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Management approach matters: meeting seagrass recovery and carbon mitigation goals



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Seagrass habitats support biodiversity, improve water quality, protect coastlines, and sequester carbon, among other essential ecosystem functions, yet they are declining worldwide due to human activity. Seagrass restoration and conservation can act as nature-based solutions for climate change, garnering growing interest from a diversity of stakeholders globally. Despite this interest, no seagrass projects have yet received carbon credits under international voluntary carbon standards. There is a clear need to better understand potential carbon mitigation outcomes of seagrass conservation and restoration practices. Here, we developed a mechanistic model based on a temperate meadow of *Zostera marina* to estimate carbon benefits (including net carbon dioxide removals, reductions, and methane and nitrous oxide fluxes) over 10 years as a result of four theoretical seagrass management scenarios, selected for their prevalence and potential—(1) restoration via seeding, (2) restoration via transplanting, (3) conserving a meadow and associated sediment from loss (e.g., from dredging), and (4) infilling an area with sediment prior to transplanting. We found significant differences and high variability in carbon benefits between these management scenarios. Restoration via transplant led to higher carbon gains than restoration via seeding, driven by more rapid areal bed expansion in transplanted meadows. However, the infill (adding 1 m of sediment) and conservation (preventing loss of 1 m of sediment) scenarios had total carbon benefits roughly 13–33 times (respectively) higher than the seeding and transplanting scenarios in which no sediment was mobilized. Within the model presented here, the minimum estimated revenue shows a 6 Ha seeding project generating as little as \$1189 over 10 years (39.6 ± 6.2 T CO₂eq), while a 100 Ha conservation project could generate a maximum of \$1.53 million over this same time period ($21,910 \pm 2196$ T CO₂eq), excluding costs for project implementation and MRV (monitoring, reporting and verification). Voluntary carbon credit revenue variability (ranging from \$198 to \$15,337 per Ha) is driven by project size, project approach, and carbon price, among other factors. This work highlights the need for careful, context-specific consideration for if and how carbon finance might support seagrass recovery goals. Seascape-level approaches that pair strategic sediment management and avoided emissions with habitat restoration may lead to the highest climate mitigation benefits, while simultaneously supporting biodiversity and other ecosystem functions.

Seagrass habitats are valued across the globe for a wide variety of ecosystem functions, acting as critical habitat for economically and ecologically valuable species, driving biogeochemical cycles, stabilizing sediments, and improving water quality¹. In recent decades, seagrass habitats have gained additional attention for their potential to sequester carbon over decadal to

centennial time scales and ameliorate ocean acidification over short timescales^{2–4}. Nonetheless, seagrass habitat is declining worldwide, largely due to human activity^{5,6}. This loss of habitat and its myriad documented ecosystem functions is apparent in many coastal ecosystems, leading to the United Nations' call for a “Decade on Restoration”^{7,8}. There is particular

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interest in deploying restoration efforts that can act as Nature-based Solutions (NbS), defined as “actions to protect, conserve, restore, sustainably use and manage natural or modified ecosystems which address social, economic and environmental challenges effectively and adaptively, while simultaneously providing human well-being, ecosystem services, resilience and biodiversity benefits”⁹. The long-term carbon storage potential has focused attention from a diversity of global stakeholders on the conservation and restoration of seagrass meadows as a Nature-based Solution for climate change mitigation^{10,11}.

As with other habitats with high potential carbon value, such as forests and tidal marshes, national and sub-national governments across the world are now working to include seagrass meadows in greenhouse gas inventories, Nationally Determined Contributions, and other climate and net-zero targets^{12–14}. Simultaneously, achieving habitat recovery goals (e.g., the UN Decade on Ecosystem Restoration, 30 × 30 initiatives) and the related recovery of ecosystem functions will require significant financial investment^{9,15–17}. Developing carbon credit markets have spurred further public and private interest, and may facilitate investment in seagrass restoration and conservation. Specifically, if NbS projects can demonstrate credible, long-term (“permanent”) carbon gains from their restoration or conservation strategies, they can earn carbon credits through a number of verified issuers and brokers. These credits can then be traded or retired to offset emissions, with high-integrity NbS projects demonstrating that they don’t lead to delayed phase-out of fossil fuels^{11,18}.

Despite potential support from carbon markets, financing from credit schemes could fail to cover the full costs of seagrass projects, depending on factors such as project scale and location, carbon price, or whether carbon credits are bundled with other financing or crediting mechanisms (e.g., biodiversity credits)^{19,20}. Nonetheless, seagrass restoration is already underway across the globe, and carbon finance could broaden the opportunities and advance seagrass recovery timelines (acknowledging the need for proper project implementation and verification within any carbon finance project). Formal verification processes for seagrass meadows have not been widely used, with only one globally accepted voluntary market seagrass methodology (Verra’s Verified Carbon Standard VM0033 methodology)^{21,22}. Despite their carbon potential, only one seagrass restoration project is undergoing verification under this standard to our knowledge (Virginia Coast Reserve, U.S.), leaving considerable uncertainty around the financial support carbon credits may offer.

The lack of formal accreditation is in part due to high uncertainty associated with evaluating carbon gains in naturally variable environments, the small relative size of many of these projects, and the numerous biogeochemical factors that projects must consider²³. VM0033 identifies carbon sequestered in living seagrass biomass, sediment carbon sequestration, emission of methane and nitrous oxide, mineral protection of carbon, inorganic carbon, soil carbon fate following erosion, and expected impacts from environmental change^{21,22}. Each of these factors also needs to be considered relative to a baseline, or business-as-usual site, in order to demonstrate that carbon was gained due to project implementation. These rigorous requirements and complex biogeochemistry can make direct evaluation of emissions reductions and removals challenging and costly in any habitat type, and are particularly lacking in seagrass meadows. As such, there is only one existing comprehensive estimate of the carbon gained from seagrass restoration available in the peer-reviewed literature¹⁹ from the Virginia Coast Reserve Long Term Ecological Research site (hereafter “Virginia LTER”).

Given the lack of accredited seagrass carbon projects, it can be an immense challenge to estimate or evaluate how carbon offsets might change over time and with different project approaches. For example, if a project proponent wanted to offset emissions via a seagrass NbS project, what approach would be best if the goal was to maximize carbon sequestration? In many cases, conservation may better achieve climate goals, however, these opportunities may not always be available. In this case, which restoration approach would be best (e.g., seeding, transplanting, passive improvement of conditions)? To date, no existing comparative work evaluates how various seagrass NbS project approaches affect carbon reduction potential over

time, despite the clear need and the fact that these approaches can lead to variation in carbon sequestration that may span orders of magnitude^{19,22,24}. Such work would inform selection of project approaches, provide foundational information to advance seagrass restoration and protection within the voluntary market, and support national and subnational climate models that require an understanding of how land use change and management practices affect underlying carbon sequestration (e.g.,^{14,25}). To address this need, we offer a mechanistic model to explore carbon offsets over 10 years associated with four theoretical seagrass NbS project approaches, selected for their prevalence and potential—(1) seeding, (2) transplanting, (3) conserving a meadow and associated sediment that would have been otherwise lost, and (4) applying a layer of sediment prior to transplanting.

