Marine Pollution Bulletin 226 (2026) 119296

Contents lists available at ScienceDirect

Marine Pollution Bulletin

journal homepage: www.elsevier.com/locate/marpolbul

Recovery timeframe is a critical component in preserving carbon stocks in disturbed tropical seagrass meadows

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ARTICLE INFO

Keywords:
Blue carbon$_1$
recovery$_2$
Remote sensing$_3$
Ecological resilience$_4$
seagrass$_5$

ABSTRACT

Seagrass meadows, though covering a small percentage of the ocean floor, play a critical role in marine carbon sequestration. However, the mechanisms behind this role remain uncertain due to challenges in mapping seagrass extent and flux, accounting for spatial variation in carbon storage, and incorporating site-specific histories. This study investigates the effects of short-term (≤10 years) seagrass loss on sediment carbon stocks within a large seagrass meadow in the Great Barrier Reef region. Using remote sensing data to construct a seagrass recovery timeline, combined with field sampling and assessments of ecological factors, we found higher sediment organic carbon in areas where seagrass coverage was retained throughout the study period. Carbon stock and sediment accumulation varied with Recovery stage and were influenced by ecological factors and the length of seagrass absence. Minimal differences in sedimentary carbon stocks were observed between retained areas and those recovering within 3–8 years, suggesting limited disturbance impacts on carbon storage when recovery occurs within this timeframe. However, significant differences in carbon stocks were observed between persistent meadow areas and sites where no seagrass recovery had occurred for ≥10 years. The study meadow contained an estimated 39,779 Mg C (to 1 m), with spatial variation linked to seagrass loss and recovery history, as well as ecological and environmental factors. Our findings offer context to the permanence of seagrass Blue Carbon during meadow loss and suggest sedimentary carbon stocks can largely persist in similar systems experiencing cycles of seagrass loss and recovery that occur over 3-to-8-year timeframes.

1. Introduction

Coastal wetlands, comprising mangroves, saltmarshes, seagrass meadows—collectively referred to as Blue Carbon habitats—function as significant carbon sinks, capable of storing and sequestering carbon at rates that exceed their losses through respiration and sediment export (Duarte et al., 2010; Duarte and Cebrian, 1996). Compared to the other Blue Carbon habitats, seagrass meadows occur at lower elevations in the tidal zone, generally along low-energy coastlines (Hansen and Reidenbach, 2012; Stevens and Lacy, 2012; Waycott et al., 2009). Seagrass meadows are also considered one of the more productive marine ecosystems, serving as a nursery habitat for marine organisms and as important feeding grounds for megafauna such as dugongs and sea turtles (Heck et al., 2003; Scott et al., 2018; Whitfield, 2017). Globally, seagrass meadows cover a mere 0.1% of the ocean floor but account for 11% of the total organic carbon sequestered in the open oceans, known as their carbon stock (Gattuso et al., 1998; Kennedy et al., 2010; NOAA, 2024). Despite their broad economic and ecological importance, seagrass meadows have declined in extent globally due to coastal development, flooding, and deteriorating water quality (Collier and Waycott, 2009; Dunic et al., 2021; Waycott et al., 2009). The potential of seagrass meadows to mitigate carbon emissions underscores an urgent need to develop a deeper understanding of their carbon sequestration mechanisms and to slow their rapid global decline.

Biotic and abiotic factors influence carbon storage in seagrass meadows (Lavery et al., 2013; Mazarrasa et al., 2021; Mazarrasa et al., 2018; Miyajima et al., 2015; Miyajima et al., 1998; Ricart et al., 2020; Ricart et al., 2015; Samper-Villarreal et al., 2016; Serrano et al., 2014). However, a limited understanding of their influence can lead to significant over- or under-estimations of stored carbon. These factors include,

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https://doi.org/10.1016/j.marpolbul.2026.119296
Received 13 May 2025; Received in revised form 17 December 2025; Accepted 13 January 2026
Available online 26 January 2026
0025-326X/

but are not limited to, differences in seagrass species, sediment composition, runoff, above and belowground plant biomass, geographical variability, and methodological inconsistencies, that often lead to inaccurate Blue Carbon assessments (Gacia and Duarte, 2001; Miyajima et al., 2015; Ricart et al., 2020; Ricart et al., 2015; Samper-Villarreal et al., 2016; Schimel et al., 1994; Serrano et al., 2014). Sedimentary carbon estimates from a single or limited number of sites are often used to represent carbon across entire meadows, overlooking spatial heterogeneity caused by local ecological drivers or the temporal stability of meadows (i.e. their short and long-term histories) (Oreska et al., 2017; Ricart et al., 2015). Adopting a tailored sampling approach that incorporates potentially influential ecological drivers of carbon sequestration, meadow histories, and applying advanced modeling techniques at both regional and global scales can improve our understanding of these complex systems and improve the reliability of Blue Carbon reporting.

Often a lack of detailed site-specific meadow histories has hindered accurate estimations of sedimentary carbon stocks and their fluxes post-disturbance. Neglecting meadow histories and lack of sedimentation information can lead to significant overestimations of carbon stocks, particularly in cases of prolonged (>10 years) or large-scale seagrass losses (i.e. loss of seagrass across an entire meadow) (Bijak et al., 2023; Dahl et al., 2025; Greiner et al., 2013; Macreadie et al., 2014; Marbà et al., 2015; Oreska et al., 2017). Long-term restoration projects have documented the erosion of sediment carbon stock following meadow loss and their gradual carbon stock recovery after restoration efforts (Greiner et al., 2013; Marbà et al., 2015). Modeling approaches have also shown that restoration actions can regenerate sediment carbon stocks, but substantial accumulation often becomes evident only after extended (~10 years) of active restoration (Duarte et al., 2023; Duarte et al., 2013; Greiner et al., 2013; Marbà et al., 2015; Oreska et al., 2017). Notably, the effects of short-term (3–5 years) losses of seagrass on sediment carbon stocks and subsequent recovery remain underexplored (Macreadie et al., 2014).

Our study aims to assess the effects of short-term (3–8 years) seagrass losses and subsequent recovery on carbon stocks within a large seagrass meadow in Far North Queensland (Australia). In particular, this study leverages the availability of high-resolution historical seagrass data, and spatial analysis techniques to build site-specific seagrass histories combined with meadow scale carbon assessments to generate accurate sediment carbon stock estimates along with sequestration rates and understand how history of seagrass loss and recovery can affect these carbon assessments. We integrate historical remote sensing data with field sampling of seagrass and sedimentary carbon to create a recovery timeline and identify which factors may influence carbon stocks within seagrass meadows in tropical Queensland.

