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TAILORED RESTORATION RESPONSE: PREDICTIONS AND GUIDELINES FOR COASTAL WETLAND RENEWAL

RESEARCH ARTICLE

Tidal restoration to reduce greenhouse gas emissions from freshwater impounded coastal wetlands

Charles Cadier[1,2], Nathan J. Waltham[3], Adam Canning[3], Scott Fry[4], Maria Fernanda Adame[1]

Freshwater impounded wetlands are created by artificially restricting coastal wetlands connection to tides. The decrease in salinity and altered hydrology can significantly increase greenhouse gas (GHG) emissions, specifically methane (CH_4). Restoration of freshwater impounded wetlands through tidal reintroduction can potentially reduce GHG emissions; however, studies in tropical regions are scare. This study investigates the potential for tidal restoration of impounded freshwater coastal wetlands by comparing their GHG emissions with tidally connected mangrove and saltmarshes in the Burdekin catchment in Queensland, Australia. We found that freshwater impounded wetlands had significantly higher CH_4 emissions ($3,633 \pm 812$ μg CH_4 m^{-2} $hour^{-1}$) than mangroves (27 ± 8 μg CH_4 m^{-2} $hour^{-1}$) and saltmarsh (13 ± 8 μg CH_4 m^{-2} $hour^{-1}$). Soil redox, moisture, carbon, nitrogen, and bulk density were all significantly correlated to methane emissions. Conversely, freshwater impounded wetlands had significantly lower nitrous oxide (N_2O) emissions (-0.72 ± 0.18 μg N_2O m^{-2} $hour^{-1}$) than mangroves and saltmarsh (0.35 ± 0.29 and 1.32 ± 0.52 μg N_2O m^{-2} $hour^{-1}$ respectively). Nevertheless, when converting to CO_2 equivalents ($CO_{2\text{-eq}}$), freshwater impounded wetlands emitted 91 ± 20 g $CO_{2\text{-eq}}$ m^{-2} $hour^{-1}$, compared to the much lower 0.8 ± 0.2 and 0.7 ± 0.2 g $CO_{2\text{-eq}}$ m^{-2} $hour^{-1}$ emission rates for mangroves and saltmarsh. In conclusion, restoration of freshwater impounded wetlands through tidal restoration is likely to result in reduced GHG emissions.

Key words: blue carbon, mangrove, methane, nitrous oxide, saltmarsh, soil indicators

Implications for Practice

- Tidal restoration of freshwater impounded wetlands could benefit from incentives linked to avoided methane emissions.
- Soil physicochemical indicators such as redox potential, carbon and nitrogen density, moisture, and bulk density can provide useful information on potential methane and nitrous oxide emissions.
- Seasonality of emissions could be important and requires further investigation.
- Reconnecting freshwater impounded wetlands is likely to result in lower greenhouse gas emissions, however, trade-offs between services provided by ponded pastures and coastal wetlands should be considered.

Introduction

Coastal wetlands are "blue carbon" ecosystems because they accumulate significant amounts of organic carbon in their soils and can be managed for climate change mitigation and adaptation (Lovelock & Duarte 2019). Coastal wetlands ecosystems are vegetated, tidally influenced wetlands, and include mangroves, tidal marshes, and seagrass meadows. Occupying less than 2% of the ocean area, they are responsible for nearly 50% of carbon burial in marine sediments and are important carbon sinks (Duarte et al. 2013). The global area of wetlands has been rapidly decreasing over the last century due to anthropogenic

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[1]Coastal and Marine Research Centre, Australian Rivers Institute, Griffith University, Nathan, QLD 4111, Australia
[2]Address correspondence to C. Cadier, email charles.cadier@griffithuni.edu.au; f.adame@griffith.edu.au
[3]Centre for Tropical Water and Aquatic Ecosystem Research (TropWATER), College of Science and Engineering, Division of Tropical Environment and Societies, James Cook University, Townsville, QLD 4811, Australia
[4]North Queensland Dry Tropics, Natural Resources Management, Townsville, QLD 4811, Australia

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activities (Davidson 2014; Murray et al. 2022). The loss of coastal wetlands areas results in the remineralization of the carbon stored in their biomass and soil, increasing greenhouse gas (GHG) emissions (Lovelock et al. 2017). It has been estimated that the alteration of coastal wetlands increases methane (CH_4) emissions by 1–2 orders of magnitude (Al-Haj & Fulweiler 2020). The conversion of coastal wetlands to agricultural lands also increases nitrogen inputs into the soil from fertilizers, resulting in higher nitrous oxide (N_2O) emissions, another potent GHG (Snyder et al. 2009). Land use and land cover change affect coastal wetlands' carbon sequestration capacity, remineralizing carbon stored in their soil, ultimately converting them from sinks to sources (Pendleton et al. 2012; Adame et al. 2021). However, while many studies have focused on the impact of land use and land cover change on carbon stocks, fewer have analyzed the effects on coastal wetland GHG emissions (O'Connor et al. 2019; Sasmito et al. 2019).

