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Greenhouse gas emissions from tropical coastal wetlands and their alternative agricultural lands: Where significant mitigation gains lie

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Abstract. Tidal coastal wetlands are crucial in regulating the global carbon budgets through carbon sequestration and greenhouse gas (GHG; CO_2, CH_4 and N_2O) emissions. The conversion of tidal coastal wetlands to agriculture land alters soil processes changing GHG emissions. However, our understanding about GHG emissions associated with land-use change of coastal wetland is limited. We measured soil GHG fluxes from mangroves, saltmarsh and freshwater tidal forest and their alternative agricultural lands including sugarcane and ponded pastures. We investigated seasonal variations in soil GHG fluxes between June 2018 and February 2020 in tropical. Australia. The wet ponded pasture had by far the highest CH_4 emissions with 1,231±386 mg m^{-2} d^{-1}, which were 200-fold higher than any other land use. Agricultural lands were the most significant sources of N_2O emissions with 55±9 mg m^{-2} d^{-1} from dry ponded pasture (wet-hot period) and 11±3 mg m^{-2} d^{-1} from sugar cane (hot-dry period), coinciding with fertilisation. The N_2O fluxes from the tidal coastal wetlands ranged between -0.55±0.23 and 2.76±0.45 mg m^{-2} d^{-1} throughout the study period. The highest CO_2 fluxes of 20±1 g m^{-2} d^{-1} were from the dry ponded pasture during the wet-hot period, while the saltmarsh had the lowest CO_2 fluxes having an uptake of -1.19±0.08 g m^{-2} d^{-1} in the dry-hot period. Overall, agricultural lands had significantly higher total cumulative GHG emissions (CH_4 + N_2O) of 7142 to 56,124 $CO_{2\text{-eq}}$ kg ha^{-1} y^{-1} compared to those of any type of tidal coastal wetlands, which ranged between 144 and 884 $CO_{2\text{-eq}}$ kg ha^{-1} y^{-1}. Converting agricultural land, particularly wet ponded pasture, to tidal coastal wetlands could provide large GHG mitigation gains and potential financial incentives.

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1 Introduction

Coastal wetlands are at the interface between terrestrial and marine ecosystems and account for 10% of the global wetland area (Lehner and Döll 2004; Yang et al. 2017). Tropical coastal wetlands are highly productive ecosystems with significant potential for providing various services such as water quality improvement, biodiversity and carbon sequestration and storage (Lal, 2008; Duarte et al, 2013; Mitsch et al, 2013). Coastal wetlands, such as mangroves and marshes, have sediments can accumulate a considerable amount of organic carbon in their soils (Kauffman et al, 2020) having a carbon sink potential of up to five times higher than terrestrial forests (Mcleod et al, 2011; Sjögersten et al, 2014). However, the anoxic soil conditions that promote carbon sequestration in coastal wetlands can also favour greenhouse gas emissions (GHGs), which contribute to global warming (Whalen, 2005; Conrad, 2009).

Coastal wetlands exchange three main GHGs with the atmosphere: carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). Sediments of coastal wetlands have spatially and temporally diverse conditions which affect the microbial processes that regulate their soil-atmospheric exchange (Bauza et al, 2002; Whalen, 2005). Emission of CO_2 from wetlands can be considered recycling of the carbon fixed through primary production, which is partly returned to the atmosphere through respiration. Emission of CH_4 is a product of anaerobic and aerobic respiration in wetland soil (Angle et al., 2017; Saunois et al, 2020). While N_2O fluxes are mainly driven by soil moisture and mediated by microbial activity under anoxic condition through denitrification and during aerobic condition through nitrification (Ussiri and Lal 2013). The total GHG budget of coastal wetlands is the result of the balance between aerobic and anaerobic sediment conditions that influence GHG fluxes from the main sources including soil and vegetation. Therefore, there are high uncertainties around global wetland GHG emissions (Kirschke et al. 2013; Oertel et al. 2016).

Despite potential GHG emissions from coastal wetlands, these are still likely to be lower than those from their alternative agricultural land uses. Agricultural practices are responsible for large amounts of GHG from their construction and throughout their productive life. Firstly, when wetlands are converted to agricultural land, the oxidation of sequestered carbon in the organic-rich soils release significant amounts of CO_2 (Kauffman et al, 2015, 2018), contributing substantially to emissions caused by land-use change (Ciais et al, 2013). Secondly, removing tidal flow and reverting coastal wetlands to freshwater ecosystems, such as during the creation of ponded pastures or dams, could result in high CH_4 and N_2O emissions (Martin et al, 2015; Maucieri et al, 2017; Capooci et al, 2019). Thirdly, the impoundment of ponds and dams can dramatically increase both CH_4 and N_2O emissions which are largely unaccounted for in GHG budgets from agriculture (Ollivier et al, 2019). Finally, the use of N fertilisers in intensive agricultural systems result in significant N_2O emissions (Rashti et al, 2015). Fertilisers may also increase CO_2 emissions by changing the balance between carbon and nitrogen within the aerobic condition of agricultural lands. Comprehensive studies accounting for simultaneous measurement of three main GHGs from wetlands and their alternate agricultural lands are very limited.

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Land-use change affects different soil properties such as microbial communities (Van Leeuwen et al, 2017), enzyme activity (Niemi et al, 2005), soil water content (Grover et al, 2012), nutrient and terminal electron acceptor availability, and inundation (Xu, Wong and Reef, 2020), all of which affect GHG emissions (Pouyat et al, 2007). These emissions are likely to be highest in tropical conditions as temperature is one of the main drivers (Oertel et al, 2016). Of the three main GHGs, CO_2 has been well studied and incorporated into global climate change models, while models that include CH_4 and N_2O still have many uncertainties (Arneth et al, 2010; Zaehle and Dalmonech, 2011; Flato et al, 2014). Emissions of CH_4 and N_2O need to be accounted for when determining the net radiative balance of an ecosystem due to their high radiative forcing potential of these gases compared to CO_2. Some of the uncertainties in these models arise from the role of natural wetlands as "sinks" or "sources", which can only be established when all GHG emissions have been accounted for (Cobb et al., 2012), with only a few complete GHG budgets compiled for tropical wetlands (e.g. Mitsch et al. 2013; Wang et al, 2014; Gütlein et al., 2018; Harris et al, 2018).