Methods

Site description

The model presented here primarily applies data from the Virginia LTER, although it could be reparameterized using inputs from similar meadows elsewhere (Table 4). The Virginia LTER hosts large meadows of eelgrass (*Zostera marina*), which is the dominant seagrass species within the site and across temperate North America and Europe²⁵. After the region experienced high meadow losses around 1933, the Virginia LTER now supports one of the longest standing seagrass restoration programs², and has high data availability on carbon reduction associated with the restoration^{19,26–28}. Specifically, this site was the only site (globally) where we could find data on all four key modeled parameters (sediment carbon accumulation, carbon in biomass, methane emissions, and nitrous oxide emissions) in a restored seagrass meadow relative to pre-restoration or unvegetated sites nearby (Supplementary Fig. 2). These data acted as a starting point for model scenarios to represent more realistic outcomes specific to this region. However, we also vary these key parameters in subsequent model simulations based on available literature to infer how carbon outcomes might change as a result (e.g., using a global average seagrass sediment accumulation rate rather than values from the LTER; see Table 5).

Modeled seagrass management scenarios

The majority of existing seagrass restoration has occurred via spreading seed or directly transplanting shoots^{2,29–32}. There are also high potential carbon offsets from projects that preserve a meadow from dredging or re-using dredge sediment because marine, soft-bottom sediments (both vegetated and unvegetated sediments) can store high densities of organic carbon^{33,34}. As a result, protecting a seagrass meadow that would have been subjected to sediment loss from activities such as dredging, bottom trawling or erosion can result in significant carbon additionality^{35–37}. Similarly, making use of dredged sediments for restoration projects when sediment carbon would have otherwise been remineralized can also lead to carbon additionality, although the appropriateness of this restoration approach will be project and site specific³⁸. Dredged sediments have been used to elevate the benthos to suitable seagrass depths in past restoration projects^{39,40}, however, no literature exists documenting this process and its subsequent effect on carbon gains to our knowledge.

We use values from existing literature, when available, to model the carbon potential of these four approaches to seagrass management (See Tables 1 and 4). To determine total carbon offsets, each management scenario subtracts the carbon gained in the business-as-usual site from the carbon gained in the managed site²². Through this exercise we gain a broader understanding of how offsets may differ between project approaches, while simultaneously identifying key knowledge gaps that impede our current ability to make these estimates.

To understand how varying approaches to seagrass management may affect relative net carbon sequestration, we develop a mechanistic model to simulate four commonly applied seagrass management “scenarios” (Table 2). Specifically, we model annual carbon offsets over one decade in the following four scenarios: (1) *Seeding*: a project that restores seagrass via seed dispersal; (2) *Transplant*: a project that restores seagrass via direct transplant of adult shoots; (3) *Infill*: a project that restores seagrass via

Table 1 | Four seagrass management scenarios are modeled, which include restoration from seed (S), transplant (T), from transplant follow placement of a sediment layer (I), and conservation (C), in which in the absence of protections, we assume the meadow and underlying sediments are lost entirely

Management Scenarios	Description and Assumptions
1: Seed (S)	Management: Seeding occurs within a small area of the project site in year 0, and the population expands according to a logistic growth curve to a maximum area of 6 ha. We assume seed project area expansion occurs more slowly than transplant project area (Supplementary Fig. 1). BAU: No seeding occurs—the site remains unvegetated and unchanged throughout the 10 year monitoring period, accumulating sediment at naturally occurring rates.
2: Transplant (T)	Management: Transplanting occurs within a small area of the project site in year 0 and expands according to a logistic growth curve to a maximum area of 6 ha. We assume transplant project area expansion occurs more quickly than transplant project area (Supplementary Fig. 1). BAU: No transplant occurs—the site remains unvegetated and unchanged throughout the 10 year monitoring period, accumulating sediment at naturally occurring rates.
3: Infill (I)	Management: A layer of sediment (“infill”) is placed evenly across an unvegetated site (to varying depths, see Table 5), across the entire project area (6 ha) in year 0, to make conditions more suitable for seagrass restoration. This infilled area is transplanted according to the same method above 1 year later (year 1) - transplanting a smaller area that expands according to logistic growth to the maximum project area by year 10 (Supplementary Fig. 1). BAU: No infill or transplant occurs—the site remains unvegetated and unchanged throughout the 10 year monitoring period, accumulating sediment at naturally occurring rates.
4: Conservation (C)	Management: Conservation measures are implemented in year 0 such that a healthy, existing (6 ha) seagrass meadow remains unchanged throughout the 10 year monitoring period (when it otherwise would have been dredged). BAU: A seagrass meadow and underlying sediments are lost (e.g., from dredging) across the entire meadow area (6 ha) in year 0, to varying sediment depths (see Table 5). In addition to loss in year 0, this site behaves as an unvegetated site would (as described in the BAU sites in all other scenarios).

Two “sites” are modeled in each scenario—a management site in which the described scenario occurs, and a BAU (business-as-usual) site in which no management intervention occurs.

Table 2 | Defined model terms are defined for each Scenario and Site

Scenario	Site	Term
Sediment		
Transplant/Seed	C _{BAU}	$A_{\text{veg}} * \rho_{\text{Sed,unveg}}$
Transplant/Seed	C _{MGMT}	$A_{\text{veg}} * \rho_{\text{Sed,veg}}$
Infill	C _{BAU}	$A_{\text{infill}} * \rho_{\text{Sed,unveg}}$
Infill	C _{MGMT}	$\{t = 0, (A_{\text{infill}} * \rho_{\text{Sed,unveg}}) + (A_{\text{infill}, t=0} * \delta_{\text{Sed,unveg}} * \rho_{\text{remin}} * d_{\text{infill layer}})\}$ $\{t > 0, (A_{\text{infill}} * \rho_{\text{Sed,veg}})\}$
Conserve	C _{BAU}	$\{t = 0, (A_{\text{conserve}} * \rho_{\text{Sed,unveg}}) + (A_{\text{conserve}} * \delta_{\text{Sed,veg}} * d_{\text{dredge}})\}$ $\{t > 0, (A_{\text{conserve}} * \rho_{\text{Sed,unveg}})\}$
Conserve	C _{MGMT}	$A_{\text{conserve}} * \rho_{\text{Sed,veg}}$
Methane & Nitrous Oxide		
Transplant/Seed	C _{BAU}	$A_{\text{veg}} * \rho_{\text{Gas,unveg}}$
Transplant/Seed	C _{MGMT}	$A_{\text{veg}} * \rho_{\text{Gas,veg}}$
Infill	C _{BAU}	$A_{\text{infill}} * \rho_{\text{Gas,unveg}}$
Infill	C _{MGMT}	$\{t = 0, A_{\text{infill}} * \rho_{\text{Gas,infill}}\}$ $\{t > 0, A_{\text{infill}} * \rho_{\text{Gas,veg}}\}$
Conserve	C _{BAU}	$\{t = 0, A_{\text{conserve}} * \rho_{\text{Gas,dredge}}\}$ $\{t > 0, A_{\text{conserve}} * \rho_{\text{Gas,unveg}}\}$
Conserve	C _{MGMT}	$A_{\text{conserve}} * \rho_{\text{Gas,veg}}$
Biomass		
Transplant/Seed	C _{BAU}	$A_{\text{veg}} * \rho_{\text{Biomass,unveg}}$
Transplant/Seed	C _{MGMT}	$\{t = 0, A_{\text{veg}} * \rho_{\text{Biomass,veg}}\}$ $\{t > 0, (A_{\text{veg},t} - A_{\text{Transplant}, t-1}) * \rho_{\text{Biomass,veg}}\}$
Infill	C _{BAU}	$A_{\text{infill}} * \rho_{\text{Biomass,unveg}}$
Infill	C _{MGMT}	$\{t = 0, A_{\text{infill}} * \rho_{\text{Biomass,unveg}}\}$ $\{t = 1, A_{\text{infill}} * \rho_{\text{Biomass,veg}}\}$ $\{t > 1, (A_{\text{infill},t} - A_{\text{infill}, t-1}) * \rho_{\text{Biomass,veg}}\}$
Conserve	C _{BAU}	$\{t = 0, A_{\text{conserve}} * \rho_{\text{Biomass,veg}}\}$ $\{t > 0, A_{\text{conserve}} * \rho_{\text{Biomass,unveg}}\}$
Conserve	C _{MGMT}	$A_{\text{conserve}} * \rho_{\text{Biomass,unveg}}$

