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Challenges to select suitable habitats and demonstrate 'additionality' in Blue Carbon projects: A seagrass case study

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ABSTRACT

Seagrass restoration has been suggested as a Blue Carbon (BC) strategy for climate change mitigation. For Nationally Determined Contributions (NDC) and carbon crediting schemes, BC projects need to demonstrate 'additionality', that is enhanced CO_2 sequestration and/or avoided greenhouse gas emissions following management actions. This typically requires determining soil carbon accumulation rates (CAR), which is often done using radionuclides or surface elevation tables to estimate sedimentation rates. Here we undertook a case study, using ^{210}Pb and ^{14}C dating, to detect possible changes in C_{org} stocks and CAR following the loss and partial recovery of *Posidonia* seagrass meadows in South Australia since 1980–90s. The ^{210}Pb data revealed a lack of accumulation of excess ^{210}Pb in most sites, suggesting negligible accumulation of sediments, intense mixing of the upper layers, or accumulation of reworked sediments, precluding the estimation of reliable CAR at decadal time scales. This limitation was also encountered with ^{14}C. The inability to compare sites over analogous periods of time prevented quantifying differences in soil C_{org} sequestration, thereby to demonstrate additionality. The lack of significant differences in soil C_{org} stocks among sites which never suffered seagrass loss, those showing recovery and those with no recovery (5.7 ± 1.2, 4.5 ± 0.7 and 3.3 ± 0.3 kg C_{org} m^{-2} within the top meter, respectively) also precluded estimates of soil C_{org} gains or losses. Our findings demonstrate that, while ^{210}Pb and ^{14}C provide important information on sediment deposition dynamics, it is not straightforward to demonstrate additionality using radionuclides in low depositional seagrass habitats exposed to hydrodynamic energy, features which may be encountered in seagrass sites. We provide insights for the selection of suitable habitats for seagrass BC projects, suggest possible alternative methods for estimating additionality, and discuss the implications of the findings for the implementation of seagrass BC strategies to mitigate greenhouse gas emissions.

1. Introduction

Blue Carbon (BC) ecosystems are declining at a rate of ~0.5–3% per year (Hamilton and Casey, 2016; Pendleton et al., 2012), creating a risk of greenhouse gas (GHG) emissions through remineralisation of their soil organic carbon (C_{org}) stocks, in addition to the loss of their C_{org} sequestration capacity (Fourqurean et al., 2012; Lovelock et al., 2017). This offers potential for the inclusion of BC ecosystems in carbon-crediting programs, through activities that enhance CO_2 sequestration and/or avoid GHG emissions through their conservation, restoration or creation (Thomas, 2014). Despite the Intergovernmental Panel on Climate Change (IPCC) recommending inclusion of BC ecosystems in national GHG inventories (Hiraishi et al., 2013), BC programs have often been excluded from financing and climate change mitigation mechanisms (Wylie et al., 2016), partly due to the lack of standardized methods for assessing change in C_{org} stocks following activities, and because of the limited number of case studies demonstrating 'additionality' (i.e. a net gain in C_{org} stocks or sequestration from a project

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scenario that would not occur in a baseline scenario) (Needelman et al., 2018; Wylie et al., 2016). These knowledge gaps have hindered the application of existing methodologies to account for GHG removals and emission reductions, such as the Verified Carbon Standard (VCS) (Macreadie, 2019; Needelman et al., 2018).

Seagrass ecosystems accumulate most of their C_{org} in their soils (Serrano et al., 2019), so demonstrating additionality requires estimates of soil C_{org} stocks and accumulation rates before and after a BC activity. Typically, this relies on using radionuclides (i.e. ^{210}Pb and/or ^{14}C) (Marbà et al., 2015) or Surface Elevation Tables (SETs) to determine soil accumulation rates (Lovelock et al., 2013; Macreadie et al., 2017; Potouroglou et al., 2017). Radionuclide-based approaches allow retrospective estimation of accumulation rates more rapidly than the decadal timescales often required for SETs, and improve the understanding of sedimentation processes acting at a site (Arias-Ortiz et al., 2018a). However, the ability to determine CARs using radioisotopes can be limited by some sedimentation processes acting at sites, including sediment mixing, bioturbation and reworking of sediments, all processes which are likely to occur in seagrass ecosystems. On the other hand, SETs have been successfully used to estimate CAR in tidal marsh and mangrove ecosystems, and few recent studies shows their usefulness in seagrass ecosystems (Githaiga et al., 2019; Johnson et al., 2019; Potouroglou et al., 2017) but they typically require periods of years to establish reliable baseline estimates of sediment accumulation rates at reference sites, which may be beyond the timelines of carbon crediting projects.