Reducing GHG emissions in tropical coastal regions can be achieved through different management and restoration mechanisms, such as the reinstallation of tidal inundation on unused agricultural land (Kroeger et al, 2017). Reinstating tidal exchange to previously drained, or ponded freshwater agricultural land can mitigate GHG emissions through several mechanisms. Tidal coastal wetlands contain sulphate, which can inhibit or reduce the production of CH_4 through the competitive metabolic edge of sulphate reducers over methanogens (Poffenbarger et al, 2011). Tidal coastal wetlands also have high primary productivity rates leading to above ground and below ground carbon production and storage (Burden et al, 2013). Finally, tidal exchange also facilitates sediment carbon accumulation through increased sediment accretion due to sediment supply through tidal inundation (Burden et al, 2013). However, the land-use change related GHG emissions could be useful for initiating restoration strategies that rely upon financial incentives such as carbon credits, as these could be a valuable mitigation strategy.

The emission factors or changes in GHG emission, from an agricultural land-use to a coastal wetland are required to evaluate the mitigation capacity of a restoration project. However, this information is very scarce in tropical regions (IPCC 2013). In mid 2000s, Australian greenhouse office initiated a national carbon accounting scheme to track sinks and sources of GHG fluxes. The purpose of the scheme was to help Australia stay on track to reduce GHG emission targets set by Kyoto Protocol (Beringer et al., 2013). However, large knowledge gaps were found in GHG flux estimates mainly due to spatial variation in ecosystems and soil types across Australia (Baldock et al., 2012). Therefore, GHG flux assessment from different land covers is important to develop emissions reduction policies. This study will fill in this knowledge gap to improve wetland accounting in GHG mitigation strategies (IPCC 2013). Additionally, the information on GHG emissions from land use management particularly in tropical climate would improve our understanding of reducing GHG emissions by different management implication paradigms (IPCC 2013).

In this study, we measured the annual GHG fluxes from different land-use types including natural tidal coastal wetlands (freshwater tidal forest, saltmarsh, and mangroves) and agricultural lands (sugarcane plantation and ponded pastures) in tropical Australia. We aimed to assess the potential benefit of restoring wetlands in agricultural landscapes for decreasing GHG emissions. Based on these case studies, our objective was to provide some management implication to reduce GHG emissions from agricultural lands in tropical regions. We hypothesised that GHG emissions from agricultural lands would be larger than those of natural wetlands and that emissions would be higher in the hot-wet season and during high tides.

2 Materials and Methods

2.1 Study sites

Wetlands in Queensland had undergone deforestation because of rapid agricultural development in the last century (Griggs. 2017). These wetlands were converted to agricultural land, mainly for sugarcane farming and grazing (Wetland*Info*, 2016). The study area is located within the Herbert River catchment in Queensland, northeast Australia (Fig 1a). The Herbert basin covers 9,842 km^2, from which 56% is used for grazing mainly ponded pastures, 31% is conserved natural land use including wetlands and forestry, 8% is sugarcane and 4% is other land uses (Wetland*Info*, 2020). The region has a tropical climate with mean monthly minimum temperature ranging from 14 to 23˚C and mean monthly maximum temperature ranging from 25 to 33˚C (Australian Bureau of Meteorology, ABM, 2020; 1968-2020). The average rainfall is 2,158 mm y^{-1} with the highest rainfall of 476 mm in February (ABM 2020; 1968-2020).

We selected five sites including three natural wetlands (Fig. 1b); a freshwater tidal forest ('18º53'45' 'S, 146º15'52"E), a saltmarsh (18º53'43' 'S, 146º15'52"E) and a mangrove forest (18º53'42' 'S, 146º15'51"E), and two common agricultural land use types in the area, sugarcane plantation (18º53'44. '6' ’S, 146º15'53.2"E) and a ponded pasture for fodder grazing. The pasture contained different levels of ponding including shallow ponds (50-100 cm depth) and wet grassy areas hereafter called "wet ponded pasture" (' 18º43'8"S, 146º15'50"E) and a "dry ponded pasture" (18º43'7"S, 146º15'50"E). All the sites were located within the same property at Insulator Creek (18°53'44.6"S 146°15'53.2"E, Fig. 1A) except for the ponded pasture, which was located about 20 km north at Mungalla Station (18°42'26"S, 146°15'18.37"E, Fig. 1A). The selected freshwater tidal freshwater wetland was dominated by *Melaleuca quinquenervia* trees, a forest type commonly known as "tea tree swamp". Seawards we sampled a saltmarsh dominated by *Sueda salsa* and *Sporobolus spp* followed by mangrove forests dominated by *Avicennia marina* with few plants of *Rhizophora stylosa* (Fig. 1B). Landwards, the wetlands were adjacent to a sugarcane farming area of ~110 ha. The sugarcane was fertilised once a year with urea at a rate of 135 kg N ha-1 and harvested during May-June, while foliage was left on the soil surface (trash blanket) after harvest. The ponded pastures in Mungalla Station extended over 2,500 ha and supported ~900 cattle throughout the year by providing fodder to

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cattle during dry periods. The selected ponded pasture was covered by *Eichhornia crassipes* (water hyacinth) and *Hymenachnae amplexicaulis* (Fig. 1C).