Net carbon benefits are determined by subtracting BAU outcomes from MGMT outcomes. Variable definitions and parameters are defined in Tables 3 and 4, respectively.

transplant after using sediment infill to alter benthic conditions prior to restoration; and (4) *Conservation*: a project that conserves a seagrass meadow and associated sediment that would have been otherwise lost (e.g., due to dredging) (Table 1).

Model assumptions

Based on standard carbon accounting principles, a project qualifying for carbon offsets must demonstrate additionality (i.e., that additional carbon was sequestered as a result of implementing the management approach). Therefore, each of the four seagrass management scenarios is also paired with a business-as-usual (“BAU”) baseline scenario, in which no management approach is taken. This is implemented by including two sites in each of the four scenarios, simulating the carbon gained from a “managed” site (seed, transplant, infill, or conservation) and a “BAU” site of equal size. In both the management and BAU sites, we assume there is no disturbance or extenuating circumstances occurring during the project period. For the BAU sites that accompany the first three treatments (Table 1), we assume that no restoration or sediment infill occurs within the boundary. For the BAU site in the fourth “conservation” scenario, we assume that no seagrass protection measures are implemented, and dredging occurs once at project onset instead. We assume BAU sites experience normal background seagrass growth or sediment accumulation rates, including after dredging/sediment loss, depending on the scenario. Key assumptions for each management treatment are defined in Table 1.

Mechanistic modeling approach

In order to estimate the total carbon benefits from each of the four seagrass management scenarios, we assume that all projects occur over 10 years, with the management action taken upon project initiation (year = 0). We assume each project restores or conserves the same area of seagrass (six hectares). The first three restoration scenarios reach full size at the 10 year mark, while the “conservation” scenario starts at full size and remains as such over time (i.e., a 6 ha meadow is conserved at year 0, and remains 6 ha over the project duration). Importantly, we assume that in the restoration scenarios, after transplant or seeding occurs, seagrass within the project boundaries expands according to a simple logarithmic growth curve, reaching its asymptote (6 ha) after 10 years (Supplementary Fig. 1; e.g., refs. 41,42). In seeding scenarios, we assume that the time required for seedlings to grow and propagate will lead to a slightly slower early meadow expansion rate, with a transplanted meadow reaching its midpoint after 4.3 years, while a seeded

Table 3 | Variable definitions of model inputs

Variable	Definition
<i>A</i>	Defined area (m ²) of the vegetated (<i>A</i> _{veg}), unvegetated (<i>A</i> _{unveg}), infill (<i>A</i> _{infill}), and conservation (<i>A</i> _{conserve}) project scenarios.
<i>δ</i>	Carbon density (g C m ⁻³) in vegetated (<i>δ</i> _{veg}) and unvegetated (<i>δ</i> _{unveg}) sediments.
<i>d</i>	Depth (m) of sediment infill layer (<i>d</i> _{infill}) or dredge depth (<i>d</i> _{dredge}).
<i>r_{remin}</i>	Proportion (%) of carbon in the dredged sediment that is remineralized during sediment movement and placement processes.
<i>ρ</i>	Carbon accumulation in living and dead above and below ground biomass (<i>ρ</i> _{biomass} in g C m ⁻²), vegetated sediments (<i>ρ</i> _{sed,veg} in g C m ⁻² yr ⁻¹), and unvegetated sediments (<i>ρ</i> _{sed,unveg} in g C m ⁻² yr ⁻¹). Methane fluxes (g m ⁻² yr ⁻¹) in vegetated (<i>ρ</i> _{methane,veg}), infill (<i>ρ</i> _{methane,infill}), dredge (<i>ρ</i> _{methane,dredge}) and BAU (<i>ρ</i> _{methane,unveg}) sediments. Nitrous oxide fluxes (g m ⁻² yr ⁻¹) in vegetated (<i>ρ</i> _{N2O,veg}), infill (<i>ρ</i> _{N2O,infill}), dredge (<i>ρ</i> _{N2O,dredge}) and BAU (<i>ρ</i> _{N2O,unveg}) sediments.
<i>C</i>	Total carbon gain or loss (g C _{eq} yr ⁻¹).

For comparison, when input parameter units were not reported in g C, they were converted to g C_{eq} using the 100-year GWP (UNFCCC, 2024).

meadow reaches its midpoint after 5.5 years (Supplementary Fig. 1). This general meadow expansion seeding curve is supported by McGlathery et al.⁴¹, which shows slow expansion in 4 years following transplant, followed by a precipitous increase. We assume logarithmic growth with a maximum areal extent, given that seagrass restoration projects will typically reach a maximum size based on depth limitations for continued seagrass expansion, or eventual blending with neighboring, naturally occurring meadows (Supplementary Fig. 1). Here, we model bed expansion according to the following equation:

$$\text{Area}_{\text{proj}} = \text{Area}_{\text{max}} / (1 + e^{\wedge}(\text{midpoint} - Y_T)) \quad (1)$$

where *Area*_{proj} is the total area at time *T*, *Area*_{max} is the total area for potential restoration, midpoint is the year at which half of *Area*_{max} is expected to be restored, and *Y_T* is the *T*-th year of the project. Within each of the four seagrass management scenarios, we include four key carbon parameters to estimate net carbon benefits, following methodologies laid out by Oreska et al.¹⁹ and the VCS VM0033²¹. These four parameters include carbon from sediment, biomass (in above and below ground live and dead biomass), methane emissions, and nitrous oxide (N₂O) emissions (Table 4). We exclude CO₂ produced from CaCO₃ production, although recognize that in some regions or meadows, this may not always be negligible (see section 4). Broadly, “carbon benefits” accrued in both sites in each scenario are defined as net carbon sequestered in sediment and biomass (reported in terms of equivalent quantity of carbon removed from the atmosphere), and the net changes in annual emissions of methane and N₂O aggregated as carbon equivalents (C_{eq}) using the 100-year Global Warming Potential (GWP100⁴³). To determine net benefit, we subtract BAU site carbon outcomes from those in the paired MGMT site (*C*_{net} = *C*_{MGMT} − *C*_{BAU}; Table 2). We define the following model inputs (Table 2) for each of these components (see Table 3 for variable definitions).

