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Groundwater driven carbon fluxes in a restored coastal saltmarsh wetland: Implications for coastal wetland restoration

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ABSTRACT

Coastal wetlands play a crucial role in the global carbon cycle, yet they have been extensively degraded over the past decade. Though restoration efforts are underway worldwide, there is limited understanding of the role groundwater plays in transporting dissolved carbon within restored wetlands. Here, we address this knowledge gap by investigating water and carbon fluxes in the restored Tomago Wetlands in coastal NSW, Australia. We aim to 1) quantify surface water exports of the main carbon components, 2) estimate greenhouse gas emissions from the aquatic parts of the wetland, and 3) determine the contribution of groundwater discharge to the surface water carbon export and greenhouse gas emissions from the wetland. Sampling took place following a wet period and a freshening event. A radon mass balance model estimated groundwater discharge contributing 10 % to the surface water flow in the restored wetland. The wetland was a source of four of the five main carbon components including DOC, DIC, alkalinity, and CO₂, with most of the exported carbon being in the form of alkalinity. Surface water DOC, DIC, alkalinity, and CO₂ exports were, 1.3 ± 0.5 , 35.5 ± 13.1 , 39.4 ± 14.6 and 6.5 ± 2.1 mmol/m²/ d, respectively. The average water to atmosphere flux of CO₂ and CH₄ were $104 \pm 209 \text{ (mmol/m}^2/\text{d)}$ and $21 \pm 100 \text{ (mmol/m}^2/\text{d)}$ 42 (μ mol/m²/d). Groundwater-driven carbon fluxes contributed 100 % of the surface water DOC, 30 % of the DIC, 15 % of the alkalinity exports, and 5 % of the overall CO_2 losses (lateral aqueous export + atmospheric emissions) from the wetland, with minor contributions to CH4. Carbon exports from both the wetland and groundwater observed in this restored wetland were found to be lower than those reported in the literature for natural or mature wetlands. This would have important implications when developing carbon accounting methods. Overall, our findings highlight the importance of groundwater in carbon transport within restored wetlands and emphasise the need for inclusion of shallow groundwater dynamics in carbon budgets and inventories of coastal wetlands that have or will undergo restoration.

1. Introduction

Coastal wetlands are a significant component of the global carbon cycle but have been degraded by extensive drainage for agriculture and development (Ouyang and Lee, 2020). Globally these modifications have resulted in a 50 % loss of tidal wetlands since 1900 (Davidson, 2014). In addition to the ecosystem destruction, the increased drainage has released stored carbon to the atmosphere and accelerated global warming (Pendleton et al., 2012). Predictions warn that under the current loss rate, a further 35 % of tidal wetlands will be lost by the year 2100, with a global economic impact of \$2.6 billion USD per year due to the additional carbon released to the atmosphere (Gulliver et al., 2020;

Pendleton et al., 2012). In response, wetland restoration has become a management priority worldwide, with the United Nations declaring 2021–2030 the "UN Decade on Ecosystem Restoration".

Australian coastal wetlands are amongst the world's largest stores of carbon, containing up to 11 % of the global coastal vegetation carbon stocks (Serrano et al., 2019). As such, the Australian government has recognised their potential role in the carbon cycle and the importance of preserving and restoring coastal wetlands. This has led to plans to include natural and restored coastal wetlands in the national greenhouse gas inventory (Macreadie et al., 2017). However, effective wetland restoration for carbon storage requires a quantitative understanding of carbon sinks, sources and transport processes within restored sites

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(Lovelock et al., 2023). Currently, limited information is available on the sources, sinks and transport processes of subsurface carbon within restored wetlands. For example, there is currently no comprehensive quantitative study investigating the role of groundwater as a pathway for carbon transport in restored coastal wetlands. As such, the carbon budget in these ecosystems is mostly conceptualised and has not been accurately quantified using field data.

Previous research related to carbon cycling in restored wetlands has primarily focused on estimating carbon burial rates and carbon dioxide (CO₂) emissions (Gulliver et al., 2020; Negandhi et al., 2019). In contrast, hydrological (surface water + groundwater) exports of carbon are typically unaccounted for, due to the often patchy and spatially heterogeneous nature of groundwater flow that makes quantifying groundwater flows challenging (Bogard et al., 2020; Sadat-Noori et al., 2016). Despite these challenges groundwater flow is known to have a role in carbon cycling of mangroves and saltmarshes ecosystems warranting its quantification in coastal carbon budgets (Alongi, 2020a; 2020b; Sadat-Noori and Glamore, 2019).

Natural geochemical groundwater tracers, such as Radon (²²²Rn), have been used to overcome some of the challenges associated with quantifying groundwater fluxes in estuaries and wetlands (Sadat-Noori et al., 2021a). This is especially relevant in conditions where insufficient hydrological data is available to develop groundwater flow models with the necessary degree of certainty (Burnett et al., 2006). Radon has the advantages of being chemically unreactive, having a short half-life of 3.8 days, and higher concentrations in groundwater compared to surface water. These characteristics make it well suited for quantifying groundwater discharge rates into surface waterbodies (Burnett and Dulaiova, 2003), which, when carbon concentrations are measured, can be used to quantify dissolved carbon export from wetlands.

While tidal restoration practices are increasing worldwide (1 million km² of degraded land has been restored up to 2020) (Sadat-Noori et al., 2021b; Waltham et al., 2020), there is very limited research on groundwater-derived fluxes of carbon and associated evasion of greenhouse gases post restoration. Quantifying these subsurface processes will ensure that restored coastal wetlands can effectively be included in carbon budgets and inventories. Furthermore, these hydrological processes are potentially more important when drained former wetlands are restored to tidal systems. This is because groundwater discharge, driven by tidal pumping, can result in increased hydrological carbon export. Such exports can be a major component of coastal wetland carbon budgets (Bogard et al., 2020; Kroeger et al., 2017).

The aim of this study is to quantify the contribution of groundwater discharge to the hydrological export of the five main carbon components and greenhouse emissions from a restored coastal wetland. We hypothesise that groundwater discharge plays an important, yet overlooked role, in carbon and greenhouse gases dynamics of restored coastal wetlands. We test this hypothesis by conducting time series measurements of the main carbon components (alkalinity, DIC, DOC, CO₂, and CH₄) and radon in surface water and groundwater of a restored coastal wetland. We quantify the surface water carbon export from the wetland, estimate atmospheric CO_2 and CH_4 emissions, calculate groundwater-derived inputs of DIC, DOC, CO₂, and CH₄ and alkalinity to the wetland, and determine the relative significance of groundwater discharge to the total dissolved carbon export from the wetland. We contribute significant knowledge to the literature of this research area by, for the first time, quantifying groundwater contributions of the five main carbon components, and their importance in carbon export and atmospheric CO2 and CH4 emissions from a coastal wetland. Furthermore, we report these fluxes from a restored wetland which has rarely been done to date.

2. Methodology

A field campaign was carried out in 2022 from August 15th to 19th (both days inclusive) at the (downstream) mouth of the 'North-South'

drain, which constitutes the Stage 1 restoration works of Tomago Wetlands, within Hunter Wetlands National Park, NSW, Australia (-32.841997, 151.754921 – Fig. 1). The site is bounded by a perimeter levee and serviced by an internal drainage system, both of which were originally constructed to facilitate land reclamation efforts in the 1920s. The original levee system underwent further development by the NSW Public Works Department between 1968 and 1980. Nevertheless, the site supports numerous endangered and vulnerable species and provides foraging and roosting opportunities for migratory shorebirds (Glamore et al., 2021). Consequently, the site was listed as a Ramsar wetland of international importance in 1982 and a major saltmarsh restoration project commenced in 2007 (Rayner and Glamore, 2010).

The low-lying area has a catchment size of around 400 ha with limited upstream freshwater inputs. The site received no rainfall during the sampling period, however, 27.2 mm of rain was recorded at Raymond Terrace (a local weather station operated by the Bureau of Meteorology (BOM), located approximately 2 km north of the study site) in the 2 weeks prior to the sampling campaign. The wider catchment received 314.2 mm of rainfall in the 1.5 months prior to the sampling campaign (July 2022), causing freshening of the main estuary (htt ps://www.bom.gov.au). The area had an average temperature of 18.3° C for the month of August 2022 (Australian winter). The creek draining the wetland is 1.45 km long and approximately 10 m wide.

The experimental approach of this study comprised the following steps:

1) hourly grab samples of surface water samples from the creek at the



Fig. 1. A) map of the Tomago wetland boundary and its location in the Hunter River estuary. B) detailed location of field campaign and long-term sampling points. Groundwater piezometer locations are shown with red circles (numbers refer to Table 1). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

mouth of the wetland were analysed for dissolved organic and inorganic carbon over a 24-hour time period to cover a full diel cycle (two tidal cycles);

2) groundwater samples taken from 10 shallow piezometers within the wetland were analysed for the same parameters to characterise the composition and variability of the groundwater endmember (for location see Fig. 1); and,

3) continuous high-frequency (15 min) instream measurements of a range of water quality parameters and dissolved greenhouse gases in the creek at the mouth of the wetland.

2.1. Surface water sampling

Surface water grab-samples were collected using an ISCO portable autosampler. Sub-samples were taken with polypropylene syringes, filtered through 0.45µm cellulose acetate filters and collected into 60 ml falcon tubes for dissolved organic carbon analysis. Samples were collected, preserved, and stored following standard sampling methods. Surface water physico-chemical parameters were measured in-situ at 15min intervals using a calibrated YSI EXO₂ water quality multiparameter sonde permanently installed in the channel. Measured variables including pH (\pm 0.1 units), conductivity (\pm 0.001 mS/cm), salinity (\pm 0.1), dissolved oxygen (DO) (\pm 0.1 mg/L), water temperature (\pm 0.01 °C) and depth (\pm 0.004 m). Fluorescent dissolved organic matter (fDOM) concentrations were also measured using the same sonde and used as a proxy for dissolved organic matter in the wetland water column. Current velocity and direction were measured in the creek using a Sontek-IQ PLUS current profiler at 15 min intervals (\pm 0.5 cm/s).