Here we attempted to quantify the change in soil C_{org} stocks and CAR following seagrass loss in False Bay (South Australia) in 1980-90s, and the subsequent recovery of the meadows at some sites. This scenario provided an ideal case study to demonstrate carbon accumulation and, therefore, additionality, in the sort of setting that might form the basis for BC projects, that is the facilitated recovery or rehabilitation of disturbed seagrass areas (Kelleway et al., 2020).

2. Material and methods

Around 10 km^2 of *Posidonia* and *Amphibolis* meadows were lost in False Bay (South Australia) in the 1980–1990s due to eutrophication, while a further 10 km^2 were degraded (Harbison and Wiltshire, 1993). Since 1993, conservation actions resulted in a reduction of nutrient fluxes and, since early 2000, partial seagrass recovery (Wiltshire, 2014). In 2014, 12 seagrass soil cores (6.3 or 7 cm inner diameter, ranging from 132 to 173 cm long) were collected at 5 m depth in three different sites: three cores were sampled in 'Resilient' *Posidonia australis* meadows unaffected by the die-off; three cores in each of two different locations with 'Recovered' *P. australis* meadows since the die-off and three cores in 'Bare' but previously vegetated soils (Fig. 1). Inside and outside core lengths were measured during collection to correct for compression during coring (Glew et al., 2001; Smeaton et al., 2020).

All cores were sliced at 0.5 to 1 cm-thick intervals for the top 20 cm and at 1 to 4 cm-thick intervals for the remaining part ('high' and 'low' resolution cores, respectively). Every core slice was dried at 60 °C until constant weight to estimate Dry Bulk Density (DBD). Every second slice was analysed for C_{org} and δ^{13}C values of the soil organic matter in all cores as per Serrano et al., 2016b. Briefly, the sample was treated with 4% HCl to remove inorganic carbon, centrifuged and the supernatant with acid residues removed by pipette. The samples were then washed with Milli-Q water, centrifuged and the supernatant again removed. The residual samples were re-dried, encapsulated and analysed using an ECS 4010 Nitrogen/Protein Analyzer (Costech Elemental Analyzer) connected to an Isotope Ratio Mass Spectrometer (Thermo-Finnegan Delta V IRMS) at UH-Hilo Analytical Laboratory (University of Hawaii). The C_{org} content (%) was reported for the bulk (pre-acidified) samples. Soil grain size was analysed in alternate slices in one core per site (high resolution cores). Soil grain size was analysed in one core per site (Fig. 2). Samples were digested with 10% hydrogen peroxide to remove organic matter and then analysed using a Beckman-Coulter laser-diffraction particle analyzer at the University of Barcelona (Spain).

To determine short-term sediment accumulation rates (SAR) and CAR, ^{210}Pb concentrations were determined on the <125 μm soil fraction of two cores from Resilient site, two cores from the Recovered sites, and one core from a Bare site (Fig. 3) by alpha spectrometry (Sanchez-Cabeza et al., 1998). Supported ^{210}Pb (^{226}Ra) was analysed by ultra-low background liquid scintillation counting (Masque et al., 2002). The Constant Flux:Constant Sedimentation (CF:CS) rate model (Krishnaswamy et al., 1971) was used to estimate the average SAR for the last century, where possible.

Radiocarbon dating of shells was conducted in all cores (one to three samples per core; Supplementary Table S1) using a NEC Pelletron 500 kV at DirectAMS (Seattle, USA). Dates were calibrated with CALIB software v.7.1 and corrected for the marine reservoir effect (i.e. by subtracting 71 years to account for the depletion of radiocarbon in the oceans relative to the atmosphere (Bowman, 1985); Supplementary Table S1). Where possible, long-term SAR and CAR (i.e. based on