High primary productivity and anoxic soils in coastal wetlands not only favor carbon burial, but also the production of CH_4 and N_2O (Martens & Berner 1974; Rosentreter et al. 2021). High organic carbon and nitrogen density provide the substrates that fuel methanogens and nitrifiers/denitrifiers, microorganisms responsible for CH_4 and N_2O emissions (Segers 1998; Wrage-Mönnig et al. 2018). Waterlogged conditions in these ecosystems lead to low redox potential and promote CH_4 production by methanogens (Bridgham et al. 2013) and N_2O production by denitrification (reduction of NO_2^- to N_2O by denitrifying bacteria) (Wrage-Mönnig et al. 2018). Wetlands are the single largest natural CH_4 source, emitting about a quarter of the global CH_4 emissions, while anthropogenic CH_4 emissions account for 54–72% of the total global flux (Bridgham et al. 2013; Mitsch et al. 2013; Oertel et al. 2016). Salinity also strongly affects CH_4 emissions, with polyhaline tidal marshes (salinity >18‰) emitting significantly less than other marshes (Bartlett et al. 1987; Poffenbarger et al. 2011). This is due to the higher concentration of sulfate in seawater, which favors sulfate reducers over methanogens in organic matter turnover (Wang 1996; Trevathan-Tackett et al. 2021). Therefore, with high carbon stocks and low CH_4 emissions, coastal wetlands are considered a natural climate solution for reducing GHG emissions (Taillardat et al. 2020; Macreadie et al. 2021).

Restoring and increasing coastal vegetation is a nature-based solution approach that is gaining more interest from managers to mitigate and adapt to climate change (Gattuso et al. 2021). There are various emerging activities and policies that support nature-based solutions to reduce emissions, such as the Reducing Emissions from Deforestation and Forest Degradation program (REDD+; IPCC 2022), payments for ecosystem services (Thomas 2014), and Nationally Appropriate Mitigation Actions (Kelleway et al. 2020). These have led to a rising interest in the potential for coastal wetland restoration to offset GHG emissions (Basconi et al. 2020). However, CH_4 emissions from coastal wetlands are variable, and while the relationship between salinity and CH_4 is well established for tidal marshes (Poffenbarger et al. 2011), recent studies have demonstrated that it does not hold for other coastal wetlands such as mangroves and seagrass meadows (Rosentreter et al. 2018; Al-Haj & Fulweiler 2020). Furthermore, while wetlands are hotspots of denitrification, a primary natural source of N_2O emissions in the atmosphere (Kulkarni et al. 2008; Wrage-Mönnig et al. 2018), there remains uncertainty about whether these ecosystems act as global sources or sinks for N_2O due to a paucity of studies investigating N_2O fluxes in coastal wetlands (Rosentreter et al. 2021). Therefore, there is a critical need to investigate GHGs fluxes changes that occur with restoration to support this nature-based solution approach.

The restriction of tidal intrusion has been an important factor contributing to coastal wetland decline (Kroeger et al. 2017; Li et al. 2018). In the Great Barrier Reef (GBR) catchments, in Australia, earth walls in the 1950s were built to provide late dry season forage for cattle and protect coastal freshwater resources from tidal intrusion, creating impounded freshwater wetlands (Challe & Long 2004; Lee et al. 2006). These freshwater wetlands also potentially support multiple services besides those to agriculture, such as biodiversity and water quality improvement (Canning & Waltham 2021). However, while wetlands provide carbon abatement services, their conversion from coastal wetlands to freshwater impounded wetlands affected ecosystem carbon balance, with increased N_2O and CH_4 emissions by a factor of 7 and 200, respectively (Iram et al. 2021). Furthermore, tidal bunds may also create a weed-choked ecosystem, with poor water quality, requiring the implementation of expensive restoration and ongoing maintenance programs to reinstate their values and services (Abbott et al. 2020; Waltham et al. 2020). Coastal wetlands impoundment alters wetlands functions through a shift in salinity and vegetation communities (Roman et al. 1984; Portnoy 1999). Hence, while there are services trade-offs from their conversion, restoring tides in artificially impounded wetlands can permanently avoid CH_4 emissions and increase soil and plant carbon sequestration (Kroeger et al. 2017; Negandhi et al. 2019).

Australia's Emissions Reduction Fund, a national voluntary scheme aiming to provide incentives to reduce emissions, has considered the introduction or reintroduction of tidal flow as one a promising mitigation strategy for meeting national carbon emission targets (Kelleway et al. 2020). To support this strategy, there is a need to investigate CH_4 and N_2O emissions under impounded conditions and after tidal flow reconnection (Kelleway et al. 2020). Increased interest in developing a coastal restoration project in the GBR provides unique opportunities to investigate baseline GHG emissions from natural and modified coastal wetlands subjected to potential tidal restoration activities (Adame et al. 2019).

The objectives of this study were to assess the CH_4, N_2O, and CO_2 emissions in freshwater impounded wetlands, mangroves, and saltmarsh, and investigate the relation between soil physicochemical properties and GHG fluxes. We hypothesized that: (1) GHGs emissions from the freshwater impounded wetlands are higher than those of mangroves and saltmarsh, suggesting that tidal restriction and restoration can significantly decrease emissions, and that (2) high surface soil carbon, nitrogen, and low redox are significantly associated to higher GHGs emissions. The outcomes will provide important information on baseline GHG emissions from natural and degraded coastal wetlands in the GBR and inform the potential for tidal restoration to contribute to meeting Australia's emission reduction targets.

Methods

Study Sites

The Burdekin catchment is in the Dry Tropics, Australia, with a mean annual rainfall of 656 mm (Australian Bureau of Meteorology [ABM] 2021). The floodplain is highly modified, containing Australia's largest sugarcane district (Davidson 2014). Kalamia Creek (19°30′06.0″S, 147°29′18.9″E) is a modified coastal wetland, with a freshwater reservoir built at the lower part of the creek to support sugarcane farming. In addition, a prawn farm at the mouth of the creek pumps saltwater in and out of the creek. The salinity of the creek between the prawn farm and the reservoir is low due to freshwater disposal from the irrigation draining network upstream, which makes its way into the upper estuary. The low salinity has contributed to the loss of saltmarsh and mangroves, being replaced with freshwater invasive plants such as *Eichhornia crassipes*, and the appearance of a eutrophic stagnant body of water in front of the reservoir (Waltham & Canning 2021). These characteristics are like those found in Plantation (19°32′03.0″S, 147°29′23.7″E) and Merryplain Creeks (19°44′05.2″S, 147°31′08.4″E), two additional sites that were sampled in this study (Fig. 1).