Radon and dissolved greenhouse gases were measured using a RAD7 device and a LI-COR LI-7810 trace gas analyser using the constant flow and showerhead Gas Equilibrium Device (GED) system. In brief, a bilge pump was used to deliver a constant, 2.5 L/min stream of water to the GED, from approximately 50 cm below the surface. The gases (i.e., radon, CO₂ and CH₄) in the water then equilibrated with the air inside the GED. The equilibrated air was then pumped through a desiccant (Drierite) to absorb excess moisture and then into the RAD7 and LI-COR instruments. Finally, the air is returned to the GED to complete the loop. Equilibrium times of 5, 20 and 30 min are required for CH₄, CO₂ and radon, respectively (Sadat-Noori et al., 2021c; Santos et al., 2012b). CH₄ fugacity was converted to concentrations based on solubility coefficient calculated as a function of water temperature and salinity (Wiesenburg and Guinasso Jr, 1979).

2.2. Groundwater sampling

A total of 10 groundwater samples were collected at different depths from the intertidal zone during the field campaign. Piezometers (50 mm ID PVC) were installed from 1 to 4 m depth using a hand auger. A peristaltic pump was used to extract samples after purging the bores until readings of DO, pH and EC stabilised. The groundwater sampling procedure for dissolved constituents after extracting water from the bores was similar to that adopted for sampling surface water. Groundwater physico-chemical characteristics were measured using a handheld water quality probe (HACH, Germany) for electrical conductivity (± 0.005 mS/cm), pH (± 0.02), dissolved oxygen (± 0.1 for < 8 and ± 0.2 for 8 mg/L), and temperature (± 0.01 °C).

2.3. Laboratory analysis

Alkalinity was determined using the Gran Titration method (Stumm and Morgan, 1981) on a 25 mL filtered subsample using a HACH digital titrator. Samples were stored in a dark and cool environment and titrated on the same day of collection. A DOC-LABOR Liquid Chromatography-Organic Carbon Detection size exclusion chromatography system (LC-OCD) with the customised software program Chrom-CALC (from DOC-LABOR, Karlsruhe, Germany), was employed to analyse total dissolved organic carbon in addition to analysing six distinct size fractions based on mass (Huber et al., 2011). Sulfate was analysed using Inductively Coupled Plasma Mass Samples (ICP-MS). All samples were analysed at the Solid State & Elemental Analysis Unit at Mark Wainwright Analytical Centre, The University of New South Wales (UNSW Sydney).

2.4. Radon diffusion and ingrowth from parent isotope

To measure radon diffusion from sediments, three sediment samples were collected from the hourly grab sampling station. In the laboratory, 200 g of sediment and 500 mL of radium free water were placed in containers and sealed until sediment–water equilibrium was reached (~1 month). This method relies on the radioactive properties of radon and assumes that after approximately six radon half-lives, which is approximately 23 days, the radon emission will achieve secular equilibrium (i.e., steady state), with the only process leading to its reduction being radioactive decay. The container was regularly shaken to ensure mixing. The equilibrated water was then transferred to a 250 ml bottle and tested for radon using the RAD7-H₂O accessory (Corbett et al., 1998).

To evaluate the contribution of radon due to the decay of its parent isotope, radium-226, two containers were filled with water from the site. The water was then processed through columns containing 15–20 g of manganese oxide fiber, known for its ability to absorb radium-226 (Moore, 2003). The fibers were then securely sealed and after six half-lives, the radon produced as a result of radium-226 decay were measured using a RAD7 device and following the methodology described by Kim et al. (2001).

2.5. Radon mass balance

A steady-state radon mass balance model was developed to calculate the groundwater discharge into the wetland. This model, extensively detailed by Burnett and Dimova (2012), takes into account various sources and sinks of radon within the system. The underlying assumption is that any unaccounted-for radon serves to balance the model and is attributed to groundwater discharge. The model considers radon sources from groundwater discharge ($Q_{gw}Rn_{gw}$), diffusion from sediments ($D_{dif}A$), and ingrowth from parent isotopes ($^{226}Ra\lambda_{222}V$), and accounts for radon losses due to radioactive decay, and atmospheric evasion. These account for all known sources and sinks of radon in the environment. The mass balance model is expressed as follows:

$$(Q_{gw}Rn_{gw}) + (D_{dif}A) + (^{226}Ra\lambda_{222}V) = (Rn_{sw}\lambda_{222}V) + (Q_{out}Rn_{out}) + J_{atm}A$$
(1)

where Q_{gw} represents groundwater discharge (m³/d), Rn_{gw} is the radon concentration in groundwater (Bq/m³), D_{dif} is radon diffusion from sediments (Bq/m²/d), (²²⁶*Ra*) is radium concentration in surface water (Bq/m³), λ_{222} represents the radon decay constant (0.181/day), *V* is the water volume in the wetland (m³), Rn_{sw} is the radon concentration in surface water (Bq/m³), $Rn_{sw}\lambda_{222}V$ is radon decay (Bq/d), Q_{out} is the water flow leaving the system (m³/s), Rn_{out} is radon concentration (Bq/ m³), J_{atm} is radon atmospheric evasions (Bq/d), and *A* is the wetland inundation surface area (m²).

Radon evasion losses were calculated using an empirical equation, typically applied in tidal environments, that relates the concentration gradients of radon between air and water, solubility (which varies with temperature and salinity), and gas transfer velocity (Borges et al., 2004a; MacIntyre et al., 1995).

$$J_{atm} = K(Rn_{Water} - \alpha Rn_{air})$$
⁽²⁾

where Rn_{Water} is the radon concentration in water, and Rn_{air} is the radon concentration in air (Bq/m³). The symbol α represents the Ostwald

solubility coefficient, a dimensionless factor that characterizes the distribution of radon between the fluid and gas phases at equilibrium. The variable K is the gas transfer velocity at the air-water boundary, measured in meters per day (m/d). In environments influenced by tides, radon evasion is influenced mainly by wind, currents, and water depth. As a result, K driven by wind, current, and depth, was estimated using equations sourced from Borges et al. (2004a) and as described in detail in previous studies (Sadat-Noori et al., 2016). The water surface area of the wetland was used to calculate the radon wind evasion whilst the area of the main drainage creek in the wetland was used to calculate radon current evasion as currents are close to zero in the low-lying areas of the wetland (Sadat-Noori and Glamore, 2019; Santos et al., 2015). Following the fundamental principles of error propagation, the uncertainty associated with each parameter was propagated into the radon mass balance model to calculate the overall uncertainty associated with groundwater discharge.

2.6. Water - air interface gas flux calculations

The gas exchange rate (CO_2 and CH_4 flux) is controlled by the concentration gradient between water and air. These fluxes were calculated utilising the equation below (Wanninkhof, 1992):

$$F = K\alpha(C_{Water} - C_{air}) \tag{2}$$

where *F* represents the flux of CO₂ or CH₄ and is measured in units of mmol/m²/day. C_{Water} indicates the partial pressure of CO₂ or CH₄ in water, whereas C_{air} denotes the partial pressure of CO₂ or CH₄ in the air, both expressed in units of µatm. The background or atmospheric concentrations were assumed to be 420 µatm for CO₂, and 2 nM for CH₄, respectively. The solubility coefficient, denoted as *a*, is computed as a function of salinity and temperature using the constants established by Weiss (1974) for CO₂ and Wiesenburg and Guinasso Jr (1979) for CH₄.

To explore the possible range of flux rates, the gas transfer velocity at the water–air interface, *K*, was calculated using various methods available in the literature. Six different authors defined *K* as a function of wind speed for wetland/estuary ecosystems (Cole et al., 2010; Cole and Caraco, 1998; Crusius and Wanninkhof, 2003; MacIntyre et al., 1995; McGillis et al., 2001; Wanninkhof, 1992). In each of these methods, the gas transfer velocity equation relies on wind speed (u) measured at a height of 10 m (m/s) and the gas-specific Schmidt number (Sc), which is influenced by salinity and temperature (Wanninkhof, 1992). The calculation of gas fluxes at the water–air interface was done using average daily wind speeds sourced from the local weather station at Raymond Terrace.

2.7. Groundwater discharge and carbon export calculations

Groundwater fluxes were determined by multiplying the groundwater discharge rate, obtained from the radon mass balance model, by the average concentration of dissolved carbon species found in groundwater. For wetland solute export (during ebb tide) and import (during flood tide), hourly water flow rates were multiplied by the timespecific solute concentrations in the water column. Subsequently, export and import rates were integrated over a diel cycle, to yield daily net export or import rates. The uncertainties associated with the radon mass balance and carbon exports were calculated using error propagation principles (Sadat-Noori et al., 2015).

3. Results

3.1. Surface water time series observations

The area received a total of 27.4 mm of rainfall two weeks prior of the sampling event (i.e., in the first two weeks of August). Additionally, 314.2 mm of rainfall fell in the 1.5 months prior to the sampling

campaign (i.e., in July); significantly higher than the average monthly rainfall (72.8 mm) for the month of July (www.BOM.com; Raymond Terrace Station). These wet conditions caused high river flow in the Hunter River and fresher than usual surface water conditions for this coastal wetland (Fig. 2).