Our findings offer insights into the Blue Carbon benefits of coastal restoration projects and small-scale carbon accounting efforts, presenting opportunities to enhance the credibility of future restoration initiatives and demonstrate the impacts of variable seagrass meadows on the permanence of carbon stocks.

2. Materials and methods

2.1. Site history

This study was conducted in a seagrass meadow in Cairns Harbour, Australia, with the study site extending from the Cairns Marina to Ellie Point, offshore from the Cairns Esplanade (Fig. 1A). Seagrass in the meadow is dominated by *Zostera muelleri* and also features *Halophila ovalis*, *Cymodocea serrulata* and *Halodule uninervis* and has been monitored annually since 2001 (McKenna et al., 2015). The meadow has a maximum mean depth below sea level of 2.26 m and covers approximately 334 ha. Over the past 15 years, the meadow has undergone significant variation in area, biomass and species composition due to natural disturbances (Reason et al., 2023). Along the Cairns Esplanade, minimal human activities affect seagrass health, with a thriving meadow observed pre-2008 (McKenna et al., 2015; Reason et al., 2024). Between 2008 and 2010, there were significant seagrass declines attributed to elevated rainfall, storm activity and cyclones, leading to nearly complete loss of the entire meadow (McKenna et al., 2015; Pollard and Greenway, 2013). Subsequent natural recovery began from a small remnant portion of the meadow in 2012 and gradually expanded thereafter over the following decade (2012−2023). Within the study site, a short-term loss of seagrass is defined as this period of ≤3 years during which bare sediment was present over most of an area of the intertidal meadow (>90%) and followed the natural recovery in the next few years. Given the relatively rapid, staggered, natural recovery of this site, and the presence of some areas with a permanent seagrass presence, along with some areas where seagrass remained absent for at least 10 years, this meadow presents an ideal study site for understanding the effects of short-term seagrass losses on sediment carbon storage.

2.2. Mapping seagrass recovery stages

We developed historical maps of the seagrass meadow extent using satellite and aerial imagery available from Queensland Globe, Planet, and NearMaps for the period 2008 to 2023, selecting for low-tide visuals (Lyons et al., 2013). Remote imagery highlighted definitive recovery boundaries as a patchwork within a former meadow boundary from 2008 to 2023. The annual dynamic of the seagrass meadow was mapped through a combined approach of imagery and biomass data collected in the field to delineate the meadow's border each year. Seagrass survey and biomass data have been collected through random sampling throughout the meadow since 2001 (Reason et al., 2024). Given the seasonal nature of tropical seagrass species and their shifting boundaries throughout the year, compounded by succession of species following the 2008–2010 seagrass loss, it was important to consider all imagery and biomass data when creating an annual boundary (McKenna et al., 2015). All imagery and biomass data taken from 2008 to 2010 was used to understand the gradual areal decline of the meadow. The meadow's recovery boundaries (beginning in 2011) were inclusive of all aggregated patches, visual presence via remote imagery, and in-situ biomass data starting from 2011, as this year began the switch from a declining meadow towards a naturally recovering meadow. Using visual image-interpretation techniques supported by false color images and image time-series, we digitized the full extent of seagrass and the Recovery stage of individual patches using QGIS and ArcGIS Pro.

After developing the annual time-series of seagrass meadow distributions, we classified each yearly map into a set of classes that describes its general Recovery stage based on when clusters of change occurred in short periods of time. Initial analysis of the meadow identified five distinct Recovery Groups: Persistent meadow (2012-onward), Early Recovery (2013–2014), Intermediate Recovery (2015–2017), Late Recovery (2018–2023), and No Recovery (2011−2023) (Supplementary Table S1).

2.3. Sediment cores and lab analysis

Sediment cores were collected within the meadow in October 2023 to assess sediment carbon across the seagrass Recovery Group areas (Fig. 1A). A total of 43 sediment cores were collected scattered randomly throughout the meadow but stratified to ensure replicate collections occurred within each of the Recovery Groups, as well as an even number of cores within bare sediment or in seagrass for all applicable groups. Six 10 cm cores were taken from the Persistent meadow, 7 from the Early Recovery, 10 from the Intermediate Recovery, 11 from the Late Recovery and 4 from area that has not recovered (Supplementary Table S1). An additional five 40 cm cores were taken in each Recovery Group. See Supplementary Materials Table S1 for a full breakdown of all Recovery Groups and core locations. All five 40 cm cores and 4 selected 10 cm cores were selected for ^{210}Pb analysis to estimate sediment and