The three creeks were recently integrated into a management strategy plan to reduce fine sediment runoff to the GBR by maintaining and restoring stream banks and coastal wetlands. Aquatic weed removal has been conducted with the objective to enhance water quality and prevent the depletion of dissolved oxygen and improve habitat for native fish migration across the floodplain (Waltham et al. 2019). For each of the three sites, we sampled an adjacent site referred to as "tidally connected", which had relatively natural mangrove and saltmarsh vegetation, to provide reference values for comparison (Fig. 1). Mangrove forests were dominated by trees of *Ceriops* spp. and *Rhizophora* spp. at Kalamia, *Ceriops* spp., *Rhizophora* spp., and *Soneracia* spp. at Plantation Creek, and *Ceriops* spp. at Merryplain Creek. Saltmarsh species were dominated by *Salicornia* spp. and *Sporobolus virginicus*. Hence, each site was composed of three types of wetlands: freshwater impounded, saltmarsh and mangroves.

Soil and Water Physicochemical Characteristics

Soil physicochemical properties were measured in each wetland types in the three creeks. Soil oxidation–reduction potential (ORP) was measured in the top 1–2 cm of the soil using a redox meter (H.Q. 11d ORP meter, Hach), standardized for the H electrode (0 mV at 25°C). The top 5 cm of the soil was sampled for moisture and bulk density after oven-drying soil samples at 105°C for 48 hour. Dry soil samples were ground (Retsch™ mill) and analyzed for N (%) and C (%) with an elemental analyzer connected to a gas isotope ratio mass spectrometer (EA-Delta V Advantage IRMS, Griffith University). Soil C and N densities were estimated from %C, %N, and bulk density. Water physicochemical properties were measured at the closest water body (<100 m) within each site. Water temperature, pH, electrical

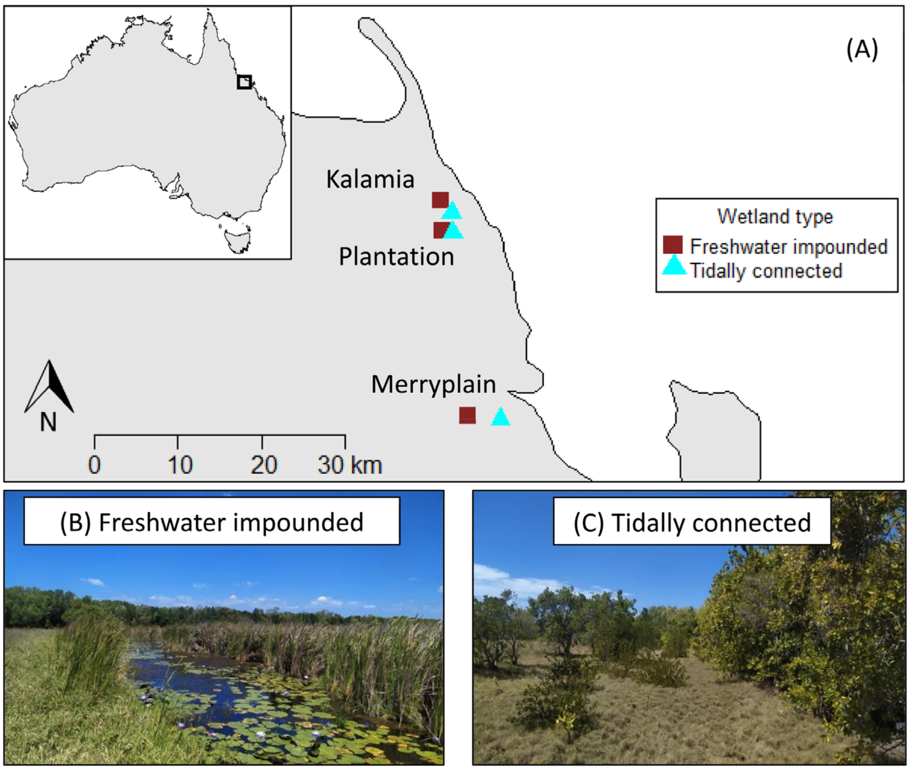

Figure 1. (A) Study sites locations in the Burdekin catchment, (B) plantation creek freshwater impounded wetland, and (C) plantation creek tidally connected wetland comprised of saltmarsh and mangroves at low tide.

conductivity, and salinity were measured using a calibrated water quality meter (ProPlus, YSI meter, Westerville, OH, U.S.A.). For nutrient analyses, surface water samples were collected using a 30-mL syringe, filtered (45 m filter), and frozen until nutrient analysis (NO_x-N and NH_4^+; colorimetric analyses based on APHA/AWWA/WPCF, 2012; Chemistry Centre, Queensland Department of Environment and Science, Brisbane, QLD, Australia).

GHG Emissions

We used static, manual chambers consisting of two units: a base ($r = 12$ cm; $h = 18$ cm) and a detachable collar ($h = 12$ cm) to measure GHG fluxes (CO_2, CH_4, and N_2O) from each site (Iram et al. 2021; Kavehei et al. 2021). We performed a preliminary sampling in Kalamia Creek site in September and December 2020 to analyze variation of GHG fluxes between the cool-dry season (June–September) and the dry-hot season (October–December), using two sets of five chambers ($n = 10$) for each wetland type. In September 2021, three sets of three chambers ($n = 9$) were set up for each wetland type in the three creeks to investigate spatial variability. The sampling was done during 1 day for each site and wetland type. The use of discrete sampling instead of continuous sampling was supported by previous experience, as GHG were most variable in space, and temporal variation within a week were small (Iram et al. 2021).