The tidal range at the site was 0.6 m and water depth in the creek varied between 0.6 and 1.2 m during the observed semi-diurnal tides (Fig. 3A). The tidal exchange significantly influenced most surface water physico-chemical parameters. Water temperature varied between 10 to 15 °C, peaking during daytime. Average dissolved oxygen was 8 mg/L with lower concentrations observed during night-time. Additionally, dissolved oxygen peaked at high tides. Radon concentrations ranged between 12 to 330 Bq/m³ with an average of 135 ± 56 Bq/m³. Higher concentrations were observed at low tide, indicating that groundwater



Fig. 2. Long-term timeseries of water level, flow, and salinity at the wetland main creek (see location in Fig. 1). The blue shaded area shows the period of the detailed sampling event and highlights that the detailed sampling took place during a time of freshening of the wetland estuary. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)



Time from 7:00 am, 17/08/2022

Fig. 3. Timeseries plot of continuous measurement of physical and chemical water quality parameters (Panel A), radon (groundwater tracer) and dissolved greenhouse gases (panel B) and discrete grab-samples of organic and inorganic carbon (Panel C). The vertical pink bars in Panels A, B and C indicate times of low tide when groundwater discharge was likely. The yellow dashed bar indicates the sampling time shown in Panel C. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

discharge occurring during low tide.

The average wetland surface water EC was 4,393 \pm 1504 (SD) μ S/ cm. This EC is significantly lower than oceanic EC, which are typically around 54,000 μ S/cm. Surprisingly, two peaks were observed for EC indicating complex surface water flow dynamics occurring over the site. One peak occurred at high tide following oceanic saline water inputs (which is expected), while the other peak in EC occurred at low tide. The second EC peak indicates an alternative source of salinity from upstream of the wetland (Fig. 3B) and/or potentially from saline wetland groundwater discharging during low tide. The observed high EC concentrations in GW and the radon (groundwater tracer) data support the latter assumption (Table 1). Surface water sulfate concentrations ranged from 114 to 398 mg/L with an average of 222 \pm 76 mg/L.

Dissolved CO₂ and CH₄ showed opposite trends to the tidal pattern with higher concentrations of greenhouse gases during low tide (Fig. 4). The two greenhouse gases were above atmospheric equilibrium (420 μ atm for CO₂ and 2 nM for CH₄) indicating the aquatic part of the wetland was a source of greenhouse gases to the atmosphere. Partial pressure of CO₂ varied between 1,590 to 5,866 μ atm with an average of 3,460 \pm 1,085 μ atm. CH₄ concentrations ranged from 8 to 65 nM and had an average of 27 \pm 11 nM.

The DIC and alkalinity showed similar trends (as both measures were dominated by HCO₃) and followed the tidal cycle, whereas DOC had an opposite trend to the tidal cycle with higher concentrations at low tide (Fig. 3C). DIC ranged from 1,079 μ M at low tide to 2,487 μ M at high tide with an average of 1,907 \pm 456 μ M. Alkalinity had an average of

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Groundwater	observations.
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Sample	DO (mg/ L)	EC (μ S/ cm)	Sal (ppt)	pН	Temp (c)	Radon (Bq/ m ³)	SO ₄ (mg/ L)	DIC (µ M)	DOC (µ M)	Alkalinity (µM)	CO ₂ (μ Μ)	CO ₂ (µ atm)	CH ₄ (nM)
GW1	1.81	37,500	23.76	6.28	14.70	14,562	2,501	17,577	1,344	11,641	5,949	88,808	29.9
GW2	0.88	7,640	4.21	5.61	15.20	3604	451	6,684	337	1,490	5,192	68,953	19.1
GW3	0.64	1,516	0.76	6.34	15.20	11,497	734	9,346	971	5,474	3,874	50,382	13.0
GW4	0.65	670	0.33	5.39	14.60	4,292	187	3,686	649	489	3,193	41,440	14.1
GW5	0.25	23,800	14.41	5.11	14.70	2,483	1,935	1,476	1,306	142	1,327	18,764	15.5
GW6	0.40	10,700	6.05	5.53	12.50	2,550	1,062	8,670	1,952	1,643	7,024	94,317	28.1
GW7	0.19	38,400	24.39	5.91	14.00	10,475	3,055	15,543	2,868	7,038	8,508	127,530	54.4
GW8	0.60	27,900	17.15	6.36	15.00	4,840	1,899	14,199	2,427	9,667	4,543	65,173	40.5
GW9	0.44	44,200	28.52	6.06	14.30	4,937	3,031	13,071	2,778	7,246	5,830	89,582	7.5
GW10	0.52	10,570	5.97	6.25	13.80	3,006	1,033	11,889	3,118	6,649	5,243	70,361	8.2
Average	0.64	20,290	13	5.9	14.40	6,225	1,588	10,214	1,775	5,148	5,068	71,531	23.0
Std.	0.46	16,159	10	0.4	0.82	4,313	1,046	5,207	991	992	2,008	30,668	15.2
Dev.													



Fig. 4. Scatter plots between surface water radon and chemical and physical parameters (Left Panel), dissolved carbon components (Center Panel), and, scatter plots of EC and dissolved carbon components (Right Panel).

 $2,\!165\pm528\mu$ M and ranged from 1215 to 2833μ M. DOC ranged from 651μ M at high tide to $1,\!222\mu$ M at low tide with an average of 876 \pm 157 μ M.

3.2. Groundwater observations

Groundwater observations are presented in Table 1. Groundwater pH ranged from 5.11 to 6.36, while DO ranged from 0.19 mg/L to 1.81 mg/L. Average pH (5.9 \pm 0.4) and DO (0.64 \pm 0.46 mg/L) in groundwater were lower than surface water observations (pH, 7.1 \pm 0.29; DO 0.8 \pm 1.2 mg/L), while groundwater had a higher average temperature (14.6 °C) than surface water (12.7 °C). Groundwater was brackish to saline with groundwater EC highly variable and ranging between 670 and 44,200 μ S/cm and an average of 20,290 \pm 16,159 μ S/cm. This was higher than the average surface water EC (4,393 \pm 1,504 μ S/cm) during sampling. Radon in groundwater ranged from 2,482 to 14,562 Bq/m³ with an average of 6,225 \pm 4,313 Bq/m³ that was more than an order of magnitude higher than average surface water radon concentration (135 \pm 59 Bq/m³). Sulfate concentrations in groundwater ranged from 186 to 3,055 mg/L with an average of 1, 588 \pm 1,046 mg/L.

Groundwater DIC ranged from 1,476 to 17,577 μ M with an average of 10,214 \pm 5,207 μ M, five-fold higher than surface water DIC. Groundwater DOC ranged from 337 to 3,117 with an average of 1,775

 \pm 991µM that was two-fold higher than the average surface water DOC. Groundwater alkalinity ranged from 141 to 11,641µM with an average

alkalinity of 5,148 \pm 992 μM that was 2-fold higher than those observed in surface water. Dissolved CO₂ concentrations ranged from 18,763 to 127,530 μM with an average of 71,531 \pm 30,668 μM , a magnitude higher than surface water observations. CH₄ concentrations ranged from 7 to 54 nM with an average concentration of 23 \pm 15 nM which was slightly lower than the average surface water CH₄ concentration (27 \pm 11 nM).

Table 2

Radon mass balance parameters (i.e., sources and sinks) and groundwater discharge rate. The areal groundwater discharge rate (cm/d) was calculated by dividing the total groundwater discharge rate by the wetland area.

Units	Value
m ²	$2,\!020,\!240 \pm 202,\!058$
m ²	$15,800 \pm 3,160$
Bq/s	18.5 ± 4.6
Bq/s	0.2 ± 0.06
Bq/s	6.8 ± 1.2
Bq/s	136 ± 49
Bq/s	$\textbf{4.9} \pm \textbf{1.0}$
Bq/s	0.5 ± 0.1
Bq/s	156.4 ± 49.3
Bq/m ³	$6,225 \pm 4,131$
m ³ /d	2170 ± 1181
cm/d	13.7 ± 7.4
	Units m ² Bq/s Bq/s Bq/s Bq/s Bq/s Bq/s Bq/s Bq/m ³ m ³ /d cm/d

3.3. Groundwater discharge and solute flux estimations

The radon mass balance model (Table 2) showed that the ground-water discharge rate was 13.3 \pm 7.4 cm/d (0.03 \pm 0.01 m³/s). This showed that the contribution of groundwater discharge to the wetland surface water flow (0.25 m³/s) was approximately 10 %. The largest sink in the radon mass balance model was atmospheric evasion of radon, while the largest source of radon to the water column was groundwater discharge. Radon sources other than groundwater (sediment diffusion and radium decay) were insignificant and accounted for less than 10 % of the required radon to complete the mass balance.

The calculated groundwater discharge rate was multiplied by the average concentration of the five carbon components in the groundwater to obtain the groundwater driven carbon fluxes (Table 3). Groundwater was a source of alkalinity, DIC, DOC, CO₂ and CH₄ to the creek. Groundwater alkalinity, DIC and DOC fluxes were 5.5 ± 4.3 , 11.0 ± 6.2 , and $1.9 \pm 1.8 \text{ mmol/m}^2/\text{d}$, respectively. The groundwater driven CO₂ flux was $5.4 \pm 4.3 \text{ mmol/m}^2/\text{d}$ and the groundwater driven CH₄ flux was $0.024 \pm 0.016 \text{ mmol/m}^2/\text{d}$.

3.4. Carbon export

The wetland was a net lateral aquatic exporter of alkalinity, DIC, DOC and CO₂ during the time of the experiment (Table 3). The wetland did not laterally export CH₄, although groundwater was a source of CH₄ to the creek (wetland water column). The wetland's DIC and DOC export were $35.5 \pm 13.1 \text{ mmol/m}^2/\text{d}$ and $1.3 \pm 0.5 \text{ mmol/m}^2/\text{d}$, respectively. Wetland alkalinity export was $39.4 \pm 14.6 \text{ mmol/m}^2/\text{d}$ and free CO₂ export was $6.5 \pm 2.1 \text{ mmol/m}^2/\text{d}$. Free CO₂ export was 20 % of the DIC export from the wetland, while alkalinity export was dominant and an order of magnitude higher than free CO₂ export.

3.5. Greenhouse gas fluxes

The average gas transfer velocity calculated using a combination of equations proposed in the literature was 0.9 ± 1.8 m/d (Table 4). These equations consider various factors including wind as the primary driver

Table 3

Groundwater fluxes of carbon into surface water and surface water carbon exports from the wetland. Atmospheric carbon fluxes are also presented. A positive value indicates export whilst a negative value indicates import.