We conducted four sampling campaigns on the selected sites for soil and gas sampling for three periods of the year which characterise the main climatic variability of the region: wet and hot (17-24 February 2020), dry and cool (17 June 2018; 31 May -7 June 2019), and a dry and hot (21-29 Oct 2018; Table 1). We conducted measurements on three days for each land use and ecosystem type (Livesley et al. 2009), except for the first sampling during the dry-cool period of 2018, when only mangroves, saltmarsh and sugarcane were surveyed for one day. The effect of tidal inundation on GHG emissions of mangroves, saltmarsh and freshwater tidal forest was also tested during the cool-dry period of 2019 by measuring GHG emissions during a low (0.7m), and a high tide (2.8m) (tidal range from Lucinda town, Lat: 18° 31' S; 146°23'E).

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Table 1. Mean daily air temperature and rainfall at the Ingham weather station during sampling.

Season	Study period	Daily min Temp (°C)	Daily max temperature °C	Rainfall (mm d^{-1})
Wet and hot	17-22, Feb, 2020	23.9 - 25.3	33.6 - 34.5	0-86
Dry and cool	17 June 2018	13.4 - 14.6	27.7 - 28.2	0
Dry and cool	31 May -6, June 2019	10.9 - 17.5	21.6 - 28.2	0-25
Dry and hot	23-29 Oct 2018	15.7 - 21.1	32.2 - 36.2	0

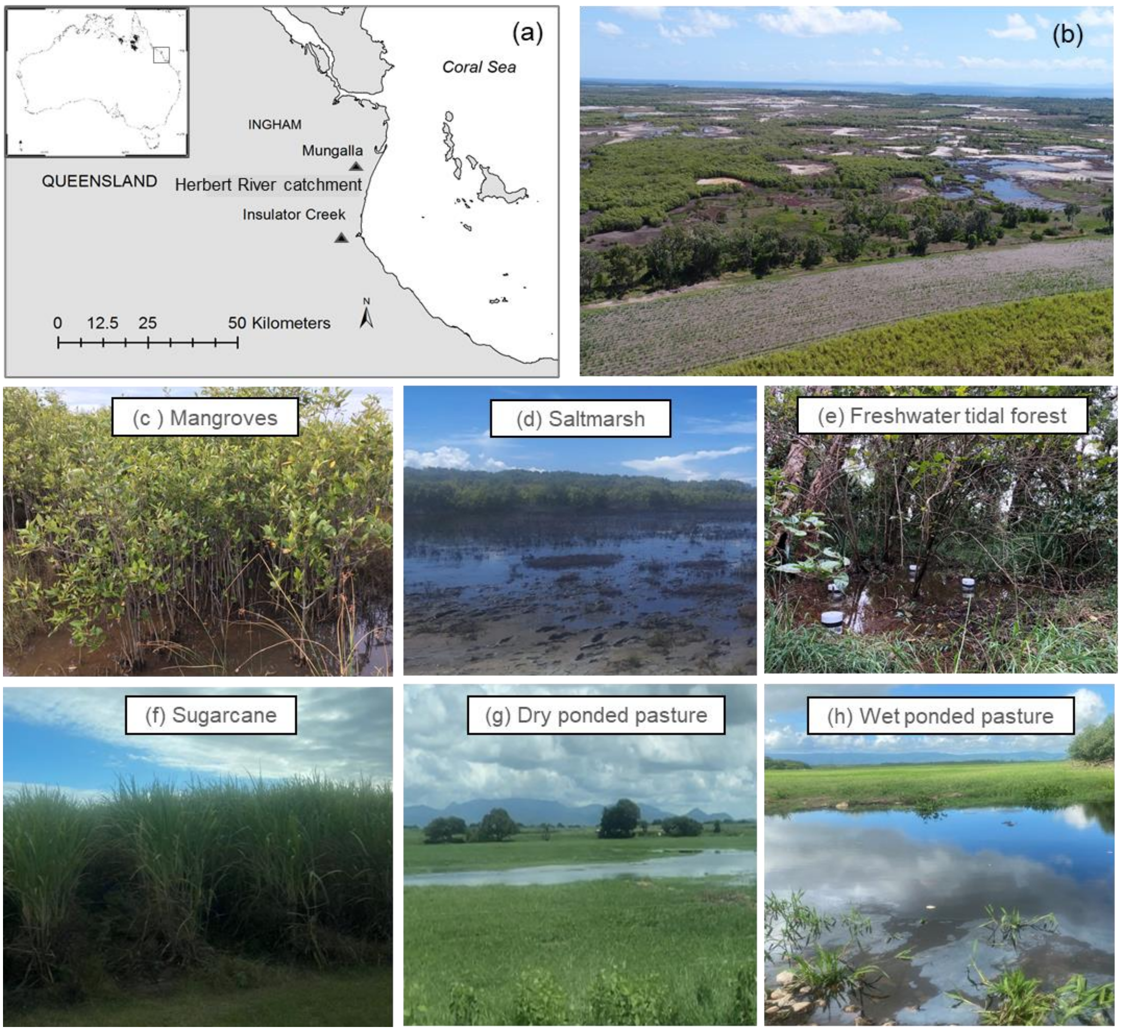

Figure 1: a) Location of sampling sites (Insulator Creek and Mungalla) in the Herbert River catchment, northeast Australia, (b) natural wetlands adjacent to sugarcane farm in Insulator Ck, and (c) mangroves, (d) saltmarsh, (e) freshwater tidal forest, (f) sugarcane, (g) dry ponded pasture and (h) wet ponded pasture. Pictures by N. Iram and MF Adame.

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2.2 Soil sampling and analysis of physicochemical properties

Soil physicochemical characteristics were measured by taking composite soil samples (n = 5; 0-30 cm) next to each gas chamber location from all study sites during the dry-hot season. The samples were obtained by inserting an open steel corer to a depth of 30 cm; the core was divided into three depths: 0-10cm, 10-20cm and 20-30 cm. Soil samples were dried in the oven at 105 ºC for 48 h to determine volumetric water content through gravimetric analysis. The volumetric water content was divided by total soil porosity to determine water-filled pore space (WFPS). Total soil porosity was calculated through the -equation: total soil porosity= 1- (soil bulk density / 2.65) where soil particle density was assumed at 2.65 g cm^{-3} (Rashti et al, 2015). Soil texture analysis (% sand, % silt, % clay) was carried for each site following a simplified method for particle size determination (Kettler et al, 2001). Soil electrical conductivity (EC) and pH were measured using a conductivity meter (WP-84 TPS, Australia) in soil/water slurry at 1:5. Soil subsamples were air-dried, sieved (2mm), ground (Retch™ mill) and analysed for %N and %C using elemental analyser connected to a gas isotope ratio mass spectrometer (EA-Delta V Advantage IRMS, Griffith University). Additionally, soil samples from the top 10 cm were collected in each sampling event to measure gravimetric soil moisture content and bulk density.