The samplings were conducted at low tide over exposed soils, where emissions of CH_4 and N_2O are likely to be highest (Kristensen et al. 2008; Rosentreter et al. 2018). Chambers were set at 5 cm deep in the soil and spaced 1–2 m apart. Trampling around the chambers was avoided during the installation of chambers and sampling. At the beginning of the experiment, the chambers were covered with lids, and air samples were extracted using a 30-mL syringe at 0, 20, 40, and 60 minutes and transferred to a 12-mL vacuumed container (Exetainer, Labco Ltd, High Wycombe, UK). GHG concentrations were analyzed using a gas chromatograph (Shimadzu GC-2010 Plus), with detection limit of 0.01 ppm for CH_4 and N_2O, and 10 ppm for CO_2. Soil temperature was recorded next to the chamber using a mercury thermometer for each sampling event, at a depth of approximately 5 cm.

After each experiment, the internal base depth of the chambers was recorded from five points to calculate headspace volume. We converted the volumetric unit concentrations to mass-based units using the ideal gas law (Hutchinson & Mosier 1981), correcting for soil temperature. The change in air pressure over time was not measured, and therefore atmospheric pressure was used to correct fluxes. The $CO_{2\text{-equivalent}}$ ($CO_{2\text{-eq}}$) values were estimated using the CH_4 and N_2O radiative balance of 25 and 298, respectively (Neubauer 2021; IPCC 2022). Dark chambers do not account for CO_2 uptake by primary production; therefore, CO_2 measurements reflect only respiration, not their full net balance. We display results as area per hour, as diurnal variation of CH_4 was not captured in this study (Sanders-DeMott et al. 2022).

Data Analysis

The assumption of normality of the data was inspected using a Shapiro–Wilk test. The GHGs emissions data from 2020 were not normal, and therefore a Scheirer–Ray–Hare test was performed using wetland type and season as predictive factors. For the GHG fluxes of 2021, only the CH_4 emissions were not normal, and thus, were log-transformed. A nested analysis of variance (ANOVA) was performed to analyze the influence of wetland type on GHGs emissions, with sites being the random factor and wetland type (freshwater impounded, mangroves, or saltmarsh) as the independent factor. When the differences were significant at $p < 0.05$, a post hoc Student-Bonferroni test was used to determine where differences lied. A principal component analysis (PCA) was performed to describe the impact of the tidal restriction on soil physicochemical parameters, and an analysis of similarities (ANOSIM) was performed to test for statistical differences of soil physicochemical parameters among wetland types. We also performed a Pearson's correlation coefficient tests to reveal relationships between these parameters and GHGs emissions. Statistical tests were performed in the R software (R Core Team 2014).

Results

Soil and Water Physicochemical Characteristics

The top 5 cm of the soil of freshwater impounded wetlands contained higher C and N density (33.5 mg C cm^{-3} and 2.5 mg N cm^{-3}) than the mangroves (17.8 mg C cm^{-3} and 1.1 mg N cm^{-3}) and saltmarsh (12 mg C cm^{-3} and 0.9 mg N cm^{-3}), particularly at Plantation Creek. Redox potential was consistently negative in freshwater impounded wetlands (−100 mV), while it was always positive in mangroves and saltmarsh (49 mV and 82 mV, respectively; Table 1). Soil density was higher in mangroves and saltmarshes (between 0.9 and 1.2 g cm^{-3}) compared to freshwater impounded wetlands in Plantation and Merryplain Creek (below 0.7 g cm^{-3}). However, in Kalamia Creek, mangroves had the lowest soil density (below 0.7 g cm^{-3}).

Water temperature recorded during sampling events was between 20.7 and 27.8°C. Kalamia and Merryplain tidal creeks had higher salinity (>26‰) compared to Plantation Creek (13.8‰; Table 2). The highest salinity recorded in freshwater impounded was 2.7‰ in Merryplain Creek. Freshwater impounded wetlands had higher pH, higher NO_x, and lower NH_4^+ than tidal creeks in Kalamia and Plantation Creek sites. This pattern was reversed in Merryplain Creek. The PO_4^- concentrations were higher in Kalamia and Merryplain Creek freshwater impounded, while Plantation's tidal creek had a higher PO_4.

The PCA showed differences in soil physicochemical parameters among wetland types (Fig. 2), with treatment clusters clearly and significantly separated over the horizontal axis (ANOSIM, $R^2 = 0.44$, $p < 0.01$). The PCA1 axis, which explained 70.2% of the soil physicochemical parameters variance between the plots, was likely to be representing the effect of the treatments. The PCA2 axis, explaining 15.2% of this variance, could be representing the effect of sites. The main vectors positively correlated to freshwater impounded variation are soil moisture and carbon and nitrogen density. The main vectors

Table 1. Soil physicochemical parameters (average ± SE) of freshwater impounded, mangrove, and saltmarsh in the Burdekin catchment s during GHGs measurements. *T*, temperature.