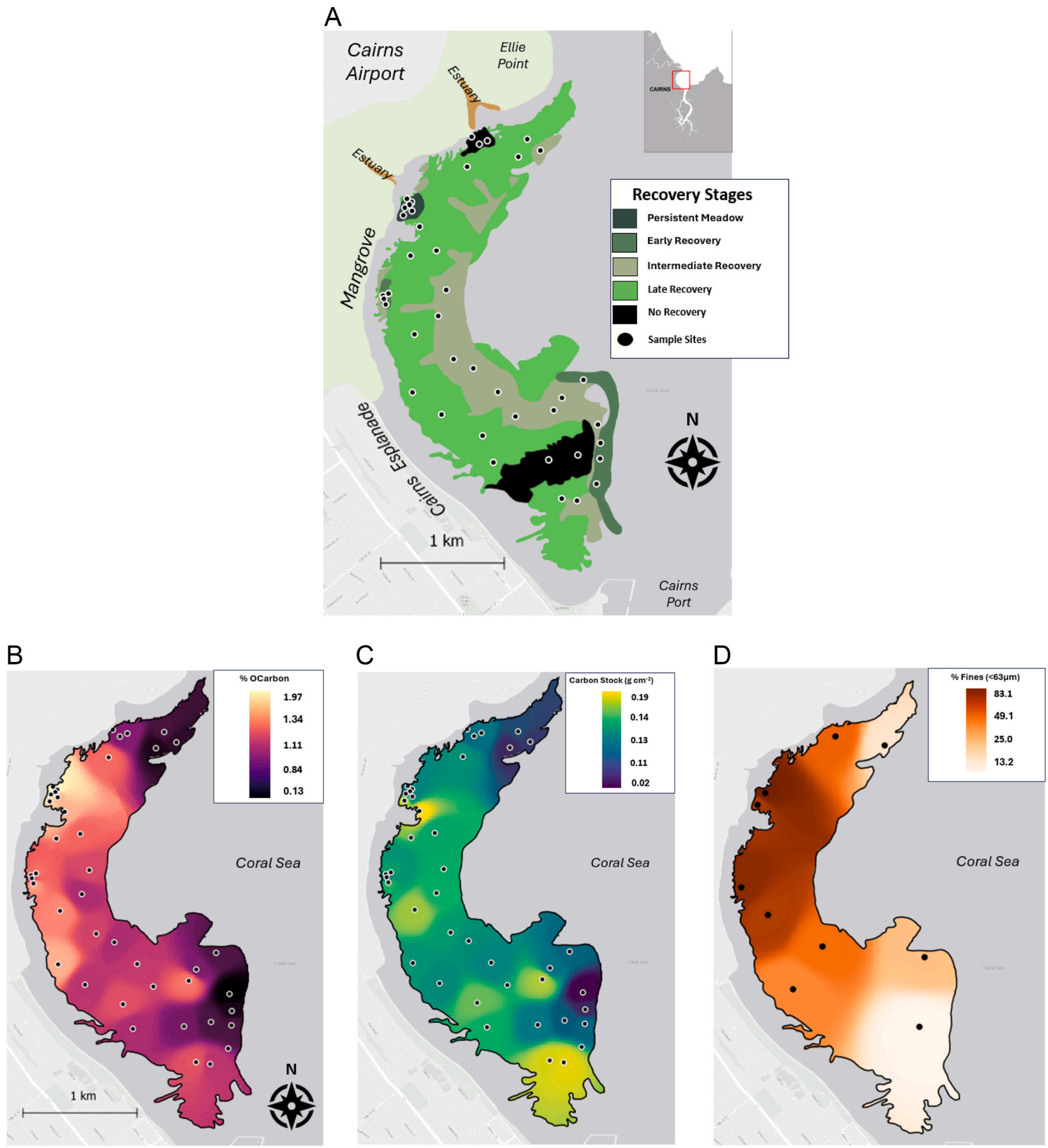

Fig. 1. Map of the seagrass meadow located off the Cairns Esplanade and the spatial distribution of carbon throughout the meadow. A) The study site located in Far north Queensland, Cairns (16.9203° S, 145.7710° E). The recovery stages were delineated using satellite and aerial imagery from 2008 to 2023. The Persistent meadow began recovery in 2012; the Early Recovery recovered from 2013 to 2014; the Intermediate Recovery recovered from 2015 to 2016; Late Recovery recovered from 2017 to 2023; and the No Recovery area has shown no recovery presence of seagrass as of the aerial mosaic obtained by TropWATER in 2023. There are 42 sample sites across the entire meadow. B) The distribution of %OC throughout the meadow. C) The distribution of C_{stock} throughout the meadow, some of the highest stocks near the southern tip of the meadow, close to the port, and near a estuary mouth. D) The distribution of %fines throughout the meadow. Maps A, B and D were created in ArcGIS Pro with the Inverse Density Weighted tool.

carbon accumulation rates across the meadow. All the cores were 5 cm in diameter. At each collection site, a GPS point was taken, seagrass (if present) was collected using 4–6 sweeps with a garden rake, and depth recorded.

Each 10 cm core was sliced into 1 cm sections and each 40 cm core was sliced at 1 cm sections for the upper 10 cm, 2 cm sections in the

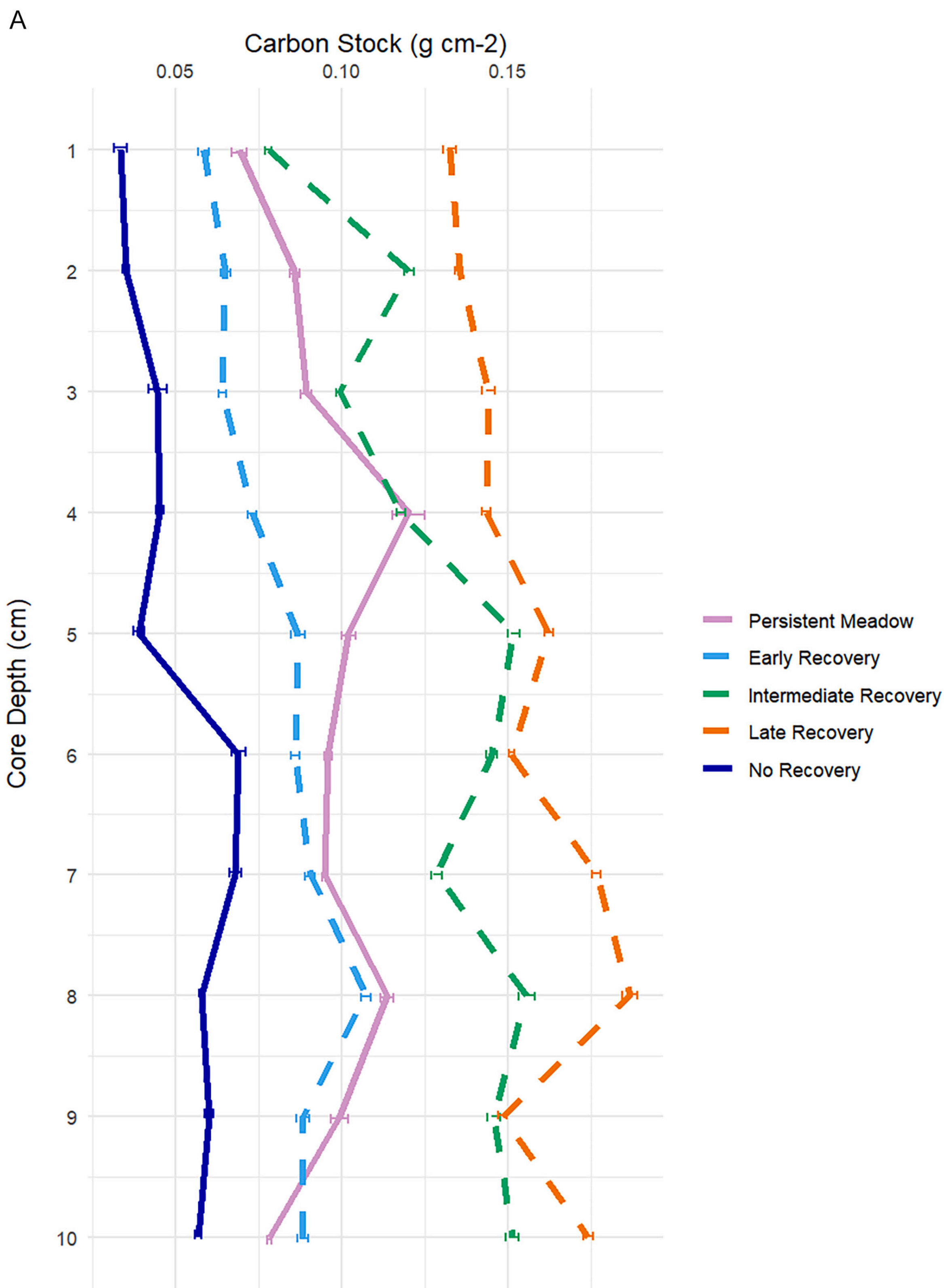

Fig. 2. Depth profiles for C_{stock} and %OC across recovery stages. A) The depth profile of $C_{stock,}$ values at each depth are summed for each Recovery Group and the standard error bar is constructed from the original values. B) The depth profile for %OC has values at each depth calculated by the average within each group. The Persistent meadow has a significantly higher %OC (Tukey test, $P < 0.05$).