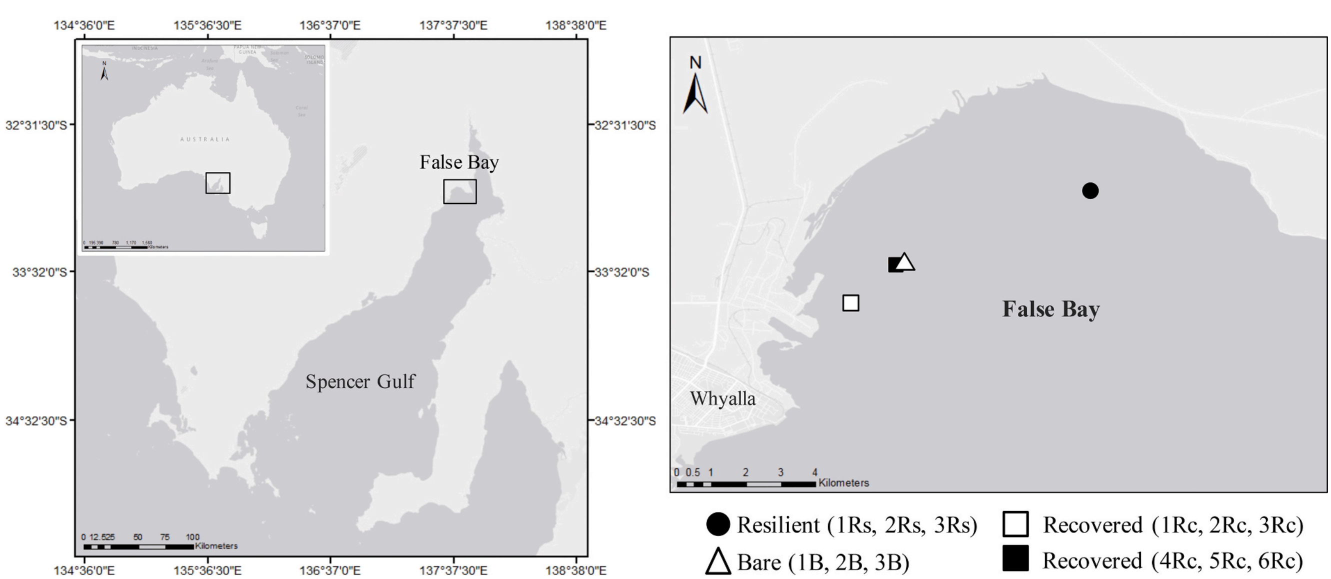

Fig. 1. Location of study sites in False Bay (North-West Spencer Gulf, Australia).