Site	*Wetland types*	*Soil T (°C)*	*Air T (°C)*	*Redox (mV)*	*Soil moisture (%)*	*Soil density ($g\ cm^{-3}$)*	*%C*	*%N*	*Carbon density ($mg\ cm^{-3}$)*	*Nitrogen density ($mg\ cm^{-3}$)*
Kalamia	Freshwater impounded	21.9	27	−65 ± 44	38.5 ± 1.7	0.78 ± 0.04	4.17 ± 0.6	0.35 ± 0.06	32.6 ± 4.7	2.8 ± 0.5
	Mangrove	22.9	24.4	28 ± 4	38.6 ± 3.3	0.68 ± 0.03	3.38 ± 0.3	0.19 ± 0.02	22.8 ± 2.2	1.3 ± 0.1
	Saltmarsh	23	23.6	78 ± 19	9.3 ± 2.2	0.96 ± 0.13	1.24 ± 0.5	0.09 ± 0.03	10.7 ± 3.8	0.8 ± 0.3
Plantation	Freshwater impounded	19.5	26.6	−40 ± 19	60.7 ± 7.5	0.38 ± 0.12	13.5 ± 4.5	0.85 ± 0.26	43.1 ± 8.2	2.8 ± 0.7
	Mangrove	21	26.9	58 ± 8	27.2 ± 1.3	0.92 ± 0.02	1.64 ± 0.08	0.10 ± 0.005	15.1 ± 0.8	1.0 ± 0.07
	Saltmarsh	28.9	25.6	96 ± 7	16.4 ± 6.5	0.99 ± 0.02	1.93 ± 0.6	0.16 ± 0.04	17.9 ± 5.6	1.4 ± 0.3
Merryplain	Freshwater impounded	22.5	22.5	−195 ± 30	45.5 ± 7.2	0.66 ± 0.21	4.50 ± 1.5	0.34 ± 0.11	24.8 ± 4.2	1.9 ± 0.3
	Mangrove	21	25	61 ± 29	23.7 ± 1.6	1.20 ± 0.09	1.31 ± 0.2	0.09 ± 0.01	15.4 ± 1.0	1.0 ± 0.07
	Saltmarsh	28.2	21.5	73 ± 10	20.2 ± 2.3	1.16 ± 0.12	0.66 ± 0.1	0.06 ± 0.007	7.4 ± 0.5	0.6 ± 0.05

positively related to mangroves and saltmarshes clusters are soil density and redox. Freshwater impounded wetlands are characterized by higher carbon and nitrogen density, and high soil moisture, while mangroves and saltmarshes are characterized by higher redox and soil density compared to the freshwater impounded wetlands.

GHG Emissions

The CO_2 and N_2O emissions in 2020 were statistically different between the cool and hot dry seasons in Kalamia Creek in 2020 ($p < 0.01$; Table S5), with higher emissions in the dry hot season in the freshwater impounded and saltmarsh (Fig. S1). The CH_4 emissions were similar between the cold and hot dry season ($p = 0.29$; Table S5) but were influenced by wetland type ($p < 0.01$; Table S5), with higher emissions in freshwater impounded wetlands.

In September 2021, freshwater impounded wetlands were a net source of CH_4, emitting on average 3,633 ± 812 μg CH_4 m^{-2} $hour^{-1}$, while mangroves and saltmarsh were emitting 27 ± 8 and 13 ± 8 μg CH_4 m^{-2} $hour^{-1}$ (Fig. 3A). Treatment had a significant effect on CH_4 emissions ($p < 0.01$; Table S1), with freshwater impounded wetlands emitting significantly more CH_4 than mangroves and saltmarsh in all sites tested ($p < 0.01$, Table S2). Although there was variability in CH_4 emissions within the different sites for freshwater impounded wetlands, they were not statistically different ($p > 0.05$, Table S3).

Freshwater impounded wetlands were sinks of N_2O in all sites (−0.72 ± 0.18 μg N_2O m^{-2} $hour^{-1}$); while mangroves and saltmarshes were sources (0.35 ± 0.29 and 1.32 ± 0.52 μg N_2O m^{-2} $hour^{-1}$, respectively) (Fig. 3B). Similar to CH_4, there was a significant effect of wetland types on N_2O emissions ($p < 0.05$; Table S1). Freshwater impounded wetlands were emitting significantly less N_2O than saltmarsh in Kalamia ($p < 0.01$) and Plantation Creek ($p < 0.01$), and significantly less N_2O than mangroves in Merryplain Creek ($p < 0.01$, Table S2). There was no effect of sites on N_2O emissions.

Emissions of CO_2 were similar between wetland types and sites, with freshwater impounded emitting 220 ± 17.8 mg CO_2 m^{-2} $hour^{-1}$, mangroves emitting 150 ± 23 mg CO_2 m^{-2} $hour^{-1}$ and saltmarshes emitting 248 ± 32 mg CO_2 m^{-2} $hour^{-1}$ (Fig. 3C). There was, however, an effect of wetland type on CO_2 emissions within and among sites ($p < 0.01$, Table S1). In Plantation Creek, saltmarsh was emitting significantly more CO_2 than freshwater impounded and mangrove wetlands ($p < 0.01$; Table S2). Plantation Creek saltmarsh was emitting significantly more CO_2 than Merryplain Creek saltmarsh ($p < 0.05$), while Merryplain Creek mangroves were emitting significantly more CO_2 than Plantation Creek mangroves ($p < 0.05$; Table S3).

GHG Emissions and Soil Physicochemical Characteristics

Redox ($R^2 = -0.65$, $p < 0.01$), carbon ($R^2 = 0.72$, $p < 0.01$) and nitrogen density ($R^2 = 0.70$, $p < 0.01$), soil moisture ($R^2 = 0.64$,

Table 2. Water physicochemical parameters of freshwater impounded and tidal creeks in the Burdekin catchment during GHGs measurements.