2.3 Greenhouse gas fluxes

The static manual gas chambers were used to measure GHG (CO_2, CH_4 and N_2O) fluxes from each site (Hutchinson and Mosier, 1981; Kavehei et al, 2021). The gas chambers were made of high-density, round polyvinyl chloride pipe and consisted of two units: a base (r =12 cm, h =18 cm) and a detachable collar (r =12 cm, h =12 cm). On each sampling date, five chambers were installed at random locations ~ 5cm deep in the soil a day before taking samples to minimise the soil disturbance because of installation. The selected locations were carefully observed to avoid including crab burrows. The depth of bases was recorded from five points within each chamber to calculate the headspace volume. At the start of the experiment, gas chambers were closed, and a sample was taken, at time zero with a 20 ml tight syringe and transferred to a 12mL vacuumed exetainers (Exetainer, Labco Ltd., High Wycombe, UK). Samples were collected in the same manner over 1 hour following the sealing of the chamber. The gas samples were collected between 9:00 to 11:00 am, which is the optimum time for minimising the diurnal variation effect on daily GHG emissions (Reeves et al, 2016). Additionally, GHG sampling on this timing minimised the variability in cumulative seasonal fluxes for intermittent manual flux measurements (Wang et al., 2011; Deng et al, 2012; Reeves et al, 2016; Rashti et al, 2015).

A propagation soil thermometer (Gardman, 64704) was inserted at ~ 5cm depth outside of each chamber 15 minutes before the reading to record the soil temperature for each gas sampling (Kavehei et al, 2020). The GHG concentrations of all samples were analysed within two weeks of sampling with a gas chromatograph (Shimadzu GC-2010 Plus). For N_2O

analysis, an electron capture detector was used with helium as carrier gas while CH_4 concentration was analysed on flame ionisation detector with nitrogen as a carrier gas. For CO_2 determination, the gas chromatograph was equipped with a thermal conductivity detector. Peak areas of the samples were compared against standard curves for the determination of GHG concentrations (Chen at al, 2012). The obtained volumetric unit concentrations were converted to mass-based units using the Ideal Gas Law (Hutchinson and Mosier, 1981; Kavehei et al, 2021). Daily cumulative GHG fluxes were calculated by modifying the equation described by Shaaban et al. (2015) as following (Eq. 1);

$$\text{Seasonal cumulative GHG fluxes} = \sum_{i=1}^{n} (\text{Ri} \times 24 \times \text{Di} \times 17.381)$$

where; Ri = Gas emission rate (mg m^{-2} $hr^{-1}$for CO_2 and μg m^{-2} hr^{-1} for CH_4 and N_2O, Di = number of the sampling days in a season and 17.38=number of weeks in each season assuming three seasons prevailed over an annual cycle. Annual cumulative GHG fluxes were calculated by integrating seasonal cumulative GHG fluxes. Total cumulative GHG emissions reported in in our research represent CH_4 + N_2O fluxes. It is customary to exclude CO_2 in total GHG fluxes for C projects. For the first sampling period during the hot and dry season (21 - 29 October 2018), gas samples were collected at 0, 20, 40 and 60 minutes from all chambers to perform linearity test for measuring increase or decrease in the concentration of the gas with time. For subsequent experiments, linearity test was performed on subset chambers for each site (Rashti et al, 2016) and R^2 value of > 0.7 was recorded for all tested samples with a linear trend for CO_2, CH_4 and N_2O over the experimental period. For comparing GHG effects of CH_4 and N_2O fluxes, $CO_{2\text{-equivalent}}$ ($CO_{2\text{-eq}}$), the measurements in our study only represented soil fluxes, mainly respiration, because the chambers were incubated in the dark, thus do not represent a full budget that would include primary production.

2.4 Statistical analyses

GHG flux data were analysed for normality through Kolmogorov-Smirnov and Shapiro-Wilk tests. When data were not normal, they were transformed (log, 1/x) to comply with the assumptions of normality and homogeneity of variances. Despite transformations, some variables were not normally distributed; thus, the differences between sites and seasons were analysed with the non-parametric Kruskal-Wallis test and Mann-Whitney U Test. The data which met the normality assumptions were analysed for spatial and temporal differences with one-way Analyses of Variance (ANOVA), where site and season were the predictive factors and replicate (gas chamber) was the random factor of the model. Additionally, a Pearson correlation test was run to evaluate the correlation of GHG with measured environmental factors. The data were analysed using a statistical program, SPSS (v25, IBM, New York, USA) and values were presented as mean ± standard error.

3 Results

3.1 Soil physicochemical properties

Soil physical and chemical parameters varied significantly among the sites. Gravimetric moisture content was highest in the ponded pasture wet (55%) and lowest in the sugarcane (12%). Freshwater tidal wetlands and saltmarsh had similar moisture content (Table 2). All tested soils were acidic with mangroves having the highest pH value with 6±0.1, followed by saltmarsh with 5.8±0.1. The lowest electrical conductivity (EC) was recorded for the ponded pasture with 247 ± 38 and 190±39 µS cm^{-1} for the dry and wet pasture, respectively. For the top 20 cm soil, the natural wetlands had significantly higher EC (1418±104, 8049±276 and 8930±790 µS cm^{-1} for tidal freshwater wetland, saltmarsh and mangroves, respectively) compared to the agricultural land (190±39 µS cm^{-1}, 247±38 and 382±11 µS cm^{-1} for wet and dry ponded pasture and sugarcane, respectively).