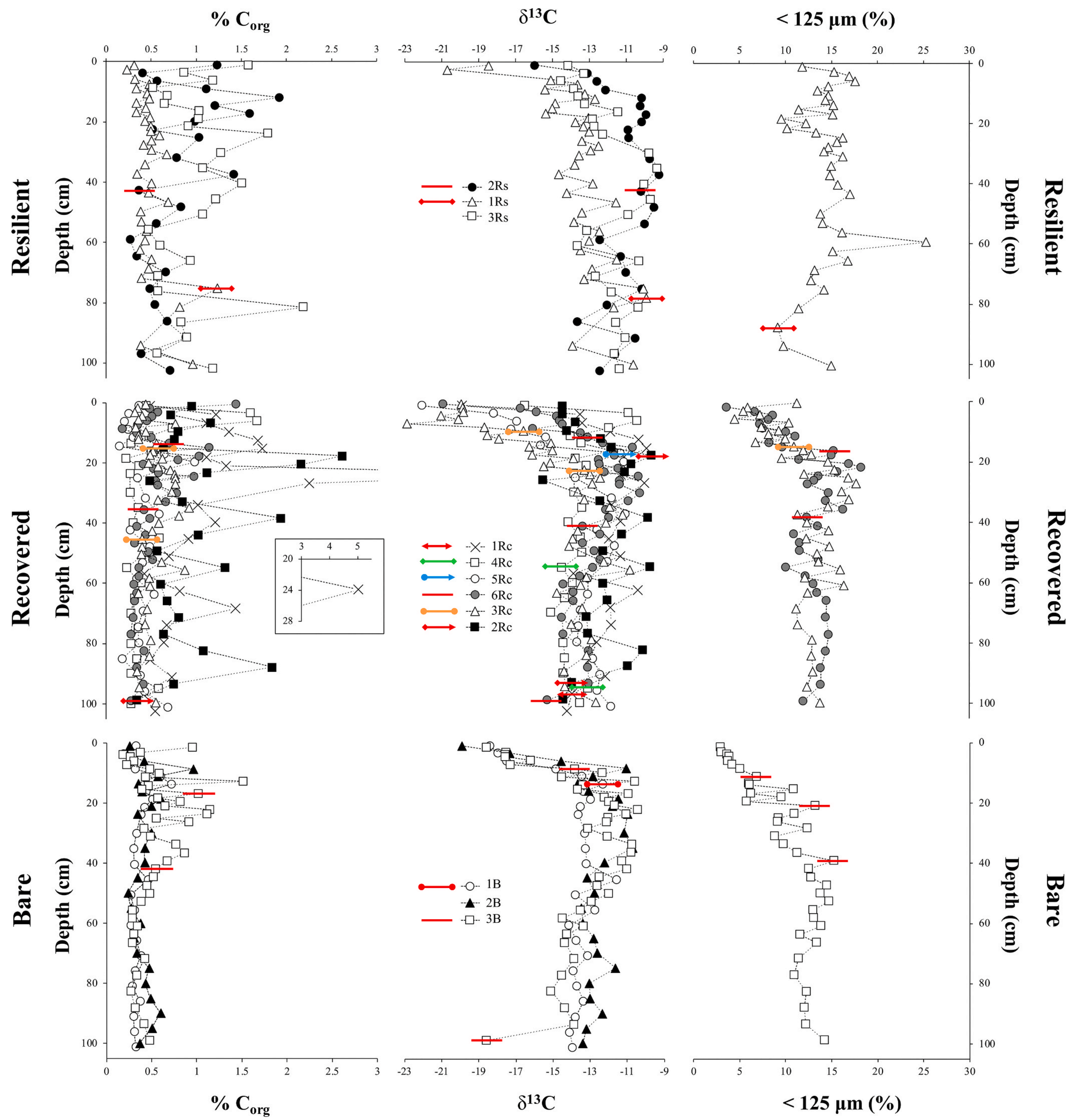

Fig. 2. Soil profiles of C_{org} (%), $\delta^{13}C$ values (‰) and % soil particles <125 μm in cores from Resilient and Recovered meadows, and Bare soil. The red horizontal lines indicate depths at which shifts were identified. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

radiocarbon results) were calculated using the Bacon R package (Blaauw and Christen, 2011).

Soil C_{org} stocks (kg C_{org} m^{-2}) and mean values (±SE) of DBD, $\%C_{org}$, $\delta^{13}C$ and <125 μm soil fraction were calculated for the top 20 and 100 cm-thick soil deposits (Table 1). CAR (g C_{org} m^{-2} yr^{-1}) were calculated by multiplying the C_{org} concentration by the mass accumulation rates (g m^{-2} yr^{-1}).

Significant shifts with soil depth in $\%C_{org}$, $\delta^{13}C$ and <125 μm soil fraction were explored using the Sequential Regime Shift Detection software (sig. level = 0.1; Fig. 2) (Rodionov, 2004). Because of our small sample sizes (n = 3, n = 6 and n = 3 for Resilient, Recovered and Bare, respectively), we applied a Kruskal–Wallis test (Table 1, Supplementary Table S2) to assess differences in DBD, $\%C_{org}$, $\delta^{13}C$ and soil C_{org} stocks (in 20- and 100 cm-thick soil deposits) among study sites. We could not test for differences in the <125 μm soil fraction because only one core per site was analysed for this variable.

The depositional environment at False Bay was characterized based on fetch and wave energy. Fetch (km) was calculated with 'fetchR' 2.1-1 (Seers, 2018) as the average fetch obtained for the wind rose quadrats corresponding to the dominant winds. The wave energy (kW/m) was

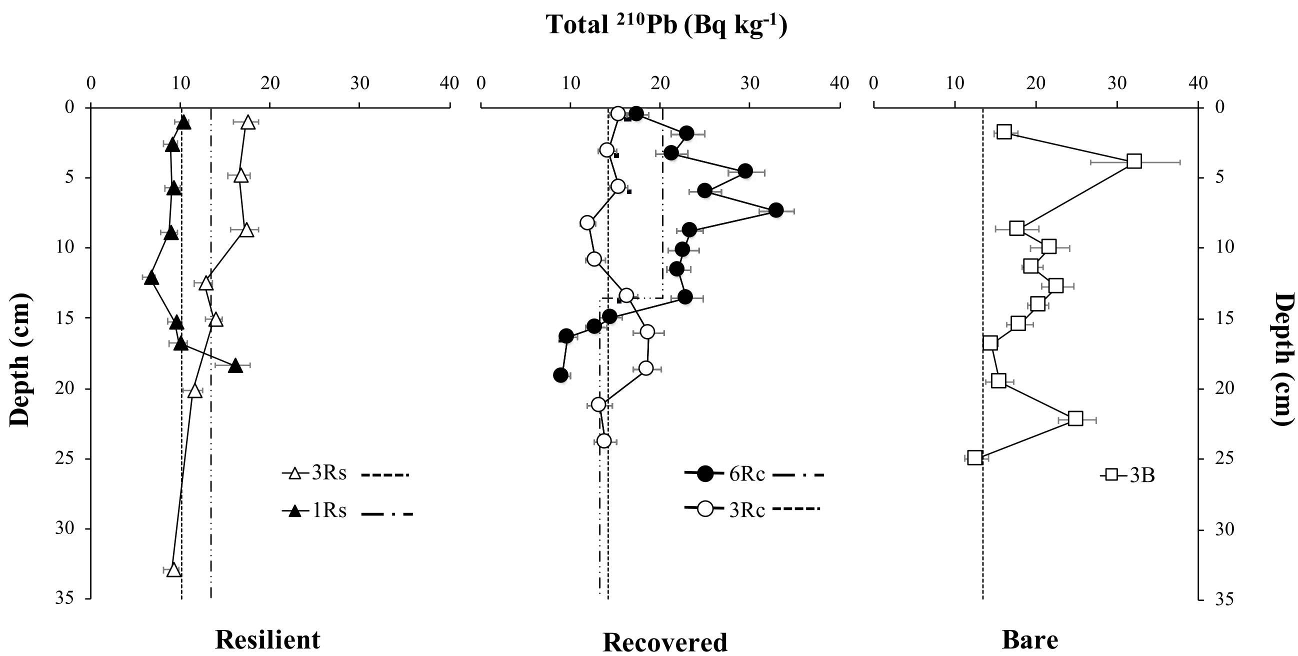

Fig. 3. Total ^{210}Pb concentration profiles for cores 3Rs and 1Rs (Resilient), 6Rc and 3Rc (Recovered) and 3B (Bare). Dotted lines indicate the supported ^{210}Pb for each core.