Site	*Wetland types*	*pH*	*Salinity (‰)*	*Water Temperature (°C)*	*NO_x-N (mg L^{-1})*	*NH_4-N (mg L^{-1})*	*PO_4-P (mg L^{-1})*
Kalamia	Freshwater impounded	6.56	0.27	24.5	0.027	0.021	0.211
	Tidal creeks	6.42	29.9	24.3	0.007	0.041	0.018
Plantation	Freshwater impounded	7.26	0.20	20.7	0.009	0.015	0.022
	Tidal creeks	6.3	13.79	20.9	0.006	0.060	0.060
Merryplain	Freshwater impounded	7.04	2.72	27.8	0.023	0.066	0.115
	Tidal creeks	7.53	26.25	22.8	0.065	0.008	0.024

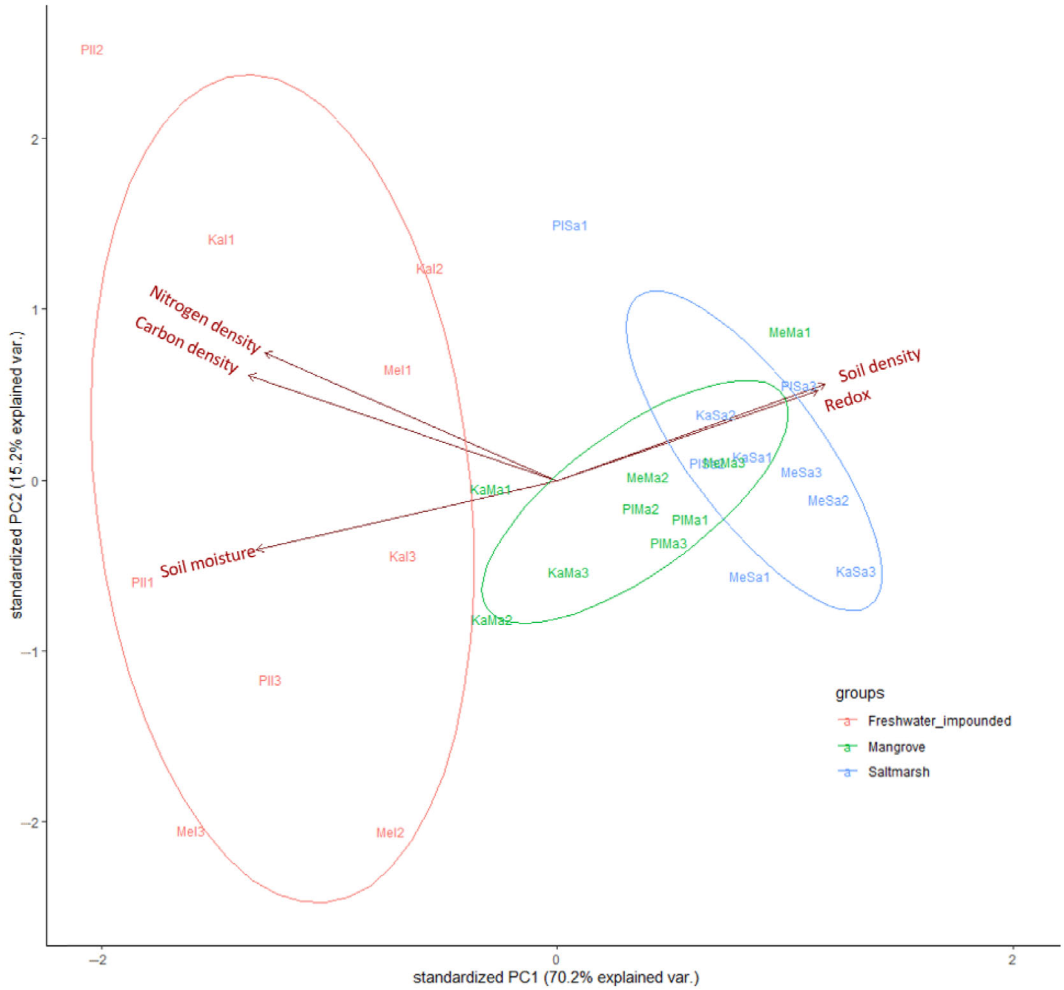

Figure 2. PCA of soil physicochemical parameters among site plots ($n = 9$) (ka, Kalamia; me, Merryplain; pl, plantation; I, impounded; ma, mangroves; Sa, saltmarsh).

$p < 0.01$), and bulk density ($R^2 = -0.49$, $p < 0.05$) were all significantly associated with CH_4 emissions ($n = 27$; Table S4). Higher CH_4 emissions were measured at sites with high soil moisture, carbon, and nitrogen density, while lower CH_4 emissions were measured at sites with high soil redox (>0 mV) and bulk density (>90 g cm^{-3}). In addition, redox ($R^2 = 0.48$, $p < 0.01$), soil moisture ($R^2 = -0.49$, $p < 0.01$), and bulk density ($R^2 = 0.51$, $p < 0.01$) were strong determinants on recorded N_2O emissions ($n = 27$). High N_2O emissions were measured in sites with high soil redox and bulk density, while lower N_2O emissions were found in sites with high soil moisture (>40%). No single soil physicochemical parameters were associated with CO_2 emissions ($p > 0.05$; $n = 27$).