Parameter	Units	GW Flux	SW Flux	Atmospheric Flux	GW Contribution (%)
SW flow	m ³ /s	$0.03 \pm$	$0.27 \pm$		9.4
		0.02	0.03		
Alkalinity	mol/d	11,175	79,534		14
		\pm 8,775	±		
			28,633		
DOC	mol/d	3,853 \pm	2,677 \pm		145
		2,638	964		
DIC	mol/d	22,721	71,815		30
		±	±		
		16,457	25,855		
CO_2	mol/d	11,002	13,194	210,150 \pm	5
		\pm 8,622	\pm 3,958	158,051	
CH_4	mol/d	0.05 \pm	-0.12	42 ± 32	0
		0.04	± 0.03		
Alkalinity	mmol/	5.5 \pm	39.4 \pm		14
	m²/d	4.3	14.6		
DOC	mmol/	$1.9 \pm$	$1.3 \pm$		145
	m²/d	1.8	0.5		
DIC	mmol/	11.0 \pm	$35.5 \pm$		30
	m²/d	6.2	13.1		
CO_2	mmol/	5.4 \pm	$6.5 \pm$	104 ± 209	5
	m²/d	4.3	2.1		
CH_4	mmol/	0.024 \pm	$-0.01~\pm$	21 ± 42	0
	m²/d	0.016	0.003		

Table 4

Gas transfer velocity models used to calculate CO_2 and CH_4 fluxes at the water–air interface. The reported errors denote the standard deviation arising from natural variability.

Equation	Variables	Gas Transfer Velocity (K) (m/d)	CO ₂ flux (mM/ m ² /d)	CH4 flux (µM/ m²/d)	Reference
$k(600) = 0.31U^2$	Wind	0.2 ± 0.4	17 ± 43	4 ± 9	(Wanninkhof, 1992)
$egin{aligned} k(600) &= \ 0.207 \ + \ (0.217 U^{1.7}) \end{aligned}$	Wind	0.3 ± 0.2	$\begin{array}{c} 32 \pm \\ 25 \end{array}$	7 ± 6	(Cole and Caraco, 1998)
$k(600) = 0.228 \times (U^{2.2} + 0.168)$	Wind	0.2 ± 0.4	$\begin{array}{c} 20 \pm \\ 40 \end{array}$	4 ± 9	(Crusius and Wanninkhof, 2003)
$k(600) = 0.0064 \times (U^{1.8}) + 0.497$	Wind	0.1 ± 0.05	$\begin{array}{c} 12 \pm \\ 5.9 \end{array}$	2 ± 1	(Cole et al., 2010)
$k(600) = 1.19^{0.35U}$	Wind	$\textbf{0.6} \pm \textbf{0.5}$	$\begin{array}{c} 67 \pm \\ 61 \end{array}$	14 ± 13	(Raymond and Cole, 2001)
$egin{array}{llllllllllllllllllllllllllllllllllll$	Wind	0.6 ± 1.1	$\begin{array}{c} 69 \pm \\ 129 \end{array}$	$\begin{array}{c} 15 \pm \\ 28 \end{array}$	(Borges et al., 2004a)
$k(600) = 1 + 1.719 v^{0.5} imes h^{-0.5} + 2.58 U$	Wind, current velocity, depth	$\textbf{5.7} \pm \textbf{15.0}$	661 ± 1711	79 ± 46	(Borges et al., 2004b)
$egin{aligned} k(600) &= \ 0.77V V^{0.5} imes \ h^{-0.5} + \ 0.266 U^2 \end{aligned}$	Wind, current velocity, depth	0.3 ± 0.5	34 ± 58	7 ± 12	(Ho et al., 2016)
$k(600) = 1.539v^{0.5} \times h^{-0.5}$	Current velocity, depth	0.3 ± 0.4	27 ± 36	6± 8	(O'Connor and Dobbins, 1958)
Average		0.9 ± 1.8	104 ± 209	21 ± 42	

of gas evasion and also account for current velocity and depth. This gas transfer velocity resulted in an average water to atmosphere CO_2 flux of 104 \pm 209 mmol/m²/d and water to atmosphere CH_4 flux of 21 \pm 42 $\mu mol/m²/d$.

4. Discussion

This study aimed to quantify the hydrological carbon exports of the five main carbon components from a restored coastal wetland and determine the contribution of groundwater inputs to the wetland's carbon export. The results indicated that, with the contribution of groundwater-driven carbon, the wetland was a source of carbon to both the atmosphere and estuary. However, the carbon exports from both the wetland and groundwater observed in this restored wetland were found to be lower than those reported in the literature for natural or mature wetlands (Alongi, 2020a; Reithmaier et al., 2023; Yau et al., 2022). In the following sections, we provide a technical discussion on the estimated groundwater discharge rate and its drivers, as well as wetland and groundwater carbon exports. Additionally, we explore greenhouse gas emissions from the wetlands and discuss the implications of the results for coastal wetland restoration.

4.1. Groundwater discharge rates and drivers

The estimated groundwater discharge rate $(13.7 \pm 5.4 \text{ cm/d})$ indicated a considerable amount of groundwater discharging into the wetland (Table 2). Radon in surface water showed a negative but weak correlation with water depth implying that tidal pumping was a partial driver of radon (Fig. 4). A potential further source of radon could be fresh surface water inputs from the upper areas of the wetland, although,

this is unlikely as, due to the gaseous nature of radon, most radon would degas and not reach the mouth of the wetland where measurements were recorded. The wide range of EC observed in groundwater samples (670–44,200 µS/cm; Table 1) indicates that mixing between freshwater and saline water was occurring in the subsurface of the wetland. This mixing can explain why no correlation exists between radon and EC in the groundwater samples (not shown) and suggests a combination of forces including marine tidal pumping and terrestrial hydraulic gradient were driving groundwater discharge in the wetland. The wet conditions prior to the sampling would have increased groundwater levels and caused higher than usual hydraulic gradients contributing to increased groundwater discharge. The lack of correlation between radon and EC in the surface water samples (Fig. 4) further supports the notion that a combination of marine saline and terrestrial freshwater sources were driving radon and groundwater discharge from the wetland.

By sampling the carbon fluxes at the mouth of the main drain of the restored wetland we captured the integrated aqueous carbon fluxes. The high frequency of sampling, continuous for ²²²Rn, CH₄, CO₂, EC, pH, temperature, and hourly for all other parameters allowed us to capture the temporal variability of the system. Our sampling period was characterized by wet antecedent conditions, typical of those experienced on the Australian east coast during the La Niña weather conditions that prevailed during 2022. The surface water composition was therefore fresher than usual, which could have biased the results in several ways. While antecedent groundwater data are not available for comparison, it is expected that the groundwater levels during the sampling campaign were higher than average due to higher groundwater recharge, resulting in a larger hydraulic gradient and increased flow towards the estuary (and potential groundwater carbon export) (Santos et al., 2013). During average (drier) conditions, groundwater flow and therefore the carbon groundwater export is likely to be lower.

Our sampling occurred during the transitional phase from spring to neap tide (avoiding either extreme) and as such, may represent the average groundwater discharge driven by tides that would occur at the site. It is also possible that the wetter conditions may have mobilized, flushed and diluted shallow and more permeable stores of carbon prior to our sampling. Such mobilization of the subsurface carbon pool by a large rainfall event was observed at the nearby Tomaree wetland (McDonough et al., 2020). A similar mobilization, flushing and dilution at our site could have led our results to be biased towards lower carbon exports. Overall, the fresher conditions of the estuary could also have altered the redox conditions across the estuary that would potentially have changed carbon speciation and fluxes (Crosswell et al., 2017). A pulse of fresh surface water could temporarily have oxidized the shallow and more permeable parts of the wetland and temporarily reduced methanogenesis and the flux of CH₄. Our sampling campaign was unable to capture such dynamics and a much longer, continuous monitoring campaign (presenting increased costs and logistical challenges) would be needed to capture the dynamics of variable climatic forcings.

The groundwater discharge rate observed here, falls within the wide range (i.e., 2 to 36 cm/d) of previously reported discharge rates from coastal estuarine and wetland systems using radioisotope tracer techniques (Chen et al., 2018; Tait et al., 2017). For determining the groundwater composition, we sampled 10 piezometers. The uncertainty of the radon groundwater end-member is a common challenge when using such techniques (Rodellas et al., 2021). This is because groundwater radon concentrations can vary significantly both temporally and spatially (Mullinger et al., 2009; Smith et al., 2008). Here the wide range of radon concentration in groundwater samples $(2,482 - 14,562 \text{ Bq/m}^3)$ led to an uncertainty of 54 % in the calculated groundwater discharge. Collecting additional groundwater samples to constrain the end-member concentration, could potentially reduce the variability in the data and produce a lower standard error. However, in a previous study, Sadat-Noori et al. (2015) conducted an experiment into the required number of groundwater samples to achieve a reduction of the standard error. Their finding indicated that the standard error showed minimal change once

the sample size reached 12. They reported a standard error of 20 % based on 27 samples. Here, the standard error to the average was 21 % (n = 10). This is similar to previous studies in Australia and overseas. For example, Atkins et al. (2013) collected 10 groundwater samples near a saltmarsh coastal creek on the east coast of Australia and reported a standard error of 25 %. Coluccio et al. (2021) sampled 10 groundwater bores from adjacent to a coastal lagoon in New Zealand which led to a standard error of 17 %.