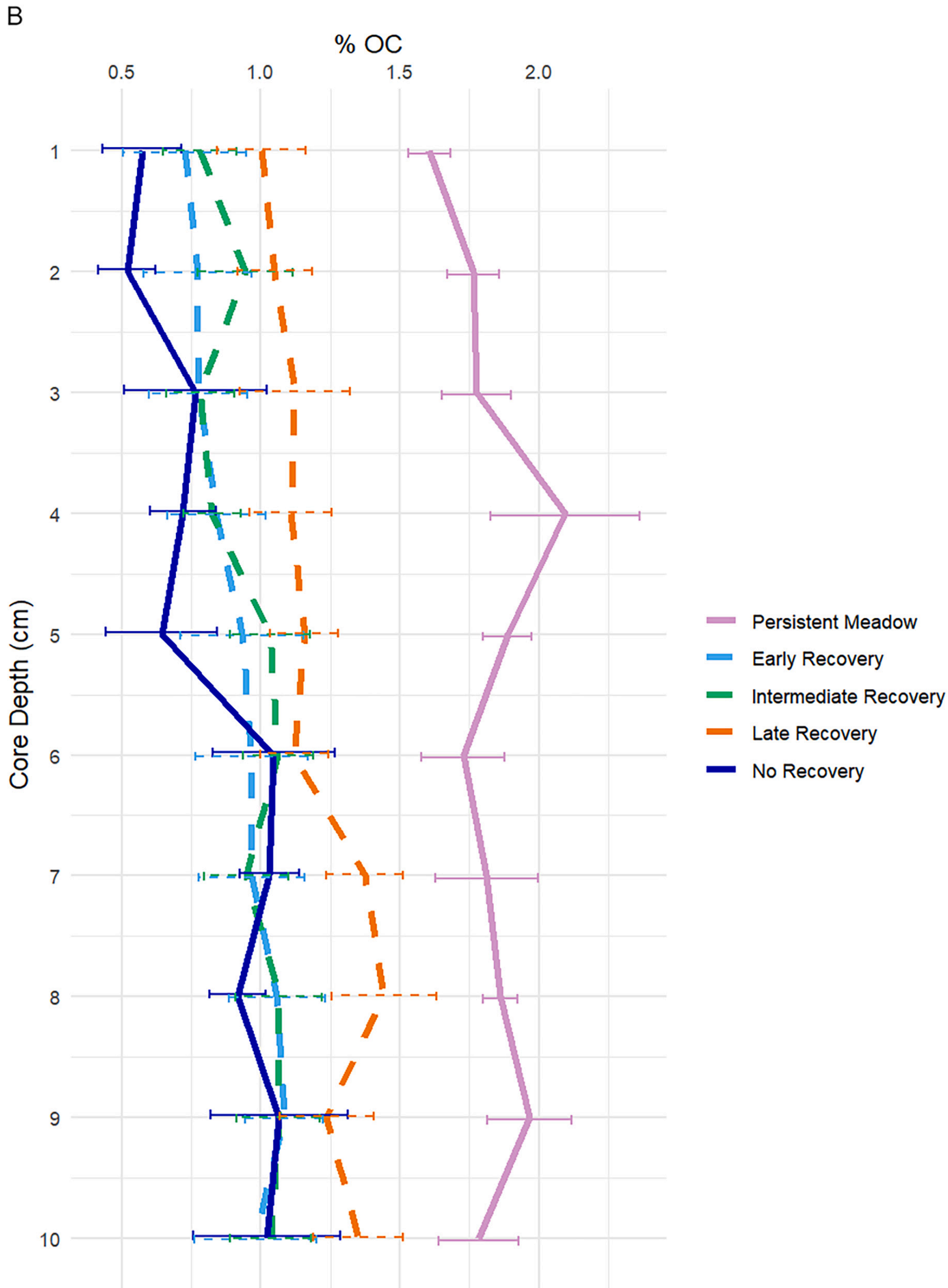

Fig. 2. (*continued*).

10–30 cm and 5 cm sections below 30 cm. Each sectioned sample weighed (wet weight), living plant matter removed, oven-dried at 60 °C for 4–9 days, and reweighed to determine bulk dry weight.

To determine carbon and nitrogen percentages, samples were cleared of any large organic/non-organic matter (>2 mm), ground, ~1 g of each sample was placed in a centrifuge tube and then acidified using HCl 2 N to remove carbonates. After overnight treatment, samples were vacuum sieved to remove acid and either freeze-dried or oven-dried to remove excess water. Each sample was weighed to calculate the percentage of inorganic carbon and reground. An Elemental Combustion System

CHNS-O (ECS 4010) was then utilized to determine %C and %N (Costech Analytical Technologies Inc., Valencia, CA, USA).

For all five 40 cm cores and four 10 cm cores, sediment accumulation rates were estimated from the ^{210}Pb concentrations profiles. The concentrations of ^{210}Pb were determined by measuring the activity of its ^{210}Po (in equilibrium) following Sanchez-Cabeza et al. (1998). Briefly, a known amount of ^{209}Po was added as a tracer to 300 mg sediment aliquots of each slice, which were digested in acid media using an analytical microwave. Following digestion, the polonium isotopes were plated onto silver disks and their emissions measured through alpha spectrometry. The ^{226}Ra concentrations were determined in selected samples along each core by gamma spectrometry via the quantification of its decay products ^{214}Pb and ^{214}Bi. Excess ^{210}Pb concentration profiles, calculated as the difference between total ^{210}Pb and supported ^{210}Pb (^{226}Ra), were used to determine the sedimentation rates over decadal time scales using the Constant Flux:Constant Sedimentation (CF:CS) and Constant Rate of Supply (CRS) models (Appleby and Oldfield, 1978; Arias-Ortiz et al., 2018; Krishnaswamy et al., 1971).