Discussion

Emissions of GHGs in coastal wetlands are a considerable aspect of their carbon sequestration potential, as they can offset

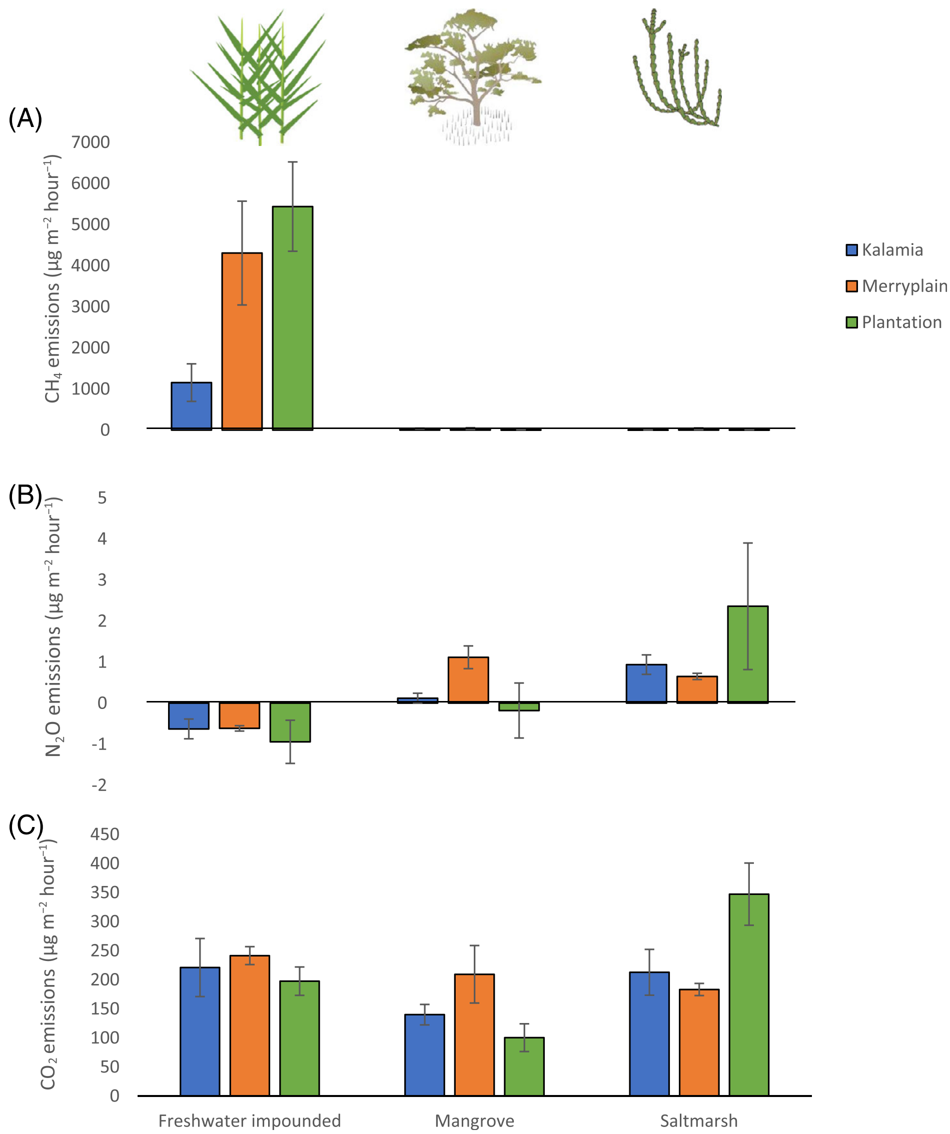

Figure 3. Average GHGs emissions (± SE) of CH_4 (μg m^{-2} $hour^{-1}$) (A), N_2O (μg m^{-2} $hour^{-1}$) (B), and CO_2 (mg m^{-2} $hour^{-1}$) (C) from each wetland type in Kalamia, Merryplain, and plantation creeks.

their carbon burial capacity (Al-Haj & Fulweiler 2020). Tidal restoration of freshwater impounded wetlands is an important climate mitigation strategy if it leads to a decrease in GHG emissions (Kroeger et al. 2017). Establishing baseline GHG emissions from freshwater impounded wetlands is essential to understand the potential for tidal restoration to reduce GHG emissions (Sanders-DeMott et al. 2022). Our study was designed to provide baseline data and outline the potential for tidal restoration as a climate mitigation strategy.

The freshwater impounded wetlands examined here were emitting more CH_4 than tidal wetlands. The restriction of tides in our study sites has transformed saline into freshwater wetlands, which has driven increased CH_4 emissions (Poffenbarger et al. 2011). Conversely to our hypothesis, freshwater impounded wetlands were net sinks of N_2O flux in September 2021, while tidal wetlands were generally small sources of N_2O. Overall, this reduction in N_2O emissions did not compensate for the increase in CH_4 emissions, with freshwater impounded wetlands emitting 91 ± 20 g $CO_{2\text{-eq}}$ m^{-2} $hour^{-1}$ compared to 0.8 ± 0.2 and 0.7 ± 0.2 g $CO_{2\text{-eq}}$ m^{-2} $hour^{-1}$ for mangroves and saltmarsh, respectively. Emissions of CO_2 associated with soil respiration were similar among wetland types, which supports the results from the review by Sasmito et al. (2019). Soil physicochemical properties of freshwater impounded wetlands were characterized by lower redox (≤40 mV), higher topsoil carbon (≥24 mg cm^{-3}), and nitrogen (≥1.9 mg cm^{-3}), and soil bulk densities compared to those of mangroves and saltmarshes.

Freshwater impounded soils emit on average 137 and 278 times more CH_4 than mangrove and saltmarsh soils, respectively, supporting the hypothesis developed by Kroeger et al. (2017). Furthermore, emissions of CH_4 did not vary between the cool-dry and hot-dry season of 2021. Thus, tidal restriction creating freshwater impounded conditions in coastal wetlands is likely the main factor leading to high CH_4 emissions. High organic matter availability is another driver of CH_4 emission (Segers 1998). The higher emissions in the Plantation Creek freshwater impounded wetland, up to 700 times more than in the adjacent mangrove and saltmarsh, can be explained by their high surface soil carbon density, which was much higher than the other wetlands examined here. Merryplain Creek freshwater impounded wetland had the lowest redox potential, creating ideal conditions to produce CH_4 by methanogens (Bridgham et al. 2013). The positive correlation found between these soil parameters and CH_4 emissions supports these statements and explains the lower CH_4 emissions found in Kalamia Creek.