calculated with 'waver' (Marchand and Gill, 2018) using fetch, water depth and wind speed (Supplementary Table S3). Meteorological data were obtained from the Bureau of Meteorology (http://www.bom.gov.au/climate/data/).

3. Results

In all sites and cores, the Corg content was relatively stable with soil depth, and was typically less than 2% C_{org} (Fig. 2). In contrast, $\delta^{13}C$ remained constant throughout the sediment profile in the Resilient meadows (average -13.7 ± 0.4‰) but decreased within the top 20 cm of Recovered and Bare sites (averaging -14.4 ± 0.9‰ and -14.6 ± 0.3‰, respectively). Similarly, the proportion of particles <125 μm remained stable in Resilient meadows (average 14.0 ± 0.6%; Fig. 2, Table 1) but decreased within the top 20 cm of Recovered meadows and Bare soils (averaging 10.27 ± 0.12% and 5.5 ± 0.6%, respectively). The DBD, % C_{org} and $\delta^{13}C$ were not significantly different among meadows types for both 20 and 100 cm-thick soils ($p \geq 0.3$; Table 1, Supplementary Table S2). The C_{org} stocks in 20 and 100 cm-thick soils were not significantly different ($p = 0.8$ and $p = 0.3$; Supplementary Table S2) among study sites (ranging from 0.5 to 1.5 and 2.7–7.7 kg C_{org} m^{-2}, respectively; Table 1).

Only one core (3Rs; Resilient meadow) showed decreasing excess ^{210}Pb concentrations in the upper 15 cm, allowing estimation of SAR (0.14 ± 0.08 g cm^{-2} y^{-1} and 0.19 ± 0.10 cm yr^{-1}) and CAR (15 ± 8 g C_{org} m^{-2} yr^{-1}). For two cores (6Rc, Recovered meadow; 3B, Bare soils), the excess ^{210}Pb concentrations in the upper 10–15 cm were relatively constant, indicating mixing. Excess ^{210}Pb was absent in the other two cores (1Rs and 3Rc; Fig. 3). Radiocarbon analyses showed soils ranging from modern to pre-Holocene, and in some cores showed the presence of pre-Holocene aged sediments at relatively shallow soil depth (Supplementary Table S1). Long-term SAR and CAR for the last 3000 years were calculated only for 3Rc (0.0551 ± 0.0013 g cm^{-2} y^{-1} and 2.90 ± 0.07 g C_{org} m^{-2} yr^{-1}) and 6Rc (0.0339 ± 0.0013 g cm^{-2} y^{-1} and 1.53 ± 0.06 g C_{org} m^{-2} yr^{-1}) in Recovered meadows. For all the other cores, the presence of dating reversals precluded estimation of a reliable long-term SAR (Supplementary Table S1).

The average fetch calculated for the dominant wind (South) at False Bay was 67 km (21, 76.5 and 92 km for Resilient, Recovered and Bare, respectively) and the average wave energy was 0.9 kW m^{-1} (0.2, 1.1 and 1.4 kW m^{-1} for Resilient, Recovered and Bare, respectively; Supplementary Table S3) across sites (located within ~10 km^2). For comparison, the same parameters were calculated for other study sites within temperate Australia where it was possible to estimate SAR in *Posidonia* spp. using ^{210}Pb. These locations were Port Pirie (SA), a more sheltered seagrass site on the eastern side of Spencer Gulf (4 km and 0.02 kW m^{-1}), and Oyster Harbour, a sheltered seagrass site in Western Australia (2 km and 0.004 kW m^{-1}) (Supplementary Table S3).

4. Discussion

This study highlights potential methodological issues associated with demonstrating additionality in seagrass BC projects, and which are related to techniques commonly used in BC studies and/or to the environmental characteristics of our study site, which may be common to other seagrass sites. While there appeared to be differences in the C_{org} stocks of the Resilient, Recovered and Bare sites, these were not statistically significant despite the $\delta^{13}C$ values and <125 μm soil particle contents providing evidence of erosion of fine sediments and seagrass-derived C_{org}, which normally suggest loss of soil C_{org} stock. However, the ^{210}Pb concentration profiles and/or poor ^{14}C-chronologies prevented us from comparing CAR and soil C_{org} stocks over standardized ages (i.e. periods of accumulation). Furthermore, the inability to determine SAR and CAR in most sites precluded demonstrating additionality, a key requirement of carbon crediting schemes. This finding contrasts with other studies of seagrass systems where it was possible to demonstrate additionality (Greiner et al., 2013; Marbà et al., 2015;), and it is likely the differences in environmental settings, such as fetch and hydrodynamic conditions, which are explored below, that account for this difference.