The mean bulk density of the top 30 cm soil of the saltmarsh (1.4±0.1 g cm^{-3}), sugarcane (1.5±0.1 g cm^{-3}) and mangroves (1.9±0.1 g cm^{-3}) was similar, while the freshwater tidal wetland had the lowest value (0.6±0.1 g cm^{-3}). For all measured ecosystems, %C was highest in the top 10 cm of the soil and decreased below 10 cm (Table 2). The highest %C was recorded in the freshwater tidal wetland (5.1±0.6%) and lowest in the saltmarsh (1.2±0.1%). Soil %N ranged from 0.1±0.0 to 0.4±0.1% at all sites, except in the freshwater tidal wetland, where it reached values of 0.6±0.0% in the top 10 cm (Table 2).

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Table 2. Physicochemical characteristics for the soil of natural wetlands and agricultural land use types for the top 30 cm of soil in tropical Australia. Fw = Freshwater tidal forest, C= carbon, N = Nitrogen, EC = Electrical Conductivity. Values are mean ± standard error (5 replicates from each site)

Ecosystem	Depth (cm)	Gravimetric moisture content (%)		pH		EC (µs cm^{-1})		Bulk density (g cm^{-3})		C (%)		N (%)	
		Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Mangroves	0-10	41.7	1.1	5.9	0.1	12550	524	1.1	0.1	2.3	0.1	0.2	0.0
	10-20	34.6	0.7	5.9	0.3	12164	905	1.3	0.0	1.7	0.2	0.1	0.0
	20-30	31.3	0.6	6.2	0.1	5560	365	1.9	0.1	0.9	0.1	0.1	0.0
	Mean	35.9	1.2	6.0	0.1	8930	790	1.9	0.1	1.6	0.2	0.1	0.0
Saltmarsh	0-10	25.6	1.2	5.8	0.2	8442	435	1.1	0.0	1.4	0.1	0.1	0.0
	10-20	26.6	0.3	5.8	0.1	8666	437	1.5	0.1	1.3	0.1	0.1	0.0
	20-30	26.4	0.2	5.9	0.3	7040	316	1.6	0.0	1.0	0.3	0.1	0.0
	Mean	26.2	0.4	5.8	0.1	8049	276	1.4	0.1	1.2	0.1	0.1	0.0
Fw tidal forest	0-10	33.4	0.5	4.4	0.2	1099	17	0.5	0.1	7.8	0.1	0.6	0.0
	10-20	24.9	0.6	4.2	0.0	1272	164	0.7	0.0	5.4	0.0	0.5	0.0
	20-30	22.4	0.7	4.2	0.1	1882	47	0.8	0.0	2.2	0.1	0.1	0.0
	Mean	26.9	1.3	4.3	0.1	1418	104	0.6	0.1	5.1	0.6	0.4	0.1
Sugarcane	0-10	9.1	0.4	5.7	0.1	429	12	1.3	0.1	1.5	0.1	0.1	0.0
	10-20	12.1	0.6	5.3	0.3	365	11	1.5	0.1	1.5	0.1	0.1	0.0
	20-30	13.7	0.2	4.7	0.2	351	2	1.6	0.1	1.3	0.1	0.1	0.0
	Mean	11.7	0.6	5.2	0.2	382	11	1.5	0.1	1.4	0.1	0.1	0.0
Dry ponded pasture	0-10	12.4	0.3	4.1	0.0	378	21	0.8	0.1	3.1	0.3	0.3	0.0
	10-20	13.6	0.1	4.4	0.1	279	60	1.2	0.1	1.6	0.4	0.1	0.0
	20-30	14.5	0.7	4.4	0.3	84	4	1.3	0.2	1.6	0.2	0.1	0.0
	Mean	13.5	0.3	4.3	0.1	247	38	1.1	0.1	2.1	0.3	0.2	0.0
Wet ponded pasture	0-10	52.1	0.4	4.8	0.0	358	71	0.6	0.1	3.6	0.3	0.3	0.0
	10-20	47.7	0.4	4.9	0.1	117	11	1.3	0.0	1.7	0.1	0.1	0.0
	20-30	46.4	0.2	5.1	0.1	95	6	1.3	0.0	1.5	0.1	0.1	0.0
	Mean	48.7	0.7	4.9	0.0	190	39	1.1	0.1	2.3	0.3	0.2	0.0

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3.2 Greenhouse gas fluxes

Soil emissions for CO_2 were significantly different among sites and times of the year (t =155.09, n =237, $p < 0.001$; Fig. 2a). Highest CO_2 emissions were measured during the wet-hot period in the dry ponded pasture, where values reached 20.31 ± 1.95 g m^{-2} d^{-1} while the lowest values were measured in the saltmarsh, the only site where the soils were a sink of CO_2 with an uptake rate of -0.59 ± 0.15 g m^{-2} d^{-1} (Fig. 2a). In the ponded pastures, CO_2 emissions were higher when dry, with cumulative CO_2 emissions of 5,748 g±303 m^{-2} y^{-1} compared to wet ponded pastures with 2,163±465 g m^{-2} y^{-1}. For wetlands, cumulative annual CO_2 emissions were highest in freshwater tidal forests with 2,213±284 g m^{-2} y^{-1}, followed by mangroves with 1,493±111 g m^{-2} y^{-1} and lowest at the saltmarsh with uptake rates of -264±29 g m^{-2} y^{-1}. During high and low tide, emissions of CO_2 in the hot-dry season were similar for mangroves ($F_{1,28}$ = 2.911, p = 0.099) and freshwater tidal forest ($F_{1,28}$ = 0.563, p = 0.459; Table 3). However, saltmarsh had significantly different lower CO_2 fluxes during the high tide ($F_{1,28}$ = 20.060, $p < 0.001$) with -1.12 ± 0.24 g m^{-2} d^{-1} compared to 0.69 ± 0.4 g m^{-2} d^{-1} during the low tide (Table 3).