4.2. Surface water carbon export from the wetland

The wetland was a source of alkalinity, DOC, DIC and CO2, except CH₄, with most of the exported carbon being in the form of alkalinity (Table 3). Very few previous studies identified alkalinity exports from saltmarsh wetlands, with none reporting alkalinity exports from a restored saltmarsh coastal wetland. Consequently, results were compared to both saltmarsh and mangrove coastal wetlands. In the current study, alkalinity exports were lower than those from a saltmarsh wetland in southern Australia (310 mmol/ m^2/d) (Faber et al., 2014), six mangrove surrounded wetlands on the east coast of Australia (67 mmol/ m^2/d) (Sippo et al., 2016), and a salt marsh wetland in China (78 mmol/ m^2/d). Conversely, the alkalinity export was higher than that from a mixed mangrove/salt marsh creek in Australia (12 mmol/ m^2/d) (Santos et al., 2019) and a small estuary surrounded by mangrove/salt marsh wetlands (27 mmol/m²/d) (Sadat-Noori et al., 2016). Export of alkalinity from saltmarsh wetlands to the ocean can have a local buffering effect on coastal acidification (Sippo et al., 2016). The primary processes contributing to the production of alkalinity are net denitrification, release of ammonia, iron reduction, dissolution of carbonate minerals, and reduction of sulfate (Hammond et al., 1999). Here, the main processes contributing to alkalinity were likely due to organic matter decomposition (high CO2 in groundwater) and sulfate reduction in the wetland (Table 1).

Surface water DOC versus EC plot indicated a convex trend (sink) with the conservative mixing line suggesting an upstream source (Fig. 4). Surface water DOC increased during low tides (Fig. 3) and showed a strong positive correlation with radon (groundwater tracer) indicating that groundwater was a likely source of DOC to the wetland (Fig. 4). In our study, DOC exports $(1.3 \pm 0.5 \text{ mmol/m}^2/\text{d})$ were lower than most DOC exports reported in the literature including from a mangrove wetland in Australia (197 mmol/m²/d) (Sadat-Noori and Glamore, 2019), a mixed mangrove/salt marsh creek in Australia (283 $mmol/m^2/d$) (Santos et al., 2019), a saltmarsh wetland in China (348 $mmol/m^2/d$). Here, the estimated DIC export was an order of magnitude greater than the commonly investigated DIC exports. A recent global study of saltmarsh reported DIC exports varying from -2 - 1200 mmol/ m^2/d global (Reithmaier et al., 2023), while in another global study, the average saltmarsh DIC export was estimated to be $121 \pm 160 \text{ mmol/m}^2/$ d (Alongi, 2020a). While the surface water DIC export observed here $(35.5 \pm 13.1 \text{ mmol/m}^2/\text{d})$ falls within the reported range it is in the lower end of the range and is 3.5-times smaller than the global average. The sources of DIC are likely from the saltmarsh environment surrounding the creek (Fig. 1). A DIC versus EC scatter plot (Fig. 4) showed a slight concave (production) trend indicating possible mid-creek inputs of DIC.

4.3. Groundwater-driven carbon fluxes to the wetland

Groundwater was a source of dissolved carbon to the wetland with most of the total groundwater carbon input to surface water being in the form of DIC (46 %), followed by alkalinity (23 %) which was mainly in the form of HCO₃, whereas the smallest contribution was from CH₄ (<1%) (Table 3). The groundwater to surface water ratios of DOC concentration indicated groundwater discharge was a source of DOC (although small) to the wetland water column. Previous studies have also reported groundwater inputs of DOC to coastal wetlands (Correa et al., 2022; Sadat-Noori and Glamore, 2019; Webb et al., 2019). The addition of DOC into surface waters from groundwater can result in elevated respiration rates and higher concentrations of CO_2 within the water column. This, in turn, can lead to conditions of supersaturation, causing the water column to become a source of CO_2 to the atmosphere (Gatland et al., 2014; Wang et al., 2022).

Previous studies in the literature reporting groundwater fluxes of the five main carbon components are shown in Table 5. The table highlights that groundwater driven DOC and DIC fluxes can be as high as 1,632 and 1,963 mmol/m²/d in tidal creeks, respectively. However, here, the groundwater driven DOC and DIC fluxes were significantly lower than those observed previously. Here, groundwater-derived CO2 fluxes (5.4 \pm 4.3 mmol/m²/d) contributed 5 % to the total CO₂ losses from the wetland (surface water exports and atmospheric emission). When compared to other studies, our groundwater-derived CO₂ fluxes fall in the lower bound of fluxes reported previously from coastal tidal wetlands which range from $\sim 1 - 217 \text{ mmol/m}^2/d$ (Wang et al., 2022). Very few studies have reported groundwater driven CH₄ fluxes in saltmarsh wetlands. Based on the available studies, groundwater CH₄ fluxes in saltmarsh wetlands range from 0.0027 to 0.9 mmol/ m^2/d (Porubsky et al., 2014; Santos et al., 2019; Schutte et al., 2020; Yau et al., 2022). Here, groundwater driven CH₄ fluxes were lower at 0.024 μ mol/m²/d. This can be due to the abundant supply of seawater sulfate suppressing methanogenesis (Raghoebarsing et al., 2006) as evidenced by the high sulfate concentrations in the wetland groundwater shown in Table 1.

During the period of past anthropogenic disturbance of the wetland, the site would have undergone substantial on-ground works. For instance, deep drainage lines were cut into the site facilitating drainage by enhancing lateral groundwater transport over vertical transport. Although the tide has been restored onsite, the drainage lines have not been infilled and lateral flow would remain a significant transport pathway (Liu et al., 2013). This has influenced the wetland hydraulics and the carbon export pathways after tidal restoration, as it ensures that deeper groundwater is still exported rapidly via the drainage lines rather than the slow wicking to the surface typically experienced in natural undrained systems (Glamore and Indraratna, 2009; Waddington and Price, 2000).

Furthermore, the drainage and desaturation of wetland sediments, which is often the main purpose of the initial altering of wetlands, leads to increased oxidation of stored organic matter (Mitsch et al., 2013). In systems with acid sulfate soils, such as those common on the east coast of

Australia, the disturbance of natural wetlands by drainage also results in the exposure of sulphidic material to atmospheric oxygen and an increase in groundwater acidity (Johnston et al., 2011). Such acidification may have implications for the stability of the sediments, which could lead to erosion and export of stored carbon. In contrast, lower pH associated with acidification may reduce the mobility of DOC, as it tends to be less mobile at low pH (Grybos et al., 2009).

As a restoration measure, the re-saturation of the wetland sediments by tidal waters would reestablish a limitation on oxygen transport into the sediment and create anoxic conditions with direct implications for organic matter degradation as it would slow down the organic matter degradation rate and subsequent fluxes of DIC and DOC (Mikha et al., 2005; Sahrawat, 2004). We hypothesise that the reduced organic matter oxidation due to restorations may be another potential reason for observed lower carbon export in the current study. However, the resaturation would also reduce the oxidation of sulfide minerals and over time reduce the acidity due to the buffering by tidal flushing. The implications of restoration on the reversal of the combined processes of oxidation and acidification and in turn on the biogeochemical capture of carbon in the subsurface over the short to long term is unknown and need further investigation. It is also recommended that future research investigates the capacity of degraded sediments to chemically sorb additional DOC upon restoration.".

4.4. Greenhouse gas emissions

Fig. 5 shows a conceptual model of the carbon dynamics in the wetland. Measurement of dissolved CO₂ and CH₄ showed that the wetland creek was a source of CO₂ and CH₄ to the atmosphere (Table 4). The average CO₂ flux (104 \pm 209 mmol/m²/d) was similar to previous studies conducted in coastal estuarine and wetland ecosystems (Ho et al., 2014; Rosentreter et al., 2017; Wang et al., 2022), but ~ 2.5 fold higher than the global average CO₂ flux from lower estuaries (19 – 59 mmol/m²/d) (Borges and Abril, 2011; Cai, 2011). The contribution of groundwater discharge to the CO₂ atmospheric fluxes from the wetland was small (5%). The average CH₄ atmospheric flux from the wetland (21 \pm 41 µmol/m²/d) was at the lower end of the range reported for global CH₄ emissions from tidal estuarine environments (40 – 600 µmol/m²/d) (Borges and Abril, 2011).

Greenhouse gas emissions from wetlands are primarily driven by biological (e.g., decomposition of organic matter by microbial

Table 5

Previous studies reporting groundwater flux of alkalinity, DOC, DIC, CO_2 and CH_4 from mangrove and salt marsh environments. Units are in mmol/m²/d. Table updated from Sadat-Noori et al., (2015).

Location	System	GW-	GW-	GW-DIC	GW-	GW-CH ₄	Source
		Alk	DOC		CO_2		
Tomago Wetlands, Australia	Coastal salt marsh wetland	5.5	1.9	11.0	5.4	0.000024	This study
Dafeng Milu National Nature Reserve, China	Salt marsh	37	0.0	42	4.8	0.003	Yau et al. (2022)
North Inlet, SC, USA	Salt marsh		19	602	57		Correa et al. (2022)
Paraty city (Rio de Janeiro), Brazil	Macrotidal Mangrove				35.5		Cabral et al. (2024)
Sapelo Island, Georgia, USA	Salt marsh					0.0027 - 1.2	Schutte et al. (2020)
Dafeng Milu National Nature Reserve, China	Salt marsh		106	21.8	0.95	0.05	Chen et al. (2022)
Southeast Australia	Macrotidal embayment	178	12.5	290			Faber et al. (2014)
Newcastle Australia	Mangrove wetland		107	954			Sadat-Noori and Glamore
							(2019)
Gold coast, Australia	Wetland-drained coastal canal		70	68			Davis et al. (2020)
Sapelo Island, Gabon	Marsh-dominated estuary						Wang and Cai (2004)
Hat Head, Australia	Tidal creek	346	1632	921		31	Sadat-Noori et al. (2016)
Okatee estuary, USA	Salt marsh/estuary		64	1079.2		0.9	Porubsky et al. (2014)
North Creek, Australia	Fresh water tidal creek			1810			Atkins et al. (2013)
Moreton Bay, Australia	Mangrove tidal creek		24	250			Maher et al. (2013)
Yarra River, Australia	Salt wedge estuary		21	349			Santos et al. (2012a)
Indian River Lagoon, USA	Coastal lagoon			120-340			Dorsett et al. (2011)
West coast of Florida, USA	Sandy beach		19–27				Santos et al. (2009)
Okatee Estuary, USA	Salt marsh/estuary		170	1963			Moore et al. (2006)
North Inlet, USA	Salt marsh/estuary			171			Cai et al. (2003)
South Carolina, USA	Tidal creek		50				Goñi and Gardner (2003)



