-derived and riparian forest-derived DOC contributions to Manton Creek.

The modeled DOC loads in Manton Creek, when combined with our synoptic campaign, suggest that most of the stream DOC was sourced from the riparian forest (~75%) rather than the seasonal wetlands (~15%). This is because subsurface water within the riparian zone (P1, Figures 1c and 2, Table S1) had DOC concentrations up to an order of magnitude higher than most wetland drains (Figure 2d) and contributed substantial amounts of water to the stream. While little attention has been given to the role of riparian forests as potential sources of DOC to headwater streams in the tropics, studies undertaken at other latitudes have long highlighted riparian forests as hot spots for DOC production and export (e.g., Fiebig et al., 1990; Ledesma et al., 2015; Lupon et al., 2023).

In comparison to surrounding savannas, riparian zones of the Australian wet-dry tropics have a higher terrestrial productivity and organic C stock (Pettit et al., 2016). These areas at the interface between the terrestrial and aquatic ecosystems can exert an important control on stream biogeochemical cycling (Naiman & Décamps, 1997). Research in temperate and boreal regions has shown that discrete points along the riparian corridor can play a significant role in transferring both DOC (Blaurock et al., 2022; Lupon et al., 2023; Ploum et al., 2021) and DIC (Leith et al., 2015; Öquist et al., 2009) to adjacent streams. It is reasonable to expect that these findings from northern regions would also apply to tropical streams. Recently Kirk and Cohen (2023) found that despite their modest spatial extent, riparian corridors supplied up to 49% of the CO_2 outgassing from a subtropical stream in Florida. While our study did not provide a conclusive answer on the role of the riparian corridor in fueling the stream C pool, indirect evidence suggests that it acted as a major source of C to Manton Creek.

Future efforts should focus on better delineating the pathways of C to tropical streams, particularly in headwater areas where the connectivity between land and aquatic environments is highest. More generally, this work illustrates the need for studies in underrepresented tropical biomes such as the wet-dry and semi-arid tropics, where C gas exchange might be different due to differences in climate patterns, topography, and terrestrial productivity.

Data Availability Statement

The data used in this paper are available in the Supporting Information. It can also be accessed in the Hydroshare repository at https://www.hydroshare.org/resource/f8d30b3669894f248de4ca415935c285.

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