For CH_4 fluxes, significant differences were observed among sites and seasons (t = 182.33, n =237, $p < 0.001$).The differences between different sites were substantial, with wet ponded pasture having significantly higher ($p < 0.001$) CH_4 emissions than any site, with rates ~200 times higher throughout the measured period (Fig. 2b). For tidal coastal wetlands, emissions of CH_4 were highest during the wet-hot season in all the sites except for the mangroves, which had similar emissions throughout the year (Fig. 2b). Overall, cumulative annual CH_4 emissions were 209±36 g m^{-2} $y^{-1}$for the wet ponded pasture followed by mangroves (0.73±0.13 g m^{-2} y^{-1}), dry ponded pasture (0.15±0.03 g m^{-2} y^{-1}), freshwater tidal forest (0.14±0.03 g m^{-2} y^{-1}), saltmarsh (0.04±0.01 g m^{-2} y^{-1}) and sugarcane (-0.04±0.02 g m^{-2} y^{-1}). For tidal coastal wetlands, CH_4 emissions did not differ significantly among mangroves ($F_{1,28}$ = 1.539, p = 0.225), saltmarsh ($F_{1,28}$ = 0.007, p =0.934), and freshwater tidal forest ($F_{1,28}$ = 2.052, p =0.163) between low and high tide (Table 3).

For N_2O fluxes, highest emissions (55 ± 9 mg m^{-2} d^{-1}) were measured in the dry ponded pasture in the hot-wet season followed by sugarcane (20 ± 3 mg m^{-2} d^{-1}) during the hot-dry period which coincides with the post-fertilisation period (Fig. 2c). Overall, dry ponded pastures had the highest cumulative annual N_2O emissions (7.99±2.26 mg m^{-2} d^{-1}), followed by sugarcane (2.37±0.68 mg m^{-2} d^{-1}), wet ponded pasture (1.32±0.33 mg m^{-2} d^{-1}), saltmarsh (0.33±0.11 mg m^{-2} d^{-1}), freshwater tidal forests (0.04±0.0 mg m^{-2} d^{-1}) and finally, mangroves (0.02±0.04 mg m^{-2} d^{-1}). However, these differences were only statistically significant when considering the interaction between time of the year and site (t =100.21, n =237, p < 0.001). For tidal coastal wetlands, N_2O fluxes for the low versus high tide were significantly different for mangroves ($F_{1,28}$ = 38.283, $p < 0.001$) with -0.74 ± 0.17 mg m^{-2} $d^{-1}$of N_2O uptake during high tide compared to 0.15 ± 0.06 mg m^{-2} d^{-1} release during low tide (Table 3). Saltmarsh showed the opposite trend with significantly higher N_2O uptake ($F_{1,28}$ = 38.313, p <

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0.001) during low tide (Table 3). Like saltmarsh, the freshwater tidal forest had significantly higher N_2O uptake during low compared to high tide ($F_{1,28}$ = 13.529, p =0.001; Table 3)

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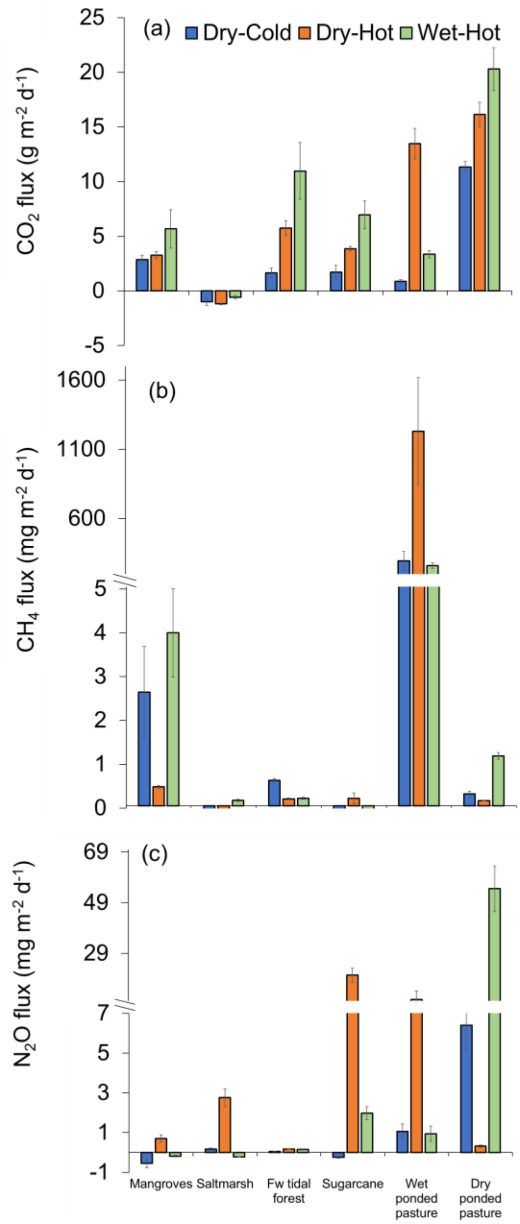
265 

Figure 2: Greenhouse gas fluxes of (a) CO_2 (g m^{-2} d^{-1}), (b) CH_4 (mg m^{-2} d^{-1}) and (c) N_2O (mg m^{-2} d^{-1}) from soils of tropical coastal wetlands (mangroves, saltmarsh, freshwater (Fw) tidal forest and their alternative land uses (sugarcane and ponded pastures) during three periods of the year: dry-cold, dry-hot and wet-hot

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Table 3. Greenhouse gas (GHG) fluxes from soils of tropical coastal wetlands: mangroves, saltmarsh, and freshwater (Fw) tidal forest during high and low tide during a dry-hot season