Fig. 5. Conceptual figure of the carbon dynamics at the site over the detailed sampling duration (i.e., diel tidal cycle). All units reported are in mmol/m²/day except for surface water, groundwater and atmospheric CH₄ fluxes which are reported in μ mol/m²/day. The positive values indicate export and negative value indicates an influx (import).

respiration) and hydrological processes (e.g., water level fluctuation, mixing of fresh and saline water, groundwater inputs) (Bridgham et al., 2013; Wang et al., 2022). Groundwater can contribute to greenhouse gas emissions by directly delivering dissolved CO2 and CH4 to wetlands, or indirectly by supplying DOC to the water column and increasing microbial respiration in the surface water (Maher et al., 2015). Here, radon showed a positive correlation with DOC, CO2 and CH4 in the water column, indicating groundwater was as a source of DOC and both gases (Fig. 4). However, flux calculations indicated that groundwater was not a significant source of CH₄ emission to the atmosphere. This is due to the presence of sulfate in the saline and brackish groundwater inhibiting significant methane production (Table 1). This study found that the quantity of CO2 released into the atmosphere from the wetland far exceeded that of CH₄, suggesting that CO₂ emissions had a greater influence on carbon gaseous fluxes. Despite this, it is important to note that CH₄ is 34 times more potent (ability to capture heat) as a greenhouse gas compared to CO2. Therefore, even minor increases in the amount of CH₄ emission could have a more significant impact on global warming than CO₂ emissions.

Overall, the wetland was a sources of greenhouse gases to the atmosphere and exported DOC, DIC and alkalinity to the ocean. While DOC can be mineralised and evade into the atmosphere as CO_2 as it moves out of the wetland, alkalinity can stay dissolved in the ocean for hundreds of years (Middelburg et al., 2020). Thus, alkalinity export can be a long-term carbon sequestration mechanism, further demonstrating the value of blue carbon ecosystems.

4.5. Implications for coastal wetland restoration

Globally, environmental markets (e.g. carbon and biodiversity markets) are being created to support blue ecosystem restoration as the world moves swiftly toward realising the potential benefits of carbon sequestration (and environmental co-benefits) (Sapkota and White, 2020). Coastal wetland restoration is known to remove atmospheric carbon, however, there is large uncertainty in the amount of carbon that is sequestrated, stored long-term in sediments, evaded back to the atmosphere, and transported to the ocean (Deb and Mandal, 2021; Negandhi et al., 2019; Rosentreter et al., 2021). Wetland carbon transported (export) to the ocean occurs though hydrological processes and, as such, any hydrological restoration methods that alter wetland hydrology can affect carbon movement, storage, and release (Bogard et al., 2020). Further, hydrological restoration effects groundwater flow, pathways and quality, all shown to be important drivers of carbon transport and evasion through groundwater (Correa et al., 2022; Schutte et al., 2020).

Traditionally, coastal wetlands have been widely reclaimed through the construction of drainage channels with one-way floodgates to prevent tidal inundation and lower the prevailing water levels in the drains. This can temporarily increase the hydraulic gradient between drainage channels and groundwater, resulting in elevated discharge and carbon input into surface waters until water levels adjust. Conversely, inundation by surface water, as typically occurs during restoration, can reduce the hydraulic gradient; potentially decreasing groundwater discharge and altering the point of discharge, and thus reducing dissolved carbon export from wetland sediments. These processes would also be transitional, as the net groundwater surplus would have to discharge, and the system would eventually achieve a new dynamic steady state. Further, reintroducing tidal flows can facilitate tidal pumping, with marine or brackish surface water flushing sediments on each tide, aided by macropores and animal burrows enhancing groundwater flow (Xiao et al., 2019). To date however, no literature reports lateral carbon export from restored coastal wetlands, highlighting a significant gap in understanding and quantification. The lack of long-term studies also means that there is very limited knowledge of the temporal dynamics of how the hydrologic and carbon fluxes adjust and change due to wetland restoration measures and the maturation of the restored wetland. The limited knowledge of carbon fate post-restoration and hydrology's role in carbon dynamics may impact global warming mitigation efforts focused on coastal wetlands.

The implications of the current study are 1) groundwater driven carbon fluxes can be exported laterally to the ocean or enter the atmosphere, and as such, the groundwater carbon fluxes should be accounted for when developing coastal wetland carbon budgets; 2) in restored sites it may be worthwhile considering infilling artificial drainage networks to reduce groundwater discharge and associated fast lateral carbon transport in drains. This ensures that restored sites have groundwater regimes similar to natural/undisturbed sites; and 3) this study highlights the important role of surface water — groundwater interactions for carbon processes and transport and should be considered when developing restoration strategies for coastal tidal wetlands, especially when the restoration aim is to sequester carbon.

5. Conclusion

This study provides novel insights into the role of groundwater discharge for carbon and greenhouse gas dynamics in a restored coastal wetland by reporting, for the first time, groundwater-driven fluxes of the five main carbon components from a restored coastal wetland. The results demonstrate that groundwater discharge can be a source of carbon to the surface water column in restored wetlands, contributing to both dissolved inorganic and organic carbon. The finding suggests that wetland restoration could reduce groundwater driven carbon inputs and the overall carbon exports from wetlands. Nevertheless, the findings highlight the importance of considering shallow groundwater flow dynamics in carbon budgets of restored coastal wetlands. Furthermore, the study highlights the complex interactions between surface water and groundwater in coastal restored wetland ecosystems and the need for integrated approaches to understand their biogeochemical processes, flow and transport mechanisms. Future research should focus on expanding the temporal and spatial scales of investigations to capture the variability inherent in these systems, as well as assessing the longterm impacts of restoration activities on carbon sequestration and greenhouse gas emissions.

CRediT authorship contribution statement

Mahmood Sadat-Noori: Writing – original draft, Visualization, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Martin S. Andersen:** Writing – review & editing, Validation, Supervision, Resources, Investigation, Data curation, Conceptualization. **Katrina Waddington:** Writing – review & editing, Visualization, Investigation, Data curation. **Jamie Ruprecht:** Writing – review & editing, Investigation, Data curation. **Tobias A. Tucker:** Writing – review & editing, Investigation, Data curation. **William Glamore:** Writing – review & editing, Supervision, Project administration, Methodology, Investigation, Data curation, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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References

- Alongi, D.M., 2020a. Carbon balance in salt marsh and mangrove ecosystems: a global synthesis. J. Mar. Sci. Eng. 8, 767.
- Alongi, D.M., 2020b. Carbon cycling in the world's mangrove ecosystems revisited: significance of non-steady state diagenesis and subsurface linkages between the forest floor and the coastal ocean. Forests 11, 977.
- Atkins, M.L., Santos, I.R., Ruiz-Halpern, S., Maher, D.T., 2013. Carbon dioxide dynamics driven by groundwater discharge in a coastal floodplain creek. J. Hydrol. 493, 30–42.
- Bogard, M.J., Bergamaschi, B., Butman, D., Anderson, F., Knox, S., Windham-Myers, L., 2020. Hydrologic export is a major component of coastal wetland carbon budgets. Global Biogeochem. Cycles 34 e2019GB006430.
- Borges, A.V., Abril, G., 2011. Carbon dioxide and methane dynamics in estuaries. Treatise on Estuarine and Coastal Science 119–161.
- Borges, A., Delille, B., Schiettecatte, L.-S., Gazeau, F., Abril, G., Frankignoulle, M., 2004a. Gas transfer velocities of CO2 in three European estuaries (Randers Fjord, Scheldt and Thames). Limnol. Oceanogr. 49.
- Borges, A.V., Vanderborght, J.-P., Schiettecatte, L.-S., Gazeau, F., Ferrón-Smith, S., Delille, B., Frankignoulle, M., 2004b. Variability of the gas transfer velocity of CO2 in a macrotidal estuary (the Scheldt). Estuaries 27, 593–603.
- Bridgham, S.D., Cadillo-Quiroz, H., Keller, J.K., Zhuang, Q., 2013. Methane emissions from wetlands: biogeochemical, microbial, and modeling perspectives from local to global scales. Glob. Chang. Biol. 19, 1325–1346.
- Burnett, W.C., Aggarwal, P.K., Bokuniewicz, H., Cable, J.E., Charette, M.A., Kontar, E., Krupa, S., Kulkarni, K.M., Loveless, A., Moore, W.S., Oberdorfer, J.A., Oliveira, J., Ozyurt, N., Povinec, P., Privitera, A.M.G., Rajar, R., Ramessur, R.T., Scholten, J., Stieglitz, T., Taniguchi, M., Turner, J.V., 2006. Quantifying submarine groundwater discharge in the coastal zone via multiple methods. Sci. Total Environ. 367, 498–543.
- Burnett, W.C., Dulaiova, H., 2003. Estimating the dynamics of groundwater input into the coastal zone via continuous radon-222 measurements. J. Environ. Radioact. 69, 21–35.
- Cabral, A., Yau, Y.Y., Reithmaier, G.M., Cotovicz Jr, L.C., Barreira, J., Broström, G., Viana, B., Fonseca, A.L., Santos, I.R., 2024. Tidally driven porewater exchange and diel cycles control CO2 fluxes in mangroves on local and global scales. Geochim. Cosmochim. Acta.
- Cai, W.-J., 2011. Estuarine and coastal ocean carbon paradox: CO2 sinks or sites of terrestrial carbon incineration? Ann. Rev. Mar. Sci. 3, 123–145.
- Cai, W.-J., Wang, Y., Krest, J., Moore, W., 2003. The geochemistry of dissolved inorganic carbon in a surficial groundwater aquifer in North Inlet, South Carolina, and the carbon fluxes to the coastal ocean. Geochim. Cosmochim. Acta 67, 631–639.
- Chen, X., Zhang, F., Lao, Y., Wang, X., Du, J., Santos, I.R., 2018. Submarine groundwater discharge-derived carbon fluxes in mangroves: an important component of blue carbon budgets? J. Geophys. Res. Oceans.
- Chen, X., Santos, I.R., Hu, D., Zhan, L., Zhang, Y., Zhao, Z., Hu, S., Li, L., 2022. Porewater exchange flushes blue carbon from intertidal saltmarsh sediments into the sea. Limnol. Oceanogr. Lett. 7, 312–320.
- Cole, J.J., Bade, D.L., Bastviken, D., Pace, M.L., Van de Bogert, M., 2010. Multiple approaches to estimating air-water gas exchange in small lakes. Limnol. Oceanogr. Methods 8, 285–293.
- Cole, J.J., Caraco, N.F., 1998. Atmospheric exchange of carbon dioxide in a low-wind oligotrophic lake measured by the addition of SF6. Limnol. Oceanogr. 43, 647–656.