The low excess ^{210}Pb concentrations (or lack of) at our site suggests very low accumulation of sediments over the last decades, likely because of the absence of fine sediment deposition or erosion, contrary to observations for other seagrass soils (Arias-Ortiz et al., 2018b; Greiner

Table 1
DBD (g cm^{-3}), %Corg, $d^{13}C$ values (‰), % soil particles <125 μm (Mean±SE) and soil C_{org} stocks (kg m^{-2}) in 20 and 100 cm-thick soil deposits in Resilient and Recovered meadows, and Bare soil.

Core section	Site	Core ID	DBD (g cm^{-3})	DBD Mean±SE	C_{org} (%)	C_{org} Mean±SE	$\delta^{13}C$ (‰)	$\delta^{13}C$ Mean±SE	<125 μm (%)	<125 μm Mean±SE	C_{org} stock kg m^{-2}	C_{org} stock Mean±SE
Top 20 cm soil	Resilient	1Rs	0.71 ± 0.02	0.66 ± 0.08	0.38 ± 0.02	0.81 ± 0.22	−15.1 ± 0.7	−13.4 ± 1.0	14.0 ± 0.6	14.0 ± 0.6	0.5	1.0 ± 0.3
		2Rs	0.50 ± 0.05		1.12 ± 0.18		−11.8 ± 0.7		–		1.0	
		3Rs	0.78 ± 0.04		0.93 ± 0.12		−13.4 ± 0.3		–		1.4	
	Recovered	1Rc	0.61 ± 0.03	0.69 ± 0.03	1.25 ± 0.14	0.78 ± 0.15	−12.7 ± 1.1	−14.4 ± 0.9	–	10.27 ± 0.12	1.5	1.0 ± 0.2
		2Rc	0.66 ± 0.03		1.2 ± 0.3		−12.7 ± 0.7		–		1.5	
		3Rc	0.78 ± 0.03		0.45 ± 0.03		−17.9 ± 0.6		9.2 ± 0.7		0.7	
		4Rc	0.72 ± 0.03		0.7 ± 0.2		−12.9 ± 0.7		–		1.0	
		5Rc	0.74 ± 0.02		0.39 ± 0.10		−16.0 ± 1.3		–		0.5	
		6Rc	0.65 ± 0.04		0.63 ± 0.09		−14.08 ± 0.67		11.4 ± 0.9		0.7	
	Bare	1B	0.75 ± 0.03	0.75 ± 0.03	0.42 ± 0.06	0.48 ± 0.04	−15.1 ± 0.9	−14.6 ± 0.3	–	5.5 ± 0.6	0.6	0.74 ± 0.06
		2B	0.81 ± 0.04		0.47 ± 0.08		−14.2 ± 1.1		–		0.8	
		3B	0.70 ± 0.03		0.57 ± 0.09		−14.5 ± 0.7		5.5 ± 0.6		0.8	
Top 100 cm soil	Resilient	1Rs	0.684 ± 0.012	0.72 ± 0.03	0.48 ± 0.03	0.76 ± 0.15	−13.5 ± 0.3	−12.2 ± 0.7	14.4 ± 0.5	14.4 ± 0.5	3.5	5.7 ± 1.2
		2Rs	0.69 ± 0.05		0.81 ± 0.09		−11.2 ± 0.3		–		5.8	
		3Rs	0.78 ± 0.03		1.00 ± 0.09		−12.0 ± 0.3		–		7.7	
	Recovered	1Rc	0.65 ± 0.02	0.73 ± 0.02	1.2 ± 0.2	0.69 ± 0.13	−12.3 ± 0.4	−13.4 ± 0.4	–	12.15 ± 0.12	7.0	4.5 ± 0.7
		2Rc	0.69 ± 0.03		1.03 ± 0.12		−12.3 ± 0.4		–		6.6	
		3Rc	0.751 ± 0.013		0.52 ± 0.02		−15.0 ± 0.5		12.0 ± 0.5		3.8	
		4Rc	0.78 ± 0.02		0.47 ± 0.07		−13.6 ± 0.3		–		3.4	
		5Rc	0.77 ± 0.02		0.41 ± 0.04		−13.7 ± 0.5		–		3.3	
		6Rc	0.72 ± 0.02		0.52 ± 0.04		−13.2 ± 0.3		12.3 ± 0.5		3.2	
	Bare	1B	0.78 ± 0.02	0.78 ± 0.02	0.36 ± 0.02	0.44 ± 0.05	−14.0 ± 0.3	−13.5 ± 0.3	–	9.8 ± 0.6	2.7	3.3 ± 0.3
		2B	0.82 ± 0.02		0.42 ± 0.03		−13.0 ± 0.4		–		3.4	
		3B	0.751 ± 0.014		0.54 ± 0.05		−13.5 ± 0.4		9.8 ± 0.6		3.6	