Freshwater impounded wetlands acted as sink of N_2O in September 2021, while mangroves and saltmarsh acted as sources of N_2O. Previous studies have displayed similar results with lower N_2O emissions in tidally restricted compared to tidally connected wetlands (Yang et al. 2017; Tan et al. 2021). This could be due to the capacity for freshwater wetland to act as sink of N_2O when vegetative growth and microbial activities are high (Kolb & Horn 2012; Zhang et al. 2021). However, N_2O was subject to high seasonal variation, with higher emissions in hot-dry season; therefore, our results from September 2021, a cold-dry season, are likely to be underestimating coastal wetlands N_2O emissions. The versatility of freshwater wetlands to act as sources or sinks of N_2O is supported by the high variability of N_2O emissions among seasons found in freshwater ponded pastures (Iram et al. 2021), influenced by dissolved inorganic nitrogen availability and water residence time (Moseman-Valtierra et al. 2011; Maavara et al. 2019). Further studies describing seasonality of N_2O emissions in freshwater impounded wetlands and tidal creeks could unravel the impact of the tidal restriction on N_2O emissions.

Emissions of CH_4 from our study sites for mangroves (27 ± 8 μg CH_4 m^{-2} $hour^{-1}$) and saltmarsh (13 ± 8 μg CH_4 m^{-2} $hour^{-1}$) were within the lower end of the global range for mangroves (-45 to 48,000 μg CH_4 m^{-2} $hour^{-1}$) and saltmarsh (-62 to 63,000 μg CH_4 m^{-2} $hour^{-1}$; Al-Haj & Fulweiler 2020). Similarly, the range of N_2O emissions in mangroves (0.35 ± 0.29 μg N_2O m^{-2} $hour^{-1}$) was within the lower end of the global range, from -8.3 to 262 μg N_2O m^{-2} $hour^{-1}$ (Murray et al. 2015, 2018; Maher et al. 2016; Rosentreter et al. 2021). Saltmarsh had higher N_2O emissions (1.32 ± 0.52 μg N_2O m^{-2} $hour^{-1}$) than mangroves, similar to other results reported in the literature (Iram et al. 2021; Rosentreter et al. 2021). Overall, our results support the hypothesis that tidal coastal wetlands are a low source of GHG (Zheng et al. 2022).

Our study contributes to the recognition of tidal restoration as a strategy to reduce GHG emissions. We demonstrated that restriction of tidal flows in these ecosystems leads to a significant increase in CH_4 emissions, increasing the radiative force of coastal wetlands. These results are similar to previous studies in other tropical regions of Australia, which showed that ponded freshwater wetlands emit many more times CH_4 than the natural coastal wetlands they usually replaced (Iram et al. 2021). A recent study carried on in North America also displayed that freshening impounded conditions led to a 50-fold increase in CH_4 emissions compared to saline tidal wetlands (Sanders-DeMott et al. 2022). Furthermore, a meta-analysis of the impact of converting coastal wetlands into constructed wetlands, cropland, or aquaculture ponds found an increase in the global warming potential associated with land use and cover change by a factor of 7 to 25 (Tan et al. 2020). Together with our results, these studies outline the prospects of tidal restoration as a strategy to reduce GHG emissions. Although the limited extent impedes them from making any substantial global impact on the climate, they are particularly relevant for countries seeking to meet climate change mitigation targets on a national scale (Taillardat et al. 2020). Tidal ingress into previously ponded wetlands has been included in Australia's first blue carbon method through the Emission Reduction Fund as an important first step forward toward emission reductions (Lovelock et al. 2022). It will create opportunities to generate carbon credits from carbon sequestration and avoided GHG emissions (Kelleway et al. 2020).

Tidal restoration could also lead to a wide range of co-benefits, such as increasing biodiversity and providing coastal protection, leading to a more resilient and functional GBR ecosystem (Barbier et al. 2011; Costanza et al. 2017; Adame et al. 2019). However, tidal restoration of previously impounded wetlands could also decrease artificial freshwater wetland habitats in the GBR; some of these wetlands are currently the only habitat remaining for freshwater species, including many birds, which have lost their habitat to agriculture in the lower floodplain (Canning & Waltham 2021). Furthermore, there is an important variation of CH_4 emissions in freshwater impounded wetlands, and a study has displayed that some may act as sinks of CH_4 (Negandhi et al. 2019). Restoring coastal wetlands through tidal restoration may be at the cost of losing the ecosystem services provided by these artificial freshwater wetlands without decreasing GHGs emissions. Tidal restoration projects in the GBR should consider the values of each wetland type and avoid the "one size fits all" approach, which could lead to inappropriate restoration outcomes (Canning & Waltham 2021). The use of relevant indicators related to restoration objectives would enable stakeholders to find the appropriate sites to undergo tidal restoration and facilitate its success (Cadier et al. 2020). Although broader ecosystem service values need to be considered, tidal reconnection of impounded freshwater wetlands could provide a GHG emission reduction option in advancing toward Australia's carbon reduction emission targets.

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Supporting Information

The following information may be found in the online version of this article:

Table S1. Multifactorial ANOVA table for the effect of wetland types
Table S2. Nested ANOVA post hoc results
Table S3. Nested ANOVA post hoc results
Table S4. Pearson correlation analyses
Table S5. Scheirer–Ray–Hare table for the effect of wetland types
Figure S1. Average GHGs emissions
Figure S2. Kalamia creek sites
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