2.4. Environmental drivers

The influence of different ecological factors on carbon storage were analyzed by first computing a range of predictor variables thought to influence %OC and C_{stock} at the meadow-scale (Supplementary Table S2). To calculate the distance to mangrove, estuary, and port entrance, the collected GPS site locations for each core were utilized with the Distance to Nearest Hub tool in QGIS (QGIS 3.34.4) and aerial photographs (Google Earth, 2013). Similarly, sediment grain size analysis was conducted for each centimeter depth across ten 10 cm cores using a Mastersizer 3000+, where the proportion of the smallest sediment fraction <63 μm (i.e. clay and silt) was calculated for each sample, hereafter referred to as %fines. Tidal height at each site was calculated as the depth below mean sea level (dbMSL) for Cairns Harbour. Also, the position of each core (in seagrass or bare sediment) is considered for analysis, of which were stratified throughout each Recovery Group. For a full list of site variables and their specific statistics for each Recovery Group, refer to Supplementary Table S3.

2.5. Analysis

The percentage of organic carbon (%OC) and carbon stock (C_{stock}) (g cm^{-2}) were measured for each depth of all cores. Depth profiles for %OC were created by averaging the %OC at each depth within each seagrass Recovery Group, while the depth profile for C_{stock} was calculated as the sum of C_{stock} at each depth within each Recovery Group (Fig. 2). An ANOVA test was performed to evaluate overall means between Recovery Groups and for significant results a Tukey HSD test was performed for %OC and C_{stock} to better understand significant differences between Recovery Groups.

To assess the relative importance of seagrass Recovery Groups and explanatory variables influencing C_{stock} and %OC at the meadow-scale, we constructed two subsets of data. The first subset included all explanatory variables except %fines, while the second included all explanatory variables. The two subsets were analyzed separately because %fines measurements were only conducted on ten cores, compared to 42 cores for the full dataset. To approximate a normal distribution, both %OC and C_{stock} were square-root transformed. A total of four models were constructed (see Supplementary Materials Table S4 and S5 for a full breakdown of spatial and non-spatial models for each data subset).

For the first two models, we employed a spatial generalized linear mixed-effects model (GLMM) using the template model builder (sdmTMB) to evaluate spatial relationships among ecological drivers (e.g., distance to mangrove, estuary, port mouth, regrowth time, position, canopy height, tidal depth below mean sea level [MSL]) and their influence on %OC and C_{stock} across sectioned core depths. Regrowth time is a continuous variable that accounts for time seagrass has been present post-disturbance, verified using satellite imagery at each sample site (e.g., Persistent meadow = 12, No Recovery = 0, Supplementary Table S2). Preliminary analysis indicated significant spatial autocorrelation among explanatory variables (Moran's I, $P < 0.05$), justifying the use of spatial models. Due to high correlation among distance-based variables (i.e., distance to mangrove, estuary, and port mouth), only distance to mangrove was included to avoid multicollinearity. Depth was included as a fixed effect, with core ID as a random effect and other continuous and categorical covariates (distance to mangrove, position, canopy height, regrowth time, and tidal depth below MSL) included as covariates. Model convergence was verified through sdreport Hessian matrix (pdHess = TRUE) and inspection of fixed-effect gradients (all <1e-3).

For the second two models, linear mixed-effects model (lmer) were used to evaluate the linear relationships between ecological drivers (including %fines) and %OC or C_{stock} across sectioned core depths. Cores lacking %fines measurements were excluded. Moran's I tests indicated no spatial autocorrelation among explanatory variables (Moran's I, $P > 0.05$), making a standard linear mixed-effects model appropriate. Depth did not significantly contribute to response variable variance and was not included in these two models. Continuous covariates were standardized to facilitate effect size comparisons.

Model diagnostics included assessment of residual normality and homogeneity using visual diagnostic plots generated with 'see' and 'performance' packages. For the spatial models, residuals were further assessed for spatial correlation using Moran's I test (via the 'spdep' package)(Bivand, 2024; Lüdecke et al., 2024). All fixed-effect estimates were reported with standard errors, z-values, and *p*-values. Model fit was also evaluated using AIC comparison between spatial and non-spatial models.

To estimate meadow-wide carbon stocks for the top 10 cm of sediment, two methods were applied. First, carbon stocks were calculated from core samples, and these values were then used alongside Empirical Bayesian Kriging (EBK) predictive modeling to predict carbon stocks across remaining unmeasured areas of the meadow. This first method was employed because EBK predictive modeling considers spatial structures and sampled data across the meadow without having to make assumptions about each predictive variable's effect size and their overall influence on %OC and C_{stock}. This differs from normal kriging methods where semivariogram parameters are known and fixed. Since each predictive variable was not individually modelled within this study, but the general assessment of their influence on the response variables was considered instead, it did not seem appropriate to use these covariates as meadow-wide predictors of %OC and C_{stock}. Instead, the EBK model takes advantage of the robust in-situ sampled data to create meadow-wide sediment organic carbon estimates. The second method employed was using the Australian (AUD) National Average for carbon stock value for coastal seagrass meadows as a baseline. An Inverse Density Weighted model was created in ArcGIS Pro, with the number of neighbors concluded to be 3, to visualize the C_{stock}, %OC, and %fines, distribution throughout the meadow (Fig. 1B, C, D).

2.5.1. Empirical Bayesian Kriging (EBK)

The EBK model was developed using ArcGIS Pro, with the number of neighboring points (set to four) determined through visual analysis. Meadow-wide sedimentary organic carbon stock estimates derived from EBK (EBK-C_{stock}) were calculated using the following equation:

$$C_{stock} = c_i * area \quad (1)$$

where c_i is the average stock across the meadow i (Mg C/ha), and *area* (ha) is the total area of the meadow.

The associated standard error for EBK-C_{stock} was calculated using ArcGIS Pro. To directly compare EBK-C_{stock} with AUD-C_{stock}, the EBK estimates were extrapolated from 10 cm to 1 m depth and the 95% confidence interval was calculated.

2.5.2. Australian National Average (AUD-C_{stock})

The AUD-C_{stock} and 95% confidence interval were calculated using the baseline national average for re-established seagrass ecosystems (Kelleway et al., 2017). The sample mean value of 112.08 Mg C/ha was multiplied by the meadow's total areal extent using the equation:

$$AUD - C_{stock} = 112.08 \text{ Mg C/hectare}^{*} \textit{ area} \text{ (ha)} \tag{2}$$

3. Results

Across the meadow, %OC ranged from 0.13 to 1.97 and organic C_{stock} estimates within the top 10 cm ranged from 0.02 to 0.19 g cm^{-2} (Fig. 2; Fig. 1B, C). The lower measures of %OC and C_{stock} were found across No Recovery areas. Of the cores sampled, the %fines (the percentage of fine sediments <63 μm) ranged from 13.2% to 83.1% across the meadow, with lower %fines found near the southern and northern parts of the meadow (Fig. 1D).