et al., 2013; Lafratta et al., 2019; Macreadie et al., 2012; Marbà et al., 2015, 2018; O. Serrano et al., 2016b). Similarly, our radiocarbon analyses revealed the presence of pre-Holocene sediments (>11,650 Cal yr BP) at relatively shallow depths (149–176 cm) in several cores (Supplementary Table S1) and SAR and CAR (only available for the Recovered meadows) below global and Australian averages (Duarte et al., 2013; Kelleway et al., 2020; Serrano et al., 2019) suggesting that erosion and/or low sediment accumulation rates may have characterized False Bay seagrass meadows in the past. The erosion or re-working of soils may have been facilitated by the seagrass die-off in False Bay, which resulted in unstable sediment easily mobilized by waves and currents (Harbison and Wiltshire, 1993). The trends in $\delta^{13}C$ values and fine sediments also indicate erosion at the sites (Fig. 2). Since the $\delta^{13}C$ values of *P. australis* range from -10 to -12‰, compared with -15 to -30‰ in seston, macroalgae and terrestrial vegetation (Svensson et al., 2007), the decrease in $\delta^{13}C$ values over the top 20 cm of Recovered meadows and Bare soil (from -12.9 ± 0.3 to -14.4 ± 0.9‰ and from -13.0 ± 0.3 to -14.6 ± 0.3‰, respectively; Fig. 2, Table 1) indicates a loss of seagrass carbon through erosion, lack of seagrass matter inputs or both (Oscar Serrano et al., 2016). On the contrary, the stable $\delta^{13}C$ values with soil depth in Resilient meadows is consistent with continuous seagrasses presence at the site over the period reconstructed. The decrease in <125 μm soil particles within the upper 20 cm of soils in Recovered and Bare sites similarly indicates erosion, resulting in soil 'sandification' (van Katwijk et al., 2010) following seagrass loss, while the relatively constant <125 μm soil particle content in Resilient sites suggests meadow stability over the period reconstructed.

Although erosion can cause significant losses of soil C_{org} stocks in seagrasses (Arias-Ortiz et al., 2018b; Macreadie et al., 2015; Marbà et al., 2015; O. Serrano et al., 2016b), no clear shifts in %C_{org} with soil depth were detected (Fig. 2) and there were no statistically significant differences in soil C_{org} stocks among sites (Table 1, Table S2). This may reflect the low proportion of fine particles at all sites (12.1 ± 0.9%; Table 1, Fig. 2), which usually have a high proportion of organic matter (Burdige, 2007), resulting in little influence of mud erosion on soil C_{org} stocks at our study site. For the last 10 years, the Recovered and Resilient

meadows have had similar shoot densities (10–220 m^{-2} and 20 to 150 m^{-2}, respectively) (Wiltshire, 2014), which could also have reduced the difference in C_{org} stocks between those sites. In fact, the accumulation of soil C_{org} in seagrasses relies, among other factors, on seagrass productivity and the burial of seagrass tissues, and on the complexity of the canopy that helps to reduce resuspension and increase SAR (Greiner et al., 2013; Mazarrasa et al., 2018). Therefore, we would have expected to see some difference in C_{org} stocks between, at least, Resilient and Bare sites. Finally, the low number of replicates, which may have limited the statistical power of the analyses, together with the high variability, may have hampered our ability to detect differences. A power analysis indicated that a sample size of 115 would be required to see a statistical difference among C_{org} stocks in the top 20 cm, the depth most affected by erosion (effect size = 0.339, power = 0.9). The above are plausible explanations for the lack of detectable differences in stocks among sites; even if significant erosion occurred, if the remaining 1 m of soil at a disturbed site contained comparable concentrations of C_{org} as at an undisturbed site, the C_{org} stocks in the top 1 m would appear similar, even though the disturbed site may have suffered a large erosional loss. Thus, losses of soil C_{org} from the disturbed meadows may still have occurred but our inability to date the soils due to erosional or lack of accumulation processes, prevents a comparison of stocks over comparable accumulation periods and thus the ability to reveal any loss.