Recall, in scenarios 3 and 4, the infill and dredging activities themselves take place only upon project commencement (*t* = 0) and then each site proceeds as a typical transplant site or unvegetated site, respectively (Table 1). As such, model terms containing “infill” and “dredge” in the equations above will be zero for all but the 1st year of carbon evaluations.

Rather than applying the mean value of each of these parameters, the model instead selects each input parameter from a normal distribution generated around the mean and standard deviation of each parameter. The model runs iteratively 60 times, drawing 60 values from each distribution. Therefore, modeled outputs reflect variation in parameter estimates as documented in the literature. All analyses were performed in R (version 4.4.2)⁴⁴.

Model parameter selection and simulations

To our knowledge, only one seagrass meadow exists where all four of these key parameters are robustly estimated with paired measurements to use as BAU sites taken before the restoration or within neighboring unvegetated areas. Specifically, the Virginia LTER hosts meadows of *Z. marina* where

restoration has occurred alongside significant carbon measurements over the past two decades^{19,27}. As such, we use these data to parameterize the base model, making the modeled carbon offsets loosely constrained by values typical of temperate meadows of *Z. marina*. Specifically, we apply the following values (Table 4) to the “base” model simulation, which result from meadows within this reserve. This reserve is located on the eastern shore of Virginia, and is characterized by bays and inlets with shallow depths, relatively good water quality, and tidal ranges of ~1.25 m (see McGlathery et al.⁴¹; Supplementary Fig. 2).

In addition to modeling carbon offsets across four separate management treatments parameterized with Virginia LTER values (Table 1), we also run four model simulations with altered input parameters to evaluate how potential carbon offsets might vary with these realistic site and project specific changes (Table 5). If not specified, all other parameters from the “Base” model simulation are applied (Table 4).

Results

Carbon gains over time

When the base scenario (B) is modeled over the course of 10 years, N₂O and CH₄ emissions increase steadily as the seagrass meadows expands, while they remain relatively constant in the conservation scenario (where the meadow remains at its full size over 10 years) (Fig. 1A, B). Annual carbon gains in biomass during peak meadow expansion (year 5) are comparable to emissions from CH₄ and N₂O, but taper off as meadow expansion slows and the meadow reaches its full size after 10 years. The largest carbon gains are associated with soil. The conservation scenario leads to large net gains in soil carbon in the 1st year, given that in the absence of a conservation intervention (i.e., the BAU scenario), we assume that the meadow (and underlying meter of sediment sediment) are lost, leading to significant loss of organic carbon. This is seen in reverse with an “infill” project where sediment is added prior to transplant leading to significant carbon gains. Carbon gains in the “infill” scenario are lower than those in the “conservation” scenario despite an equal layer of sediment loss or gain (respectively) because we assume that 50% of the organic carbon in applied infill sediment will be remineralized during the movement and placement of the sediment layer.

When summed across years, modeled results show significant differences in carbon outcomes between approaches (scenarios). Specifically, transplanting leads to carbon gains roughly ~2.5 T C_{eq} higher than those from seeding, driven by more rapid areal bed expansion in transplanted meadows (“B” simulation; Table 6). However, “Infill” and “Dredge” scenarios have total carbon offsets roughly 13 to 33 times (respectively) higher than the seeding and transplanting scenarios in which no sediment is mobilized (“B” simulation; Table 6).

Model simulations

Modifying key parameters from the “Base” model simulation enables evaluation of how carbon gains may change under various management scenarios or environmental conditions. Sediment again largely drives carbon gains, yielding large differences in net carbon benefits when altering key

Table 4 | Model parameter inputs, which define the “Base (B)” model scenario

Parameter	Variable ID	Habitat Type	Mean	Error (SE)	Units	Data Source
Sediment (accumulation)	$\rho_{\text{Sed,veg}}$	Vegetated	36.68	2.79	$\text{g C}_{\text{eq}} \text{m}^{-2} \text{yr}^{-1}$	Greiner et al. ⁴⁵
Sediment (accumulation)	$\rho_{\text{Sed,unveg}}$	Unvegetated	0	0	$\text{g C}_{\text{eq}} \text{m}^{-2} \text{yr}^{-1}$	Greiner et al. ⁴⁵
¹ Sediment (stock)	$\delta_{\text{Sed,veg}}$	Vegetated	5578	454	$\text{g C}_{\text{eq}} \text{m}^{-3}$	McGlathery et al. ⁴¹
¹ Sediment (stock)	$\delta_{\text{Sed,unveg}}$	Unvegetated	2774	404	$\text{g C}_{\text{eq}} \text{m}^{-3}$	McGlathery et al. ⁴¹
Biomass	$\rho_{\text{Biomass,veg}}$	Vegetated	49.4	12.21	$\text{g C}_{\text{eq}} \text{m}^{-2}$	McGlathery et al. ⁴¹
Biomass	$\rho_{\text{Biomass,unveg}}$	Unvegetated	0	0	$\text{g C}_{\text{eq}} \text{m}^{-2}$	McGlathery et al. ⁴¹
Methane	$\rho_{\text{CH}_4,\text{veg}}$	Vegetated	4.58	3.03	$\text{g C}_{\text{eq}} \text{m}^{-2} \text{yr}^{-1}$	Oreska et al. ¹⁹
Methane	$\rho_{\text{CH}_4,\text{unveg}}$	Unvegetated	0.57	0.40	$\text{g C}_{\text{eq}} \text{m}^{-2} \text{yr}^{-1}$	Oreska et al. ¹⁹
N ₂ O	$\rho_{\text{N}_2\text{O},\text{veg}}$	Vegetated	5.07	3.38	$\text{g C}_{\text{eq}} \text{m}^{-2} \text{yr}^{-1}$	Oreska et al. ¹⁹
N ₂ O	$\rho_{\text{N}_2\text{O},\text{unveg}}$	Unvegetated	1.69	0.85	$\text{g C}_{\text{eq}} \text{m}^{-2} \text{yr}^{-1}$	Oreska et al. ¹⁹
Dredge depth	d_{dredge}	Vegetated	−1	NA	m	Van Maren et al. ⁹⁰ ; Howarth and Stewart. ⁹¹
Sediment layer depth	d_{infill}	Unvegetated	1	NA	m	Merkel ⁵⁶ ; Flindt et al. ⁴⁰
Remineralization (%)	r_{remin}	Unvegetated	50	NA	%	De Borger et al. ⁹² ; Graca et al. ⁹³

All parameters were all collected from within the Virginia LTER in beds of restored *Z. marina*. Biomass data include above and below ground live and dead biomass. 1: Sediment stock data result from the top 5 cm of sediment, extrapolated to 1 m (reflective of stocks from a meadow that may have persisted through time in the region).