GHG	Land-use type	High tide Mean	SE	Low tide Mean	SE
CO_2 (g m^{-2} d^{-1})	Mangroves	2.55	0.37	3.25	0.57
	Saltmarsh	-1.12	0.24	0.69	0.40
	FW tidal forest	2.97	1.35	5.35	2.68
CH_4 (mg m^{-2} d^{-1})	Mangroves	3.38	0.98	236	73
	Saltmarsh	-0.13	0.06	-25	6
	Fw tidal forest	1.10	0.52	457	108
N_2O (mg m^{-2} d^{-1})	Mangroves	-0.74	0.17	0.15	0.06
	Saltmarsh	0.19	0.06	-0.14	0.04
	FW tidal forest	0.06	0.01	-0.25	0.16

3.3 Greenhouse gas emissions and environmental factors

Overall, we found not one single parameter could explain GHG emissions from all sites except land-use. The CO_2 emissions were not significantly correlated to bulk density (R^2 =0.026 p = 0.918 n=18), % WFPS (R^2=-0.003 p = 0.99 n =18), or soil temperature (R^2=0.296 p = 0.233, n =18). Similarly, soil CH_4 emissions were not correlated with bulk density (R^2 = -0.096 p = 0.706 n =18), % WFPS (R^2 = 0-.224 p = 0.372, n =18) or soil temperature (R^2 =0.286 p = 0.25 n=18). Finally, no correlation was found between N_2O emissions and bulk density (R^2= -0.349 p = 0.156 n =18), % WFPS (R^2= -0.34 p = 0.168 n =18), or soil temperature (R^2= -0.241 p = 0.335 n =18).

4 Discussion

Across the agricultural land use types in our study area, very high GHG emissions were measured. The highest emitter was by far the wet ponded pastures, with CH_4 emissions that were 200 times higher than any other measured land uses. The dry pasture was also a high emitter of CO_2 at 20.31 ± 1.95 g m^{-2} d^{-1} and of N_2O at 55 ± 9 mg m^{-2} d^{-1}, especially during hot periods. Natural wetlands had significantly lower CH_4 emissions, with saltmarshes having the lowest. Overall, natural wetlands had very low total cumulative GHG emissions at 1,263 $CO_{2\text{-eq}}$ kg ha^{-1} yr^{-1} compared to their alternate agricultural land uses, which emitted 87,156 $CO_{2\text{-eq}}$ kg ha^{-1} yr^{-1}. These results confirm our hypothesis that coastal tidal wetlands, even freshwater ones, can be a viable land-use to reduce GHG emissions from current agricultural land.

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The GHG emissions varied with season, with an overall increase in emissions during the hottest and wettest time of the year. The emissions of CO_2 and N_2O were highest when temperatures were > 38°C. Similar results have been shown in terrestrial forests, where N_2O emissions increased with temperature, explaining 86% of the flux variations (Schindlbacher et al, 2004). Emissions of CO_2 were associated with temperature, although the correlation was not significant, meaning that differences in temperature within season would affect emissions. Highest CO_2 emissions from all land-use types during the early wet season could be attributed to "Birch effect" which refers to short term but a substantial increase of respiration from soils under the effect of precipitation during early wet season (Fernandez-Bou et al, 2020). These findings are in accordance with recent studies which reported that CO_2 fluxes from subtropical mangroves were largest when the temperature was highest, and in periods of the year with reduced salinity (Liu and Lai 2019). The soil from mangroves within our study sites were always sources of CO_2, partly because the incubation was done in dark chambers

Emissions of CH_4 were not significantly associated with the environmental factors measured in this study. However, CH_4 emissions were highest during the hot-dry season. Increased CH_4 emissions with temperature could be attributed to the increase of the activity of methane-producing microbes in the soil (Ding et al, 2004). Additionally, high temperatures increase plant growth, providing the substrate for CH_4 production by root exudates (Yvon-Durocher et al, 2011). The relatively low CH_4 emissions from all the natural wetlands could be attributed to the presence of terminal electron acceptors like iron, sulphate, manganese and nitrate which result in low rates of methanogenesis (Fumoto et al, 2008; Kögel-Knabner et al, 2010; Sahrawat, 2004). For example, sulphate reducing bacteria outcompete methane-producing bacteria (methanogens) in the presence of high sulphate concentrations in tidal wetlands, resulting in low CH_4 production. Additionally, competition between methanogens and methanotrophs (CH_4 consuming bacteria), could result in a net balance of low CH_4 production despite freshwater conditions (Maietta et al. 2020).

We found high variability in CH_4 emissions within land-use types that were very close to each other (Fig. 1b). These differences might be attributed to the microbial community composition and abundance, which can change rapidly over small spatial scales (Martiny et al, 2006; Drenovskyet al, 2010). In general, emissions of CH_4 were best explained by land-use type. Our study is consistent with the recent findings, which reported very high variations in GHG fluxes from mature, converted, and restored land use types for mangroves (Cameron et al, 2020). Our findings also corroborate studies on GHG emissions in a Mediterranean climate, where drained agricultural land use types, pasture and corn were the largest CO_2 emitters compared to restored wetlands (Knox et al. 2015). Our findings are also in agreement with other studies which reported drained, and degraded peatlands were large net sources of CO_2 (Nieveen et al, 2005; Veenendaal et al, 2007; Hirano et al, 2012). Overall, our study provides evidence that despite GHG emissions from wetlands, these are lower than for some agricultural land uses. Additionally, some of the wetland types, such as marshes were consistently found to be sinks of CO_2 and CH_4. This has been previously reported, with studies finding that some intertidal wetlands can be sinks of GHG at least under some conditions or during some times of the year (Knox et al, 2015; Maher et al, 2016)

The GHG emissions from wetlands have an extensive range. For CO_2 fluxes, they can range between −139 and 22,000 mg m^{-2} d^{-1} (Stadmark and Leonardson 2005; Morse et al. 2012), for CH_4, from −1 to 418 mg m^{-2} d^{-1} (Allen et al. 2007; Mitsch et al 2013; Cabezas et al. 2018), and for N_2O, from −0.3 to 3.9 mg m^{-2} d^{-1} (Hernandez and Mitsch 2006; Morse et al. 2012). The GHG fluxes measured in this study are within the lower end, with ranges from -1191 to 10, 970 mg m^{-2} d^{-1} for CO_2, from −0.2 to 3.9 mg m^{-2} d^{-1} for CH_4, and −0.2 to 2.8 mg m^{-2} d^{-1} for N_2O.