It was not feasible to determine baseline conditions and to calculate avoided CO_2 emissions or enhanced C_{org} sequestration following seagrass revegetation (i.e. demonstrating additionality). In fact, to estimate avoided emissions we should assume that loss of C_{org} happened following disturbance and therefore that a significant difference between C_{org} stocks of different sites was found. We estimated the amount of soil C_{org} that could potentially have been eroded and remineralised following seagrass loss at False Bay. To do this, we calculated the difference in soil C_{org} stocks between Resilient and Bare sites (2.4 kg C_{org} m^{-2}; see Table 1) and scaled this up to the area of seagrass that has been lost (10 km^2). We then assumed that 25–100% of this C_{org} stock has been remineralised, based on the approach of Pendleton et al., (2012), to provide a range of possible remineralised C_{org} mass. Based on this, we estimated that seagrass loss resulted in the erosion of 6,000–24,000 tonnes of C_{org}, suggesting that conserving the lost area of seagrass would have avoided the emission of 22,000–88,000 tonnes of $CO_{2\text{-eq}}$. However, demonstrating this requires showing that following disturbances, emissions would occur; and we are unable to check this hypothesis because we cannot establish reliable chronologies and, therefore, cannot compare like-aged sections of soil.

Radionuclides have been used elsewhere to demonstrate additionality or CAR in other *Posidonia* meadows (Greiner et al., 2013; Macreadie et al., 2015; O. Serrano et al., 2016b; Serrano et al., 2019), including Port Pirie (Lafratta et al., 2019), located 30 km away, and Oyster Harbour (Marbà et al., 2015), an estuary in Western Australia. However, these sites had up to 34- and 225-fold lower average fetch and wave energy (Supplementary Table S3) than False Bay, a factor conductive of net accumulation of ^{210}Pb and sediments under calm conditions. Indeed, mixing at these sites was limited allowing SAR estimates based on ^{210}Pb profiles. The comparatively high fetch and wave energy across False Bay sites could explain the general lack of sediment accumulation, mixing and erosion showed by the ^{210}Pb profiles. This would indicate that while ^{210}Pb techniques are appropriate to estimate CAR and additionality at low-energy sites, they will be less useful at high-energy sites where accumulation of sediments is small and, in addition, mixing, erosion or reworking of sediments can be intense. Since establishing a baseline and demonstrating additionality are requirements of carbon verification schemes (Clean Energy Regulator, 2018), a project established at False Bay would likely be ineligible for crediting at this stage. Alternative methods such as SETs and marker horizons (Needelman et al., 2018) could be explored to allow direct assessment of SAR and CAR at annual and decadal time scales and compensate for some of the limitations of radioisotopes such as mixing and the accumulation of reworked sediments. Those methods have been used extensively in mangroves and tidal marshes (Cahoon et al., 2019; Krauss et al., 2014; Webb et al., 2013) but few studies have successfully used them in seagrasses (Githaiga et al., 2019; Johnson et al., 2019; Potouroglou et al., 2017). They also face practical constraints (Kairis and Rybczyk, 2010; Rumrill and Sowers, 2008), such as resuspension of sediments during measurement and possible scouring formation surrounding the poles, and further studies are required to demonstrate their feasibility in seagrass ecosystems. However, even if SETs were to permit SAR and CAR to be estimated over annual timescales, they may not reveal longer-term processes, such as periodic erosion, until they actually occur. Undertaking radioisotope studies in the pre-project phase would provide insights into the medium-term (decadal) processes occurring at a site and that might occur during a project's lifetime, for example episodic erosion events. There is value, therefore, in using both SETs and radioisotopes to gain insights into short- (annual) and medium-term (decadal) processes operating at a site. This may help to identify sites that have higher chances to successfully meet the requirements of a BC project.

Our findings suggest that seagrass BC projects should be preceded by an assessment of whether the site meets the requirement to demonstrate additionality and therefore whether restoration is worth considering in the context of BC projects. This study shows that it may not be possible to demonstrate additionality in low deposition and high hydrodynamic environments, highlighting the critical need for careful site and methods selection for BC projects. We conclude that depositional environments with high soil C_{org} content and low hydrodynamic energy are more likely to meet the requirements for successful BC projects, including demonstrating additionality and higher carbon abatement potential (Lovelock et al., 2017; Mazarrasa et al., 2018). Alternative methods to estimate additionality, such as SETs and marker horizons (Needelman et al., 2018), should be also explored to estimate CAR at annual and decadal scales, as they can be less sensitive to the factors that can prevent determination of CAR using radioisotopes, such as mixing and the accumulation of reworked sediments.

Authors contribution

P.L., O.S., S.G. and M.F. conceived the project and study design. P.L., O.S., S.G. and M.M. collected the samples. A.L., P.L., O.S. and P.M. interpreted the data. A.L. performed data analyses and drafted the manuscript. All authors contributed to writing and editing the manuscript.

Data accessibility

The dataset supporting this article is available online at https://ro.ecu.edu.au/datasets/38/

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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.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ocecoaman.2020.105295.

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