- Coluccio, K.M., Santos, I.R., Jeffrey, L.C., Morgan, L.K., 2021. Groundwater discharge rates and uncertainties in a coastal lagoon using a radon mass balance. Journal of Hydrology, 126436.
- Corbett, D., Burnett, W., Cable, P., Clark, S., 1998. A multiple approach to the determination of radon fluxes from sediments. J. Radioanal. Nucl. Chem. 236, 247–253.
- Correa, R.E., Xiao, K., Conrad, S.R., Wadnerkar, P.D., Wilson, A.M., Sanders, C.J., Santos, I.R., 2022. Groundwater carbon exports exceed sediment carbon burial in a salt marsh. Estuar. Coasts 45, 1545–1561.
- Crosswell, J.R., Anderson, I.C., Stanhope, J.W., Van Dam, B., Brush, M.J., Ensign, S., Piehler, M.F., McKee, B., Bost, M., Paerl, H.W., 2017. Carbon budget of a shallow, lagoonal estuary: transformations and source-sink dynamics along the river-estuaryocean continuum. Limnol. Oceanogr. 62, S29–S45.
- Crusius, J., Wanninkhof, R., 2003. Gas transfer velocities measured at low wind speed over a lake. Limnol. Oceanogr. 48, 1010–1017.
- Davidson, N.C., 2014. How much wetland has the world lost? Long-term and recent trends in global wetland area. Mar. Freshw. Res. 65, 934–941.
- Davis, K., Santos, I.R., Perkins, A.K., Webb, J.R., Gleeson, J., 2020. Altered groundwater discharge and associated carbon fluxes in a wetland-drained coastal canal. Estuar. Coast. Shelf Sci. 235, 106567.
- Deb, S., Mandal, B., 2021. Soils and sediments of coastal ecology: a global carbon sink. Ocean Coast. Manage. 214, 105937.
- Dorsett, A., Jennifer Cherrier a, Martin, J.B., Cable, J.E., 2011. Assessing hydrologic and biogeochemical controls on pore-water dissolved inorganic carbon cycling in a subterranean estuary: A 14C and 13C mass balance approach. Marine Chemistry 127, 76–89.
- Faber, P.A., Evrard, V., Woodland, R.J., Cartwright, I.C., Cook, P.L., 2014. Pore-water exchange driven by tidal pumping causes alkalinity export in two intertidal inlets. Limnol. Oceanogr. 59, 1749–1763.
- Gatland, J.R., Santos, I.R., Maher, D.T., Duncan, T.M., Erler, D.V., 2014. Carbon dioxide and methane emissions from an artificially drained coastal wetland during a flood: Implications for wetland global warming potential. J. Geophys. Res. Biogeo. 119, 1698–1716.
- Glamore, W., Indraratna, B., 2009. Tidal-forcing groundwater dynamics in a restored coastal wetland: implications of saline intrusion. Aust. J. Earth Sci. 56, 31–40.
- Glamore, W., Rayner, D., Ruprecht, J., Sadat-Noori, M., Khojasteh, D., 2021. Ecohydrology as a driver for tidal restoration: Observations from a Ramsar wetland in eastern Australia. PLoS One 16, e0254701.
- Goñi, M.A., Gardner, I.R., 2003. Seasonal dynamics in dissolved organic carbon concentrations in a coastal water-table aquifer at the forest-marsh interface. Aquat. Geochem. 9, 209–232.
- Grybos, M., Davranche, M., Gruau, G., Petitjean, P., Pédrot, M., 2009. Increasing pH drives organic matter solubilization from wetland soils under reducing conditions. Geoderma 154, 13–19.
- Gulliver, A., Carnell, P.E., Trevathan-Tackett, S.M., de Paula, D., Costa, M., Masqué, P., Macreadie, P.I., 2020. Estimating the potential blue carbon gains from tidal marsh rehabilitation: a case study from South Eastern Australia. Front. Mar. Sci. 7, 403.
- Hammond, D., Giordani, P., Berelson, W., Poletti, R., 1999. Diagenesis of carbon and nutrients and benthic exchange in sediments of the Northern Adriatic Sea. Mar. Chem. 66, 53–79.
- Ho, D.T., Ferrón, S., Engel, V.C., Larsen, L.G., Barr, J.G., 2014. Air-water gas exchange and CO2 flux in a mangrove-dominated estuary. Geophys. Res. Lett. 41, 108–113.
- Ho, D.T., Coffineau, N., Hickman, B., Chow, N., Koffman, T., Schlosser, P., 2016. Influence of current velocity and wind speed on air-water gas exchange in a mangrove estuary. Geophys. Res. Lett. 43, 3813–3821.
- Huber, S.A., Balz, A., Abert, M., Pronk, W., 2011. Characterisation of aquatic humic and non-humic matter with size-exclusion chromatography–organic carbon detection–organic nitrogen detection (LC-OCD-OND). Water Res. 45, 879–885.
- Johnston, S.G., Keene, A.F., Bush, R.T., Sullivan, L.A., Wong, V.N.L., 2011. Tidally driven water column hydro-geochemistry in a remediating acidic wetland. J. Hydrol. 409, 128–139.
- Kroeger, K.D., Crooks, S., Moseman-Valtierra, S., Tang, J., 2017. Restoring tides to reduce methane emissions in impounded wetlands: A new and potent Blue Carbon climate change intervention. Sci. Rep. 7, 11914.
- Liu, Y.N., Ni, H.W., Zeng, Z.W., Chai, C.R., 2013. Effect of disturbance on carbon cycling in wetland ecosystem. Adv. Mat. Res. 610, 3186–3191.
- Lovelock, C.E., Adame, M.F., Bradley, J., Dittmann, S., Hagger, V., Hickey, S.M., Hutley, L.B., Jones, A., Kelleway, J.J., Lavery, P.S., 2023. An Australian blue carbon method to estimate climate change mitigation benefits of coastal wetland restoration. Restor. Ecol. 31, e13739.
- MacIntyre, S., Wanninkhof, R., Chanton, J., 1995. Trace gas exchange across the airwater interface in freshwater and coastal marine environments. Biogenic trace gases: Measuring emissions from soil and water, 52-97.
- Macreadie, P.I., Ollivier, Q., Kelleway, J.J., Serrano, O., Carnell, P.E., Lewis, C.E., Atwood, T., Sanderman, J., Baldock, J., Connolly, R.M., 2017. Carbon sequestration by Australian tidal marshes. Sci. Rep. 7, 44071.
- Maher, D.T., Santos, I.R., Golsby-Smith, L., Gleeson, J., Eyre, B.D., 2013. Groundwaterderived dissolved inorganic and organic carbon exports from a mangrove tidal creek: The missing mangrove carbon sink? Limnol. Oceanogr. 58, 475–488.
- Maher, D.T., Cowley, K., Santos, I.R., Macklin, P., Eyre, B.D., 2015. Methane and carbon dioxide dynamics in a subtropical estuary over a diel cycle: Insights from automated in situ radioactive and stable isotope measurements. Mar. Chem. 168, 69–79.
- McDonough, L.K., O'Carroll, D.M., Meredith, K., Andersen, M.S., Brügger, C., Huang, H., Rutlidge, H., Behnke, M.I., Spencer, R.G., McKenna, A., 2020. Changes in groundwater dissolved organic matter character in a coastal sand aquifer due to rainfall recharge. Water Res. 169, 115201.

Mikha, M.M., Rice, C.W., Milliken, G.A., 2005. Carbon and nitrogen mineralization as affected by drying and wetting cycles. Soil Biol. Biochem. 37, 339–347.

Mitsch, W.J., Bernal, B., Nahlik, A.M., Mander, Ü., Zhang, L., Anderson, C.J., Jørgensen, S.E., Brix, H., 2013. Wetlands, carbon, and climate change. Landsc. Ecol. 28, 583–597

Moore, W.S., 2003. Sources and fluxes of submarine groundwater discharge delineated by radium isotopes. Biogeochemistry 75–93, 75–93.

Moore, W.S., Blanton, J.O., Joye, S.B., 2006. Estimates of flushing times, submarine groundwater discharge, and nutrient fluxes to Okatee Estuary, South Carolina. J. Geophys. Res. Oceans 1978–2012, 111.

Mullinger, N.J., Pates, J.M., Binley, A.M., Crook, N., 2009. Controls on the spatial and temporal variability of 222Rn in riparian groundwater in a lowland Chalk catchment. J. Hydrol. 376, 58–69.

Negandhi, K., Edwards, G., Kelleway, J.J., Howard, D., Safari, D., Saintilan, N., 2019. Blue carbon potential of coastal wetland restoration varies with inundation and rainfall. Sci. Rep. 9, 4368.

O'Connor, D.J., Dobbins, W.E., 1958. Mechanism of reaeration in natural streams. Trans. Am. Soc. Civ. Eng. 123, 641–666.