Initial ANOVA results suggested a significant difference between % OC across Recovery Groups ($F_{(4,423)} = 37.93$, $P < 0.001$). There was a significantly higher %OC within the Persistent area of the meadow compared to the other Recovery Groups when a Tukey test was run (Fig. 1B) (Persistent-Early Recovery ($T = -10.3$, $P < 0.001$); Persistent-Intermediate Recovery ($T = -10.2$, $P < 0.001$); Persistent-Late Recovery ($T = -7.3$, $P < 0.001$); Persistent-No Recovery ($T = -9.7$, $P < 0.001$)). Within the Persistent meadow, %OC in the top 10 cm was fifteen times higher than sites marked as No Recovery and twice as high as all other Recovery Groups (Fig. 2B; Supplementary Table S3). When comparing the Persistent meadow C_{stock} (mean = 0.0139, standard deviation = 0.00592) to No Recovery (mean = 0.0102, standard deviation = 0.00417) there was a significant difference ($T = -3.7$, $P = 0.0019$). However, this was not the case when comparing the Persistent meadow to the other Recovery Groups (Early, Intermediate and Late) (Fig. 1C).

By analyzing the ^{210}Pb concentration profiles and noting the declining rate of excess ^{210}Pb, we can estimate sediment accumulation rates across Recovery Groups from the 40 cm cores. The ^{210}Pb concentration profiles revealed varying trends in sediment accumulation across Recovery Groups within the meadow (Supplementary Fig. S1). In the Persistent meadow, low excess ^{210}Pb was detected only in the upper 0–5 cm, while the Intermediate Recovery group exhibited a decreasing trend of excess ^{210}Pb from 0 to 10 cm for one of the 40 cm cores. The Late Recovery group displayed excess ^{210}Pb across 0–15 cm in the 40 cm core. The ^{210}Pb concentration profiles for the 40 cm cores allowed estimating upper limits for sediment accumulation rates of 0.50 ± 0.15 mm yr^{-1} (Persistent), 1.4 ± 0.2 mm yr^{-1} (Intermediate), and 1.7 ± 0.3 mm yr^{-1} (Late). In contrast, the No Recovery group showed no measurable excess ^{210}Pb, indicating negligible net fine sediment accumulation. The Early Recovery group also lacked detectable excess ^{210}Pb in the upper 0–8 cm, likely due to no net accumulation of sediments during the last decade, precluding an estimate of sediment accumulation rates (Supplementary Fig. S1). In the Intermediate and Late Recovery 40 cm sediment cores, the concentration profiles of excess ^{210}Pb suggest the occurrence of sediment mixing. The other four 10 cm cores analyzed and taken at different sites within the Intermediate and Late Recovery zones, had little to no detectable excess ^{210}Pb, suggesting no net accumulation of sediments and high levels of sediment mixing.

The spatial distribution of both C_{stock} and %OC exhibited considerable variation throughout the meadow (Fig. 1B, C), with persistent areas of the meadow demonstrated higher %OC levels. Grain size varied from the north to the south of the meadow and correlated significantly with % OC (Fig. 1D). The spatial variation in %fines was positively correlated with both %OC and C_{stock} across the meadow and across depths (Fig. 1D; Fig. 3).

We found significant spatial autocorrelation among sqrt(%OC) (Moran's $I = 0.747$, $P < 0.001$) and sqrt(C_{stock}) covariates (Moran's $I = 0.417$, $P < 0.001$, $n = 428$), justifying the use of spatial models. Two spatial models were fit, one for %OC and one for C_{stock}, with square-root transformed response variables and standardized predictor variables (Supplementary Table S4). Spatial model diagnostics indicated successful convergence (Hessian matrix positive definite, maximum gradient <1 e-3), and residuals were approximately normally distributed with no remaining spatial autocorrelation (Moran's I: %OC = -0.008, $P = 0.91$; $C_{stock} = -0.005$, $P = 0.74$). The spatial models also reported the spatial standard deviation (σ_O: %OC = 0.203, C_{stock} = 0.016), residual error variance (σ_E: %OC = 0.103, C_{stock} = 0.063), and

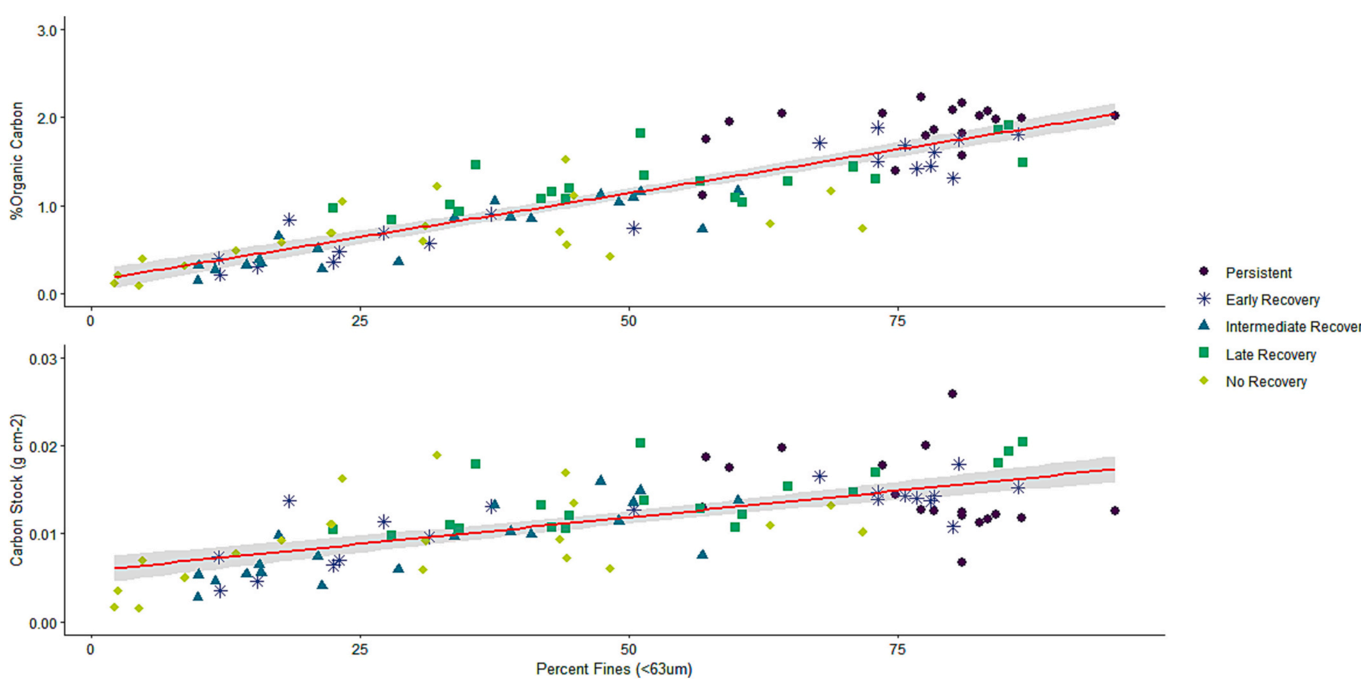

Fig. 3. Linear correlation plot of carbon vs. %fines. Linear correlation plots, using a linear regression model wherein the predictor variable is %fines, using all depths (0–40 cm) to visualize the positive relationship between %OC and C_{stock} with %fines, represented on the x-axis. All raw data from all %fines tested cores and across all depths ($n = 98$) is represented in both plots. The RMSE for %OC is 0.288 and is 0.00358 for C_{stock}.