Table 5 | Summary of the five modeled scenarios, aimed at evaluating potential variation in carbon offsets based on changing key model input parameters

Model simulation name	Input parameter changes	Justification and references
Base (B)	None (See Table 4)	See Table 4
Zero Emissions (ZE)	$C_{\text{Methane}} \text{ \& } C_{\text{N}_2\text{O}} = 0$ We assume no difference in CH ₄ and N ₂ O emissions between the vegetated and unvegetated sites.	CH ₄ and N ₂ O emissions often fall below the 5% <i>de minimis</i> threshold ²² . Asplund et al. reports no significant difference in emissions between vegetated and unvegetated sites ⁹⁴ .
High Sediment Accumulation (HSA)	$\rho_{\text{veg}} = 138 \pm 38 \text{ g C m}^{-2} \text{yr}^{-1}$	Global mean seagrass sediment accumulation rate ⁹⁵ .
Low Infill Depth (LID)	$d_{\text{infill}} = 0.1 \text{ m}$ (apply a thinner layer of sediment addition)	Some projects (e.g., “thin layer placement” or “sand capping” projects) place a thin layer of sediment prior to restoration ⁴⁰ .
Fishery Dredge Depth (FDD)	$d_{\text{dredge}} = 0.1 \text{ m}$ (dredge to 0.1 m, rather than 1 m)	Some fisheries (e.g., scallop) dredge shallower than maintenance dredging projects (e.g., navigational channel) ⁹¹ .

The “Base (B)” scenario is defined in Table 4. The “references” column lists relevant literature to justify these input parameter changes.

sediment assumptions. For example, in the FDD simulation, decreasing the dredge depth from 1 meter (e.g., a channel dredging project) to 0.1 m (e.g., depth of a dredge fishery) significantly decreases carbon benefits by resuspending and removing less sediment in the BAU site (Fig. 2C). Yet in either case and in the infill scenario, sediment carbon gains eclipse those occurring from natural meadow carbon sequestration, as is seen in the associated “Seed” and “Transplant” scenarios (Fig. 2A). This is unsurprising given sediment accumulation rates in seagrass meadows are estimated at 5 to 6 mm/yr, meaning that dredging to 1 meter leads to rapid loss of sediment that may have taken around 170 to 200 years to accumulate^{45,46}. This concept is similarly represented when viewing the “HSA” simulation, in which we assume that the meadow accumulates sediment at a relatively high rate, applying the global average, which is nearly four times higher than what was estimated from restored meadows with the Virginia LTER represented in the “Base” simulation (138 ± 38 vs. $37 \pm 3 \text{ g C m}^{-2} \text{yr}^{-1}$, respectively; Tables 4, 5). While this high sediment accumulation leads to greater carbon gains relative to the “Seed” and “Transplant” scenarios in other simulations (nearly four times higher; Table 6), it is still relatively insignificant when compared to the sediment carbon gains in “Infill” and “Conservation” scenarios (but may be comparable to a conservation project with small sediment disturbance, Fig. 2A). For similar reasons, the “LID” simulation leads to lower sediment carbon gains, proportional to the change in infill depth (Table 6; Fig. 2).

As described in section “Carbon gains over time”, CH₄ and N₂O emissions are also eclipsed by sediment parameters in the “Infill” and

“Conservation” scenarios. However, if restoration approaches are taken with no additional sediment management, we see that in the absence of these trace gas emissions, the carbon gains increase by about 18% from the “Base” to “ZE” simulations for both seeding and transplanting (Table 6), when using the emissions estimated in Oreska et al.¹⁹ (Table 4). Over time and for large projects, this could translate into significant alterations in carbon credits or climate mitigation potential.

If we assume the carbon offsets from the base model simulation (Table 6) represent Verified Carbon Units (“VCU”, equivalent to the reduction or removal of one tonne of CO₂eq) within the voluntary carbon market, we can further understand the carbon market potential of such activities. The price of carbon can vary widely and has gone up considerably over the past decade⁴⁷. Some relevant market indices showing prices around \$30 USD/CO₂eq⁴⁸, but with examples of prices over \$100 USD/CO₂eq for cutting-edge projects or those with co-benefits²³. While variation in carbon pricing is beyond the scope of discussion here, we apply prices of \$30 and \$70 USD/CO₂eq to the base simulation to roughly estimate potential revenue from the implementation of these four carbon project approaches (Table 7). From this, we see that for a 6 ha project over 10 years (a standard project evaluation period), seeding and transplanting projects would make very little revenue from sale of VCUs, particularly given this revenue does not include the cost of the project or emissions from project activities.

Fig. 1 | The net carbon values (annual) are indicative of carbon sequestered or emitted in the management sites minus that from the BAU sites (A: Nitrous Oxide; B: Methane; C: Biomass; D: Soil). Plots result from the “Base” model simulation (Table 4). Gray areas display the 95% CI.

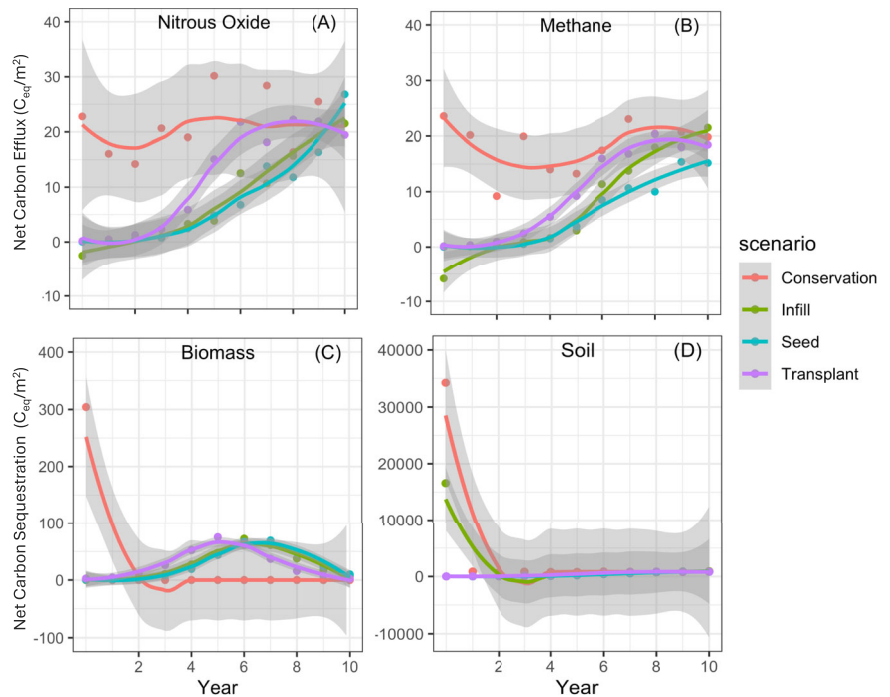


Table 6 | Net cumulative carbon gains (Metric Tonnes Ceq) from five model simulations, which are outlined in Table 5 and displayed in Fig. 2