Ouyang, X., Lee, S.Y., 2020. Improved estimates on global carbon stock and carbon pools in tidal wetlands. Nat. Commun. 11, 1–7.

Pendleton, L., Donato, D.C., Murray, B.C., Crooks, S., Jenkins, W.A., Sifleet, S., Craft, C., Fourqurean, J.W., Kauffman, J.B., Marbà, N., 2012. Estimating global "blue carbon" emissions from conversion and degradation of vegetated coastal ecosystems. PLoS One 7, e43542.

Porubsky, W.P., Weston, N.B., Mooreb, W.S., Ruppel, C., Joye, S.B., 2014. Dynamics of submarine groundwater discharge and associated fluxes of dissolved nutrients, carbon, and trace gases to the coastal zone (Okatee River estuary, South Carolina). Geochim. Cosmochim. Acta 131, 81–97.

Raghoebarsing, A.A., Pol, A., Van de Pas-Schoonen, K.T., Smolders, A.J., Ettwig, K.F., Rijpstra, W.I.C., Schouten, S., Damsté, J.S.S., Op den Camp, H.J., Jetten, M.S., 2006. A microbial consortium couples anaerobic methane oxidation to denitrification. Nature 440, 918–921.

Raymond, P.A., Cole, J.J., 2001. Gas exchange in rivers and estuaries: Choosing a gas transfer velocity. Estuar. Coasts 24, 312–317.

Rayner, D., Glamore, W., 2010. Tidal innundation and wetland restoration of Tomago wetland: Hydrodynamic modelling. UNSW Water Research Laboratory—Technical Report, Sydney.

Reithmaier, G.M., Cabral, A., Akhand, A., Bogard, M.J., Borges, A.V., Bouillon, S., Burdige, D.J., Call, M., Chen, N., Chen, X., 2023. Carbonate chemistry and carbon sequestration driven by inorganic carbon outwelling from mangroves and saltmarshes. Nat. Commun. 14, 8196.

Rodellas, V., Stieglitz, T.C., Tamborski, J.J., van Beek, P., Andrisoa, A., Cook, P.G., 2021. Conceptual uncertainties in groundwater and porewater fluxes estimated by radon and radium mass balances. Limnol. Oceanogr. 66, 1237–1255.

Rosentreter, J.A., Maher, D.T., Ho, D.T., Call, M., Barr, J.G., Eyre, B.D., 2017. Spatial and temporal variability of CO2 and CH4 gas transfer velocities and quantification of the CH4 microbubble flux in mangrove dominated estuaries. Limnol. Oceanogr. 62, 561–578.

Rosentreter, J.A., Al-Haj, A.N., Fulweiler, R.W., Williamson, P., 2021. Methane and nitrous oxide emissions complicate coastal blue carbon assessments. Global Biogeochem. Cycles 35 e2020GB006858.

Sadat-Noori, M., Glamore, W., 2019. Porewater exchange drives trace metal, dissolved organic carbon and total dissolved nitrogen export from a temperate mangrove wetland. J. Environ. Manage. 248, 109264.

Sadat-Noori, M., Santos, I.R., Sanders, C.J., Sanders, L.M., Maher, D.T., 2015. Groundwater discharge into an estuary using spatially distributed radon time series and radium isotopes. J. Hydrol.

Sadat-Noori, M., Maher, D.T., Santos, I.R., 2016. Groundwater discharge as a source of dissolved carbon and greenhouse gases in a subtropical estuary. Estuar. Coasts 1–18.

Sadat-Noori, M., Anibas, C., Andersen, M.S., Glamore, W., 2021a. A comparison of radon, heat tracer and head gradient methods to quantify surface water-groundwater exchange in a tidal wetland (Kooragang Island, Newcastle, Australia). J. Hydrol. 598, 126281.

Sadat-Noori, M., Rankin, C., Rayner, D., Heimhuber, V., Gaston, T., Drummond, C., Chalmers, A., Khojasteh, D., Glamore, W., 2021b. Coastal wetlands can be saved from sea level rise by recreating past tidal regimes. Sci. Rep. 11, 1–10.

Sadat-Noori, M., Rutlidge, H., Andersen, M.S., Glamore, W., 2021c. Quantifying groundwater carbon dioxide and methane fluxes to an urban freshwater lake using radon measurements. Sci. Total Environ. 149184.

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Sahrawat, K.L., 2004. Organic matter accumulation in submerged soils. Adv. Agron. 81, 169–201.

Santos, I.R., Burnett, W.C., Dittmar, T., Suryaputra, I.G., Chanton, J., 2009. Tidal pumping drives nutrient and dissolved organic matter dynamics in a Gulf of Mexico subterranean estuary. Geochim. Cosmochim. Acta 73, 1325–1339.

Santos, I.R., Cook, P.L., Rogers, L., De Weys, J., Eyre, B.D., 2012a. The "salt wedge pump": Convection-driven pore-water exchange as a source of dissolved organic and inorganic carbon and nitrogen to an estuary. Limnol. Oceanogr. 57, 1415–1426.

Santos, I.R., Beck, M., Brumsack, H.-J., Maher, D.T., Dittmar, T., Waska, H., Schnetger, B., 2015. Porewater exchange as a driver of carbon dynamics across a terrestrial-marine transect: insights from coupled 222 Rn and pCO 2 observations in the German Wadden Sea. Mar. Chem. 171, 10–20.

Santos, I.R., de Weys, J., Tait, D.R., D. Eyre, B., 2013. The Contribution of Groundwater Discharge to Nutrient Exports from a Coastal Catchment: Post-Flood Seepage Increases Estuarine N/P Ratios. Estuaries and Coasts 36, 56–73.

Santos, I.R., Maher, D.T., Eyre, B.D., 2012b. Coupling automated radon and carbon dioxide measurements in coastal waters. Environ. Sci. Tech. 46, 7685–7691.

Santos, I.R., Maher, D.T., Larkin, R., Webb, J.R., Sanders, C.J., 2019. Carbon outwelling and outgassing vs. burial in an estuarine tidal creek surrounded by mangrove and saltmarsh wetlands. Limnol. Oceanogr. 64, 996–1013.

Sapkota, Y., White, J.R., 2020. Carbon offset market methodologies applicable for coastal wetland restoration and conservation in the United States: a review. Sci. Total Environ. 701, 134497.

Schutte, C.A., Moore, W.S., Wilson, A.M., Joye, S.B., 2020. Groundwater-driven methane export reduces salt marsh blue carbon potential. Global Biogeochem. Cycles 34 e2020GB006587.

Serrano, O., Lovelock, C.E., Atwood, T.B., Macreadie, P.I., Canto, R., Phinn, S., Arias-Ortiz, A., Bai, L., Baldock, J., Bedulli, C., 2019. Australian vegetated coastal ecosystems as global hotspots for climate change mitigation. Nat. Commun. 10, 1–10.

Sippo, J.Z., Maher, D.T., Tait, D.R., Holloway, C., Santos, I.R., 2016. Are mangroves drivers or buffers of coastal acidification? Insights from alkalinity and dissolved inorganic carbon export estimates across a latitudinal transect. Global Biogeochem. Cycles 30, 753–766.

Smith, C.G., Cable, J.E., Martin, J.B., Roy, M., 2008. Evaluating the source and seasonality of submarine groundwater discharge using a radon-222 pore water transport model. Earth Planet. Sci. Lett. 273, 312–322.

Stumm, W., Morgan, J.J., 1981. Aquatic chemistry: an introduction emphasizing chemical equilibria in natural waters. Wiley, New York.

Tait, D.R., Maher, D.T., Sanders, C.J., Santos, I.R., 2017. Radium-derived porewater exchange and dissolved N and P fluxes in mangroves. Geochim. Cosmochim. Acta 200, 295–309.

Waddington, J., Price, J., 2000. Effect of peatland drainage, harvesting, and restoration on atmospheric water and carbon exchange. Phys. Geogr. 21, 433–451.

Waltham, N.J., Elliott, M., Lee, S.Y., Lovelock, C., Duarte, C.M., Buelow, C., Simenstad, C., Nagelkerken, I., Claassens, L., Wen, C.K., 2020. UN Decade on Ecosystem Restoration 2021–2030—what chance for success in restoring coastal ecosystems? Front. Mar. Sci. 7, 71.

Wang, Z.A., Cai, W.-J., 2004. Carbon dioxide degassing and inorganic carbon export from a marsh-dominated estuary (the Duplin River): A marsh CO2 pump. Limnol. Oceanogr. 49, 341–354.

Wang, Z.-L., Sadat-Noori, M., Glamore, W., 2022. Groundwater discharge drives water quality and greenhouse gas emissions in a tidal wetland. Water Sci. Eng. 15, 141–151.

Wanninkhof, R., 1992. Relationship between wind speed and gas exchange over the ocean. J. Geophys. Res. Oceans 1978–2012 (97), 7373–7382.

Webb, J.R., Santos, I.R., Maher, D.T., Tait, D.R., Cyronak, T., Sadat-Noori, M., Macklin, P., Jeffrey, L.C., 2019. Groundwater as a source of dissolved organic matter to coastal waters: insights from radon and CDOM observations in 12 shallow coastal systems. Limnol. Oceanogr. 64, 182–196.

Wiesenburg, D.A., Guinasso Jr, N.L., 1979. Equilibrium solubilities of methane, carbon monoxide, and hydrogen in water and sea water. J. Chem. Eng. Data 24, 356–360.

Xiao, K., Wilson, A.M., Li, H., Ryan, C., 2019. Crab burrows as preferential flow conduits for groundwater flow and transport in salt marshes: a modeling study. Adv. Water Resour. 132, 103408.

Yau, Y.Y., Xin, P., Chen, X., Zhan, L., Call, M., Conrad, S.R., Sanders, C.J., Li, L., Du, J., Santos, I.R., 2022. Alkalinity export to the ocean is a major carbon sequestration mechanism in a macrotidal saltmarsh. Limnol. Oceanogr. 67, S158–S170.