Matern range (km) over which spatial correlation declined (%OC = 1.03 km, C_{stock} = 0.64 km), indicating that the spatial structure adequately captured autocorrelation in the data.

When compared to linear mixed models via AIC (%OC $AIC_{linear} = -186.65$; %OC $AIC_{spatial} = -313.03$; C_{stock} $AIC_{linear} = -2043.85$; C_{stock} $AIC_{spatial} = -2222.37$), the spatial models provided a substantially better fit. In the %OC spatial model, depths 3–10 cm were significant ecological drivers (Supplementary Table S4), while other predictors had small effect sizes and were not significant. In the C_{stock} spatial model, all predictor variables were highly significant but with small effect sizes. Confidence intervals for covariate effects may be conservative.

For the %fines subset (n = 98), no spatial autocorrelation was detected (%OC Moran's $I = -0.0733$, $P > 0.05$; C_{stock} Moran's $I = 0.0366$, $P > 0.05$), and linear mixed-effects models were therefore used. In these models, canopy height and dbMSL were slightly significant predictors ($P < 0.05$) (See Supplementary Table S5). For C_{stock}, regrowth time and position had a small but significant negative effect ($P < 0.05$). Across both linear models, %fines strongly positively influenced both C_{stock} ($lm_{estimate} = 0.021$, $P < 0.001$) and %OC ($lm_{estimate} = 0.23$, $P < 0.001$). Adjusted R^2 values indicated that these linear models explained a substantial proportion of variance for both C_{stock} ($R^2 = 0.57$) and %OC ($R^2 = 0.81$).

Following AUD methods, the carbon stock for the 2023 Cairns meadow is 37,446 Mg C with a 95% confidence interval of [34,980 Mg C, 39,908 Mg C]. This was compared to the EBK methods of carbon stock estimation which is 39,779 Mg C with a 95% confidence interval of [37,714 Mg C, 41,844 Mg C] (Table 1).

4. Discussion

4.1. Influence of seagrass loss and recovery on carbon storage

The history of seagrass loss and the mosaic of subsequent Recovery stages at our study site provided a unique opportunity to examine the impacts of various periods of seagrass loss on the sediment carbon storage capacity in a seagrass meadow. Our results show that for this meadow the C_{stock} for the top 10 cm remained similar between areas of the meadow where seagrass returned within 3–8 years and areas of the meadow where seagrasses remained throughout the study period. It wasn't until up to 10 years of seagrass absence that we recorded significantly lower carbon stocks. This has important implications for understanding the permanence of sedimentary organic carbon (SOC) stocks in these types of meadows and suggests that SOC stock can remain intact where seagrass losses are relatively short term and seagrasses can return within three to eight years. Many seagrass meadows around the world, and particularly those in the tropics, experience cycles of loss and recovery that occur on similar timeframes to our study meadow (Carter et al., 2023; Carter et al., 2022; Rasheed et al., 2014). For these types of meadows, the temporary loss of seagrass for up to 8 years may not result in large losses of carbon, especially under similar natural disturbance pressures to those in our study.

Persistent areas and late recovery areas consistently had higher C_{stock} across all sediment depths compared to areas with no seagrass recovery. While this pattern could suggest that at 10 years of seagrass loss cause the depletion of sediment carbon stocks deeper in the sediment profile, we cannot conclusively attribute these differences to temporal depletion, as site-level variability in baseline characteristics could also contribute. These results are generally consistent with other studies where C_{stock} increased steadily with recovery over 18 years after major disturbances (McGlathery et al., 2012), although shorter disturbances to seagrass may have less pronounced effects on sediment carbon stocks (Macreadie et al., 2014). This highlights that outcomes may differ depending on the nature and duration of disturbance and the recovery potential of a seagrass meadow. Large and prolonged disturbances are potentially necessary to substantially impact ecosystem services such as Blue Carbon (Marbà et al., 2015; McGlathery et al., 2012; Moksnes et al., 2021). In our study, the periods of seagrass loss were relatively short, and most recovery areas did not show substantial disturbances to sediments. Under these circumstances it appears that the SOC stocks can remain present with significant losses of sedimentary organic carbon only occurring once seagrass have been absent for more than 8 years.

Our results show that although the Persistent seagrass area had the highest percentage of organic carbon (%OC) sediment, this pattern did not correspond to higher carbon stocks (C_{stock}) across the meadow. Instead, C_{stock} tended to be higher in areas that recovered more recently. This mismatch arises because %OC reflects the composition of the sediment, whereas C_{stock} integrates both %OC and sediment mass per unit area. Thus, even though Persistent areas had higher %OC, they did not necessarily accumulate enough sediment or bulk density to produce higher C_{stock}. This indicates that regrowth or recovery time of seagrass alone does not explain carbon storage patterns. Rather, other environmental factors, such as sediment grain size, sedimentation and accumulation rates, and hydrodynamic conditions, likely interact with recovery processes to drive spatial variability in carbon storage observed across the meadow (Bijak et al., 2023; Mazarrasa et al., 2018; Samper-Villarreal et al., 2016).

Sediment grain size emerged as a key factor influencing %OC and C_{stock} in our study with a positive correlation between %fines and organic carbon content. Fine-grained sediments, such as silt and clay, offer more binding sites for organic matter, due to higher surface area, than coarser grains and are thus positively correlated to organic carbon content (Bergamaschi et al., 1997; De Falco et al., 2004; Keil and Hedges, 1993; Oreska et al., 2017; Serrano et al., 2016). The proximity of persistent patches of seagrass to mangroves and estuarine inputs likely contributed to high %OC through fine-sediment deposition and enhanced canopy trapping effects (Gacia et al., 1999; Samper-Villarreal et al., 2016). These areas generally had a higher % of fines than the other recovery areas.