Simulation ID	Scenario	Net C gain (MT Ceq)	SD Net C gain (MT Ceq)
Base	Conservation	358.2	35.9
Base	Infill	164.7	28.0
Base	Seed	10.8	1.7
Base	Transplant	13.3	2.1
ZE	Conservation	353.1	34.9
ZE	Infill	174.8	25.6
ZE	Seed	12.8	1.1
ZE	Transplant	15.6	1.3
HSA	Conservation	435.8	51.7
HSA	Infill	206.6	35.6
HSA	Seed	37.0	9.3
HSA	Transplant	48.3	13.9
LID	Conservation	362.4	30.3
LID	Infill	28.7	4.0
LID	Seed	11.0	2.0
LID	Transplant	13.6	2.2
FDD	Conservation	56.7	6.3
FDD	Infill	170.5	21.9
FDD	Seed	11.0	1.8
FDD	Transplant	13.4	2.1

These values do not account for emissions resulting from other project activities such as use of vehicles to conduct restoration.

Discussion

We found that while seagrass restoration and conservation can lead to additional carbon sequestration, the seagrass management approach taken can significantly influence associated carbon gains and associated revenue from VCUs. Typical seagrass restoration approaches involve directly

transplanting seagrass shoots or seeding (e.g., ^{33,49}). Modeled results of seagrass management outcomes demonstrated a clear difference (~2.5 T Ceq over 6 ha) in carbon outcomes between these two approaches, based on the rate of meadow expansion. However, both modeled approaches would generate little revenue from the sale of VCUs over 10 years (less than \$1000/Ha).

Carbon gains from transplant and seed restoration approaches are far outpaced by those incorporating the addition or preservation of sediment (one meter), which can result in carbon gains an order of magnitude higher (with soil carbon sequestration from the latter approaches eclipsing that of the former approaches; Fig. 1D). Revenue estimates from conservation and infill scenarios result in significantly higher per hectare VCU revenues (~\$3000–\$15,000 per Ha) than traditional restoration approaches (~\$200–\$570 per Ha). If the size or duration of projects increased, increases in revenue potential could also be expected. These findings align with Oreska et al.¹⁹ stating that “carbon offset credits currently provide a marginal incentive for seagrass restoration”. However, if there are opportunities where restoration can be opportunistically paired with sediment management, these incentives can greatly increase, particularly if carbon prices increase and for larger projects. For example, if a 100 Ha seagrass meadow could be protected from dredging or erosion of 1 meter of sediment following seagrass loss, this has the potential to generate \$1.5 Million USD from VCUs in the first decade of crediting alone, assuming a high relative carbon price (\$70 USD/Tonne). Applying sediment management approach may not be ecologically or logistically feasible in all contexts, and the suite of factors that may contribute to seagrass persistence and expansion must still be considered^{32,50}. Determining where such conservation and restoration opportunities might exist (including those involving sediment management) requires strong spatial datasets, seagrass monitoring, site-specific knowledge, and enforceable conservation measures, in addition to ensuring demonstrable carbon benefits from project implementation. It remains unclear how many of and where opportunities with high relative carbon benefits exist across the globe.

The seagrass model presented here serves as a framework to conceptualize expected carbon outcomes from seagrass conservation and restoration based on the best available science. However, variations in environmental conditions, restoration success trajectories, and management approaches can all affect a project’s carbon outcomes. This model

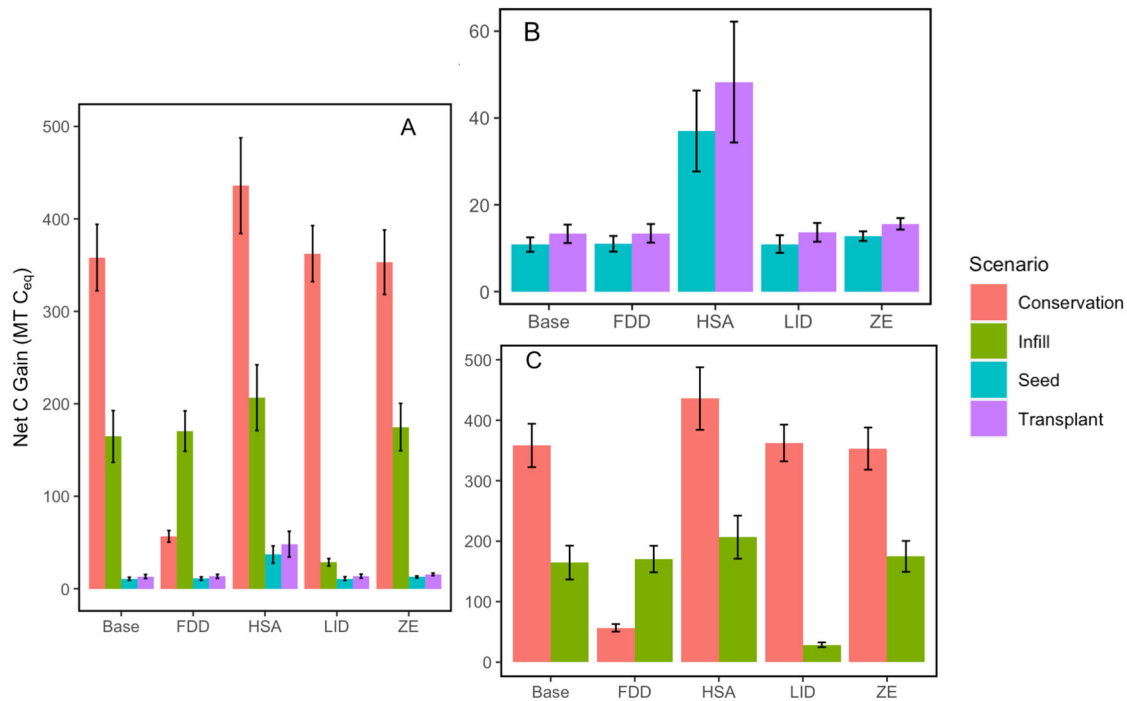


Fig. 2 | Net cumulative carbon gains (Metric Tonnes Ceq) from five model simulations, which are outlined in Table 5. A is split into two subplots to show the relative differences between scenarios that do not use sediment management approaches (transplant and seed; **B**) and those that do (conservation and infill; **C**).

FDD Fisheries Dredge Depth (10 cm), HSA High Sediment Accumulation ($138 \pm 38 \text{ g C m}^{-2} \text{ yr}^{-1}$), LID Low Infill Depth (10 cm), ZE Zero Emissions (no methane or nitrous oxide emissions).