4.1 Management implications

Under the Paris Agreement, Australia has committed to reducing GHG emissions 26 - 28% below its 2005 levels by 2030. Any GHG mitigation strategy should be based on robust GHG flux quantification from different land-use scenarios. With annual emissions of 153.0 million tonnes of carbon dioxide equivalent (Mt $CO_{2\text{-eq}}$ y^{-1}), Queensland is a major GHG emitter in Australia (~ 28.7% of the total in 2016; www.stateoftheenvironment.des.qld.gov.au). Of these emissions, about 18.3 Mt $CO_{2\text{-eq}}$ y^{-1} (14%) are attributed to agriculture, while land-use change, and forestry emitting another 12.1 Mt $CO_{2\text{-eq}}$ y^{-1} (www.stateoftheenvironment.des.qld.gov.au). Production of CH_4 from ruminant animals, mostly cattle, contributed 82% of agriculture-related emissions (www.daf.qld.gov.au).

This study provides evidence for three management actions to reduce GHG emissions. The conversion of agricultural lands both ponded pastures and sugarcane to intertidal wetlands including mangroves, marshes and freshwater tidal wetlands seems to be a promising land-use change management option for GHG mitigation. Our results showed that wet and dry ponded pastures emitted 56 $CO_{2\text{-eq}}$ ton ha^{-1} $y^{-1}$and 24 $CO_{2\text{-eq}}$ ton ha^{-1} $y^{-1}$of total GHG (CH_4 + N_2O) compared with 0.2 $CO_{2\text{-eq}}$ ton ha^{-1} y^{-1}, 0.1 $CO_{2\text{-eq}}$ ton ha^{-1} y^{-1} and 0.9 $CO_{2\text{-eq}}$ ton ha^{-1} y^{-1} from mangroves, freshwater tidal forest and saltmarshes respectively. This implies that up to 56 $CO_{2\text{-eq}}$ ton ha^{-1} y^{-1} emissions could be potentially avoided by restoring wet ponded pastures to coastal wetlands. Furthermore, financial incentives through the inclusion of restored agricultural lands in C markets could be an alternative income source for farmers. For instance, our results suggested that wet pasture landowners could get ~AUD $ 860 $ha^{-1}yr^{-1}$ assuming carbon value of AUD $15.37 per ton of $CO_{2\text{-eq}}$ (Australian Emission Reduction Fund, December 2018). Recent studies suggested that legal enablers are in place for the conversion of ponded pastures to tidal wetlands (Bell-James and Lovelock 2019) supporting our recommendation for inclusion of these ecosystems on carbon mitigation programs. However, actual GHG mitigation gains could be even greater, if accounted for the carbon sequestration in the vegetation and soil after wetland restoration.

Another management option would be to reducing the area of wet pastures to minimise CH_4 emissions. For example, our study showed that dry pastures produced significantly less CH_4 ~0.005 kg ha^{-1} d^{-1} than wet pastures 6 kg $ha^{-1}d^{-1}$ Assuming the average cow produces 141 g CH_4 per day (McGinn et al, 2004) and our study area supported around 900/2500 ha cattle throughout the year, cattle would be responsible for only 0.05 kg $ha^{-1}d^{-1}$of CH_4 emissions. This means that nearly 99% of the CH_4 emissions came from the wet pastures while dry pasture and grazing cattle had a low share in total CH_4

emissions. Therefore, land use management of wet pastures which are used to feed grazing cattle in Queensland may be a significant opportunity to reduce agriculture related CH_4 emissions.

Finally, fertiliser management in sugarcane could reduce N_2O emissions. Higher N_2O emissions of 17.63 mg m^{-2} d^{-1} were measured in sugarcane following fertilisation, during the dry-hot season. Comparatively, natural wetlands had low N_2O emissions (0.16 to 2.79 mg m^{-2} d^{-1}), even with the saltmarsh being an occasional sink. Thus, improved management of fertiliser applications could result in GHG emission mitigation through reduced N_2O emissions. Some activities include split application of nitrogen fertiliser in combination with low irrigation, reduction in fertiliser application rates, substitution of nitrate-based fertiliser for urea (Rashti et al, 2015), removing mulch layer before fertiliser application (Pinheiro et al, 2019; Xu et al, 2019) or conversion of sugarcane to wetlands.

5 Conclusion

To our best knowledge, this is the first study to report GHG emissions mitigation gains from a wide range of tropical natural wetlands including mangroves, saltmarshes and freshwater tidal wetlands and their alternative agricultural land use types. The significantly higher emissions from agricultural land use types as compared to tropical coastal wetlands were most likely due to land-use as any other physical or chemical factors could not explain the differences observed. Restoration of wet ponded pastures and sugarcane to coastal tidal wetlands, even freshwater tidal forests, has the potential to mitigate total GHG emissions (CH_4+ N_2O) derived from agricultural activities. The GHG emissions from ponded pastures were particularly alarming, with values 200-fold than any other land use. However, these massive emissions provide an opportunity to target ponded pastures for consideration in GHG mitigation programs, which could deliver financial incentives for farmers, and many co-benefits derived from coastal wetland restoration. Overall, the data from this study will contribute significantly to our understanding of land use effects on GHG budgets in Australia, and in similar tropical regions around the world.

Author contribution

Iram, N. and M.F. Adame designed the project, Iram, N, B. Shahrabi Farahani and E. Kavehei carried out experiments, Iram, N., E. Kavehei and M.F. Adame analysed the data. Iram, N prepared the manuscript with contributions from D.T. Maher, S.E., Bunn, M. Rezaei Rashti and M.F. Adame.

Competing interests

The authors declare that they have no conflict of interest.

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