Based on the Pb-210 profiles, the absence of net fine-sediment accumulation in areas of no recovery, despite proximity to an estuary mouth, underscores the importance of seagrass in sediment retention and accumulation. The absence of seagrass in this area for over a decade may have impeded suspended sediments—and associated allochthonous carbon—from settling (Gacia et al., 1999). Conversely, the areas of later recovery showed a 1.3-fold higher C_{stock} than the areas of no recovery, likely due to its proximity to sediment-rich sources (e.g., the port mouth). Since these areas of later recovery showed combined evidence of sediment mixing, likely due to seagrass absence for up to 8 years in some areas, as well as high rates of sediment accumulation, our Pb-210 analyses point towards rapid sediment retention and recovery of carbon stocks once seagrass is present and established. (Serrano et al., 2016). This suggests that even delayed seagrass recovery can facilitate sediment carbon sequestration.

Other ecological drivers such as canopy height, depth below mean sea level, time since seagrass regrowth occurred, and core location influenced %OC or C_{stock}. However, these patterns had inconsistent effects across models and likely linked to other unaccounted factors, such as seagrass species composition and small-scale current patterns, which

Table 1
For the Empirical Bayesian Kriging (EBK) methods, the carbon stock was determined for all 43 cores across the meadow by summing all depths (0–10 cm), and then using the EBK model through ArcGIS Pro (number of neighbors = 3), an average carbon stock was determined for the entire meadow and multiplied by 334 ha. The Australian National Average is determined by multiplying the national average 112.08 Mg C/ha by 334 ha.

Method	Meadow C_{stock} (Mg C)	95% Confidence Interval
Empirical Bayesian Kriging	39,779	±2064
Australian National Average	37,446	[34,980 Mg C; 39,908 Mg C]

add complexity to smaller carbon storage models (Kennedy et al., 2022; Zuur et al., 2009). The mixed effects and significance of environmental covariates in the linear models highlight the considerable complexities that influence carbon storage within seagrass meadows. The differences in seagrass assemblages between recovered areas for example is likely to be important to consider for species-specific suspended-particle trapping (Barcelona et al., 2021; Gacia and Duarte, 2001; Hansen and Reidenbach, 2012; Hendriks et al., 2008; Samper-Villarreal et al., 2016). Our study did not account for differences between species as obtaining such time-series species composition data in turbid tropical waters is challenging from both field studies and remote sensing (Duarte et al., 2010). With added species-specific seagrass monitoring as well as an understanding of local current patterns, Blue Carbon assessments would be able to account for the positive correlation between sediment accumulation through increased deposition and seagrass species with denser canopy heights (Gacia and Duarte, 2001; Samper-Villarreal et al., 2016). Addressing these ecological complexities in future studies is needed to refine carbon storage in regional and global baseline models.

We calculated similar total meadow organic carbon sediment stock estimates for the meadow using both the Empirical Bayesian Kriging (EBK) and Australian National Averages (AUD) method. While the EBK model produced slightly higher carbon stock values than the AUD method, the minimal difference between them suggests that both approaches adequately capture fine-scale carbon stock variability within this tropical seagrass meadow at this location when a robust number of samples can be taken within a meadow of interest (Kelleway et al., 2017). However, this consistency may not apply to other meadows where some studies only have the ability to take limited in-situ samples. We recommend further research comparing the Australian National Average with more detailed, site-specific measurements and modelled assessments to improve the accuracy and confidence of Blue Carbon stock estimates, particularly in dynamic tropical environments.

Our study emphasizes the importance of meadow-scale carbon estimates incorporating landscape configurations, historical baselines, and tailored modeling to capture spatial variability when robust in-situ sampling is unavailable (Bijak et al., 2023). In Australia, initiatives such as BlueCAM and the Carbon Farming Initiative signal a promising future for the economic benefits of Blue Carbon restoration projects (Angus, 2022; Kelleway et al., 2017). However, seagrass habitats require more research to align with these policies, given their monitoring challenges compared to mangroves and saltmarshes (Lovelock et al., 2022). Strengthening field sampling and modeling techniques to establish accurate landscape configurations and historical data will be essential to bridge this knowledge gap. The results for our meadow suggest that the permanence of Blue Carbon stocks may be more resilient to cyclic variability in seagrass presence and a potential pathway for these types of meadows to be included in restoration programs and funding mechanisms based on return of Blue Carbon.

4.2. Conclusions

Seagrass meadows in Cairns exhibit significant heterogeneity in sediment carbon storage due to the interplay of ecological drivers and the seagrass loss and recovery processes. Short-term seagrass losses (3–8 years of absence) did not lead to substantial sedimentary organic carbon depletion. In sites where seagrass recovery was delayed, carbon stocks still increased rapidly once regrowth began, suggesting that even delayed recovery of seagrass can have positive implications for fine-sediment retention and carbon recovery. These findings highlight the potential for natural recovery or early-onset restoration projects to protect seagrass Blue Carbon stocks by returning seagrass within 3–8 years of loss, at least for our study meadow. Improved understanding of ecological drivers, combined with historical baseline data, is desirable for accurate regional and global carbon stock estimates where concentrated sampling is unavailable. Integrating in-situ sampling, remote sensing, and modeling approaches will enhance the accuracy of carbon assessments, supporting restoration goals and policy alignment. Verifiable data on Blue Carbon habitats can help communities more accurately value coastal wetlands, refine restoration timelines and goals, and inform global conservation efforts.

CRediT authorship contribution statement

Taylor Renee Condron: Visualization, Validation, Methodology, Investigation, Data curation, Conceptualization, Writing – review & editing, Writing – original draft. **Timothy M. Smith:** Supervision, Resources, Methodology, Funding acquisition, Data curation, Conceptualization, Writing – review & editing. **Paul H. York:** Supervision, Methodology, Funding acquisition, Data curation, Conceptualization, Writing – review & editing. **Nicholas J. Murray:** Supervision, Methodology, Conceptualization, Writing – review & editing. **Pere Masque:** Validation, Data curation, Conceptualization, Writing – review & editing. **Michael A. Rasheed:** Supervision, Resources, Methodology, Investigation, Funding acquisition, Conceptualization, Writing – review & editing.

Funding sources

This work was financially supported by funding provided by BHP and the Australian Research Council Linkage Grant LP210300851. Seagrass monitoring was funded by Ports North.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests:

Timothy M. Smith, Paul York, and Michael Rasheed reports financial support was provided by BHP Group Ltd. Timothy M. Smith, Paul York, and Michael Rasheed reports financial support was provided by Australian Research Council. Timothy M. Smith, Paul York, and Michael Rasheed reports financial support and travel were provided by Ports North. If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgments

We would like to thank the TropWATER James Cook University team for all their help in the field, and specifically Rainy Comely, Stephanie Duce and Scott Smithers whose help within the lab, GIS analysis and sediment knowledge proved invaluable.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.marpolbul.2026.119296.

Data availability

Data will be made available on request.

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