Table 7 | Revenue generated over 10 years from sale of VCUs based on carbon prices of \$30 and \$70 USD per VCU (reflected in the ranges) and based on the average Net C gain (Table 6) over two potential project areas of 6 Ha and 100 Ha

Scenario	Net carbon gains (MT CO ₂ eq/Ha)	Revenue (\$USD)/Ha	Revenue (\$USD): 6 Ha Project	Revenue (\$USD): 100 Ha Project
Conservation	219.1	\$6573–\$15,337	\$39,438–\$92,022	\$657,297–\$1,533,693
Infill	100.7	\$3022–\$7052	\$18,133–\$42,311	\$302,225–\$705,196
Seed	6.6	\$198–\$462	\$1189–\$2775	\$19,818–\$46,242
Transplant	8.1	\$244–\$569	\$1464–\$3417	\$24,406–\$56,946

Note that the Net C gains in Table 6 have been converted to Metric Tonnes/CO₂eq (standard VCU units). Revenue is not inclusive of project costs.

enables us to understand how key parameters might influence carbon gains, informing areas of opportunity and improvement.

The presented model assumes logarithmic growth, with a slight delay in growth in seeding compared to transplant projects. It also assumes the perfect survival and persistence of meadows through time. While a necessary model component, it should be recognized that seagrass restoration is not always successful, and global and regional assessments show very mixed results. Specifically, many seagrass restoration projects fail, or if they do succeed, may not always follow such idealized trajectories^{32,50}. Given these trajectories underpin the model, understanding and applying in situ restoration trajectories can lead to more informed estimates of carbon gains. Moreover, seagrass species, biomass, coastal geomorphology, hydrodynamics, and sediment type can all influence organic carbon stocks and accumulation⁵¹. For example, the sediment accumulation rates from the Virginia LTER meadows used in the “Base model”⁴⁵ are three to four times lower than the global average accumulation rate (see “HSA” simulation), and sediment carbon stocks can also vary over orders of magnitude³³. Thus, while the model presented here may reasonably estimate carbon reduction potential from temperate beds of perennial *Z. marina*, the environmental

conditions, location, seagrass species, and associated starting carbon conditions can all alter expected outcomes. Reparameterization of models informed by new data will aid in understanding this variability and its effect on potential carbon credit generation from projects under different conditions.

The conservation scenario also assumes that in the absence of management intervention, dredging (or a related driver of sediment loss) occurs once over the entire meadow, leading to the complete loss of seagrass and underlying sediment. Dredging can occur to varying depths (centimeters to meters) for multiple activities. For instance, dredging for clearance of navigational channels could occur over depths of multiple meters but infrequently (e.g., decadal), whereas dredge fisheries can disturb sediments to shallower depths (e.g., 6–15 cm) but occur far more frequently^{49,52,53}. Modeling carbon fluxes from dredging is an active area of research and can be far more complex than what is modeled here, meriting exploration. Additional stressors such as MHWs can also lead to seagrass and sediment loss^{36,54,55}, however, they are more challenging to prevent than point-source stressors like dredging, complicating the efficacy and carbon outcomes of implementing a conservation measure. The volume and depth of sediment

loss and associated carbon remineralization from navigational dredging, dredge fisheries, and natural erosion following seagrass loss can also be highly variable, which could drastically alter emissions and credit revenue outcomes. Relatedly, infill practices are already being applied in restoration projects, however, very little work has been done to evaluate the carbon emissions associated with these practices, preventing robust estimates of potential gains. For example, some restoration projects may apply sediment to raise elevation to suitable depths, while others may use it to “cap” underlying unstable, fine-grained sediment^{38,40,56}.

We modeled four key carbon pools and fluxes (biomass, soil, methane, nitrous oxide). However, other carbon fluxes can influence total carbon offset potential, depending on the context, species, environmental setting, or other factors. For example, CO₂ emissions produced from calcium carbonate (CaCO₃) production have the potential to reduce total project carbon offsets - a process that could occur in meadows with high sediment carbonate concentrations^{57–59}. Data from the Virginia LTER sediments indicate extremely low carbonates and no significant difference between unvegetated and restored seagrass sites^{19,55,60}. While this provides justification for the exclusion of CaCO₃ production in the model as applied in the model Virginia LTER system, it may not be a fair assumption for all seagrass meadows.

We also assume here that there are no additional CO₂ emissions produced from the implementation of the project activity. In some contexts, emissions may be below the *de minimis* threshold (representing less than 5% of total project GHG emissions benefits)²¹. However, the conservation BAU (in which the meadow loses surface sediments) and infill management scenarios could potentially result in significant CO₂ emissions from non-sediment sources (e.g., construction and vehicle emissions). Project-specific considerations would need to accurately incorporate these emissions into the total carbon benefits.

Lateral flows of carbon may further complicate a project’s carbon offsets and are not included in the conceptual model here⁶¹. Within existing carbon offset frameworks, care is required to ensure no “double counting” of carbon across landscapes occurs²¹. For instance, doubling counting could occur in a scenario where a salt marsh restoration project receives carbon credits, but “credited” carbon from this salt marsh is subsequently exported to a neighboring seagrass habitat for burial in sediment and credited again within the seagrass habitat. Conversely, laterally exported seagrass carbon could be unaccounted for in recipient habitats. Emerging work studying carbon transport across seascapes informs our understanding of how frequently these flows occur and the permanence of carbon in recipient landscapes^{62,63}. Nonetheless, future work that more holistically evaluates carbon across seascapes, rather than at project-specific boundaries, can elucidate the prevalence of potential double-counting, sedimentary autochthonous carbon contributions, and more generally, inform carbon dynamics in coastal zones. Despite these model caveats, we posit that the relative differences between carbon offsets in these unique management scenarios provide novel insights into how a project’s carbon offsets will vary based on these approaches.

Ensuring the durability (“permanence”) of seagrass carbon removals and reductions remains an integral part of the application of these NbS and to eligibility for carbon credit finance. A range of factors, particularly climate stressors such as MHWs, may result in the reversal of seagrass carbon removals, but there is limited data at present to help understand the likelihood of these reversals^{36,55,64,65}. This is an active area in carbon markets, with Verra considering the establishment of a long-term reversal monitoring system for nature-based carbon credits⁶⁶. Relatedly, we simulate over 10 years, a standard evaluation period under VCS guidelines. Within the VCS, projects must be monitored for 40 years, which could result in ongoing carbon sequestration, increasing total potential carbon benefits. Conversely, this time could also allow stressors such as MHWs or storms to reduce potential benefits - a risk that may increase as the climate changes⁶⁷.

This study focused on carbon benefits over an initial period (10 years) based on relevant, available experimental data. However, seagrass conservation and restoration may provide long-term, ongoing carbon removals over centennial timescales as sediments continue to accumulate⁶⁸. As

longer-term seagrass carbon data become available (e.g., carbon accumulation rates), future work could more accurately model the potential for continued removals over decades. Furthermore, well-designed, scientifically robust NbS established with the support of local communities and a commitment to long-term management can ensure project resilience, making carbon storage more durable, and facilitating the persistence of the wider benefits provided for people and nature⁶⁹. Achieving this may entail larger-scale system interventions, potentially at the catchment scale, incorporating terrestrial conservation and management activities to support the health of downstream seagrass meadows (e.g., reducing runoff)⁷⁰.

This work elucidates how significantly the variability in seagrass project approaches and carbon fluxes can affect carbon offsets and potential VCU generation - meriting the need for strong datasets that encompass and demonstrate this variability to support and revise these estimates. The model development process simultaneously provided insight into the available data, uncertainty, and gaps informed by the current evidence base. For instance, despite an abundance of data on sediment carbon stocks and to a lesser extent, sediment carbon sequestration rates, very few of these data are available in restored meadows and relative to a pre-restoration state (or to a neighboring unvegetated site for comparison). Specifically, the Virginia LTER was the only site globally where data from all four key carbon parameters could be found from restored, vegetated sites, and unvegetated sites—limiting our ability to make data-informed estimates from seagrass meadow in any other locales. As new data become available, models can be iteratively improved and move towards a better understanding of seagrass NbSs.

Data on methane and nitrous oxide fluxes in seagrass meadows are particularly sparse and highly variable in space and time⁷¹. From what data are available, some suggest that seagrass can act as net nitrogen sinks (reducing N₂O emissions), others suggest no additional methane or N₂O emissions relative to unvegetated sites, while others still suggest higher emissions relative to unvegetated sites^{19,72–74}. No data exist on these fluxes in the same location before, during, and after seagrass restoration or recovery. The uncertainty surrounding the magnitude and drivers of these emissions in seagrass meadows represents a large gap in knowledge, which could significantly improve carbon offset estimates in these habitats.

While the importance of preserving and managing carbon in sediment has been presented conceptually in the literature^{35,75}, little quantitative work has been conducted despite the potential carbon benefits, as highlighted here. Further, it is well documented that dredging or other causes of seagrass and sediment loss (e.g., marine heat waves, “MHWs”) occur across the globe, yet there is very little literature on these events’ frequency, depth of associated sediment loss, and subsequent impacts on carbon stocks^{36,64,65}. Given the large potential emissions from sediment loss and the mitigation potential from preventing such activities (e.g., dredging, bottom trawling)^{35,76,77}, a deeper understanding of these effects in situ on carbon stocks is required. Such data could improve future models, and more broadly, guide where habitat protections might lead to the greatest climate mitigation potential. For instance, here we assume that 50% of the carbon in applied sediment is mobilized and remineralized during the infill process, which can drastically alter carbon outcomes. This value and other associated emissions will likely vary based on numerous factors, such as sediment source and time exposed to oxic conditions. However, there are no empirical data documenting remineralization values with respect to coastal restoration projects to our knowledge.

To receive avoided emissions credits (i.e., the conservation scenario), projects must sufficiently demonstrate that the driver of emissions (e.g., dredging) would occur under BAU conditions, and that carbon credits could facilitate the cessation of that driver (i.e., meeting additionality requirements). Similarly, in the infill scenario, as much of the carbon value is contained in the infilled sediment, crediting potential and revenue may depend on where the sediment is sourced from, with a need to ensure that the carbon benefit is not simply payment for carbon transferred from a pre-existing sediment stock with no net removal. To the authors’ knowledge, there is not currently a standard approach to tracking and rewarding carbon benefits from infilling. Given the climate value indicated here, and that this

approach is currently being applied, future crediting programs could consider including this activity^{39,40}.

It should also be noted that the size of many seagrass restoration projects may challenge their application in carbon markets. For example, mangrove restoration projects receiving credits within the voluntary carbon market can be over 10,000 Ha, while seagrass restoration projects typically have much smaller footprints³⁰. Meta-analyses of regional and global seagrass restoration show projects typically transplant small areas (tens to hundreds of square meters), and it can take decades to restore tens to hundreds of hectares across large regions, largely due to the challenges with restoration implementation, success, and monitoring^{32,50}. Nonetheless, some larger-scale opportunities may exist^{2,78,79}. Advances in seagrass restoration, technologies, approaches to group projects when crediting, and better identification of where large-scale opportunities exist can all facilitate potential support from carbon credits.

Although we demonstrate carbon benefits exclusively herein, not all seagrass NbS projects may prioritize this ecosystem function nor be well-poised to take advantage of carbon financing. It is well recognized that seagrass meadows provide a range of other ecosystem services in addition to carbon removal and storage⁸⁰. Future work could simultaneously evaluate how other essential ecosystem functions might recover over time, enabling insight as to how to prioritize and invest in broader, multifunctional nature recovery. There may also be opportunities for other (i.e., non-carbon) crediting programs for additional ecosystem services (e.g., nutrient reduction⁴⁶) or supporting wider environmental aims (e.g., biodiversity credits, as above) to provide additional revenue to support a seagrass project. Policies and non-credit based financial support for seagrass habitats may also play a key role in protection and restoration. There are still challenges to overcome in how to operationalize blended financial models; such as whether multiple credits from a project can be “stacked” or must be “bundled” into a single higher-value credit^{81,82}. These blended models may ultimately make seagrass protection and restoration viable in locations where it would not be if funded through carbon credits alone.

We conclude that the model presented here demonstrates the high variability in carbon benefits between seagrass management approaches, with those adding 1 m of sediment prior to restoration, or preventing loss of 1 m of sediment, sequestering roughly 13 to 33 times (respectively) higher than approaches with no sediment mobilization. Estimated revenues from carbon credits also vary significantly, ranging from \$198 (seeding) to \$15,337 (conservation) per Ha. Carbon market funding opportunities for NbS, including seagrass habitat restoration, are contingent on project size, project approach, and carbon price, among other factors. While sediment management approaches can maximize carbon gains, carbon mitigation is not the only ecosystem function of interest^{83–85}. Gains in biodiversity and other key ecosystem functions may be more tightly associated with the seagrass habitat itself and therefore more agnostic to associated sediment management. As such, we argue that seascape-level approaches that pair strategic sediment management with multi-habitat restoration are most likely to lead to the desired portfolio of ecosystem functions including climate mitigation and biodiversity support, among others. The synergistic connections between restored seagrass meadows and the wider seascape have supported the functionality between multiple interacting habitats, carbon gains, and restoration outcomes^{86,87}. These interconnections and synergies are important to consider when scaling up coastal NbS. Furthermore, the protection and restoration of seagrass meadows should also be supported for their intrinsic ecological and biodiversity value. Public and private initiatives increasingly recognize and support these aims as highlighted by recent international commitments to increase natural ecosystem area and the emergence of market-based approaches^{88,89}.

Data availability

Data that support the findings of this study have been accessed from LTER Data Catalog at <http://www.vcr.lter.virginia.edu/cgi-bin/browseData.cgi> and all applied data are also in the manuscript. The seagrass management model is shared via Github at https://github.com/maward-sci/c_scenarios.

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Author contributions

M.W., C.D., M.M.D., A.S., and L.M.W. contributed to the study conception and design. M.W. and C.D. led the modeling and data analysis. M.W. wrote the main manuscript text. All authors reviewed the manuscript.

Competing interests

L.M.W. is a member of the npj Ocean Sustainability journal's Editorial Board.

Additional information

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