# Restoring blue carbon systems

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## Impact Statement

Coastal habitats such as mangroves, tidal marshes and seagrasses have faced extensive loss in area due to human activities such as land use change and pollution. This loss occurs despite the value of these ecosystems as important reservoirs of 'blue carbon', with much of their carbon stored in the soil column. Recent interest has turned to blue carbon conservation and restoration, with managers and policy makers around the world setting targets to restore habitats to generate carbon credits or meet national targets for climate change mitigation that are larger in magnitude than targets previously set. This article collates the empirical evidence base for how restoration activities can positively impact various parts of the blue carbon cycle and contribute to climate change mitigation. However, carbon benefits will only happen if projects can overcome the various socioeconomic, governance and biophysical constraints to restoration that currently limit our ability to restore coastal landscapes at the scale required to tackle the climate change challenge.

1

## Abstract

Mangroves, tidal marshes and seagrasses have experienced extensive historical reduction in extent due to direct and indirect effects of anthropogenic land use change. Habitat loss has contributed carbon emissions and led to foregone opportunities for carbon sequestration which are disproportionately high due to high 'blue carbon' stocks and sequestration rates in these coastal ecosystems. As such, there has been a rapid increase in interest in using coastal habitat restoration as a climate change mitigation tool. This review shows that restoration efforts are able to increase blue carbon stocks by as much as three times, while also having a positive impact on various gaseous fluxes. However, blue carbon increases are spatially variable, due to biophysical factors such as climate and geomorphic setting. While there are potentially hundreds of thousands of hectares of land that may be biophysically suitable for restoration, these activities are still often conducted at small scales and with mixed success. Maximising the potential for carbon gains through blue carbon restoration will require managers and coastal planners to overcome myriad socioeconomic and governance constraints related to land tenure, legislation, target setting and cost, which often push restoration projects into locations that are biophysically unsuitable for plant colonization.

## Introduction

Natural climate solutions — actions that protect, sustainably manage and restore ecosystems — are an important tool in mitigating climate change and keeping global temperatures below a 2°C increase by the end of this century. Natural climate solutions, such as those provided by in global forests, wetlands, grasslands and agricultural lands, could provide one-third of the cost-effective climate mitigation needed to achieve this goal, equivalent to 23.8 Pg $CO_2$ $yr^{-1}$ (Griscom et al., 2017), while providing a wide range of co-benefits such as food provision, livelihoods and cultural services to local communities.

A natural climate solution that has gained substantial interest over the last decade is the conservation and restoration of blue carbon. The current definition of blue carbon refers to the carbon sequestered and stored in 3 specific coastal ecosystems: mangroves, tidal marshes and seagrasses. Mangroves are a community of salt-tolerant trees covering >145,000 $km^2$ (Jia et al., 2023) in the tropics, subtropics and

2

warm temperate regions, and in some instances are able to store almost 2,800 Mg C ha$^{-1}$ in soil layers 1-6 m deep (Adame et al., 2021). Tidal marshes cover 53,000 km$^2$ globally (Worthington et al., 2023), while estimates of global seagrass extent vary from ~160,000 km$^2$ (McKenzie et al., 2020) to ~1.65 million km$^2$ (Jaythilake & Costello, 2018), depending on methods used for mapping these systems. Together, these three ecosystems store >30,000 Tg C (Macreadie et al., 2021).

Despite their importance for climate change mitigation, blue carbon ecosystems have experienced substantial declines in area due to human land use change and coastline modification, and continue to be lost around the world (Fluet-Chouinard et al., 2023). Mangroves were coarsely estimated to be lost at rates of 1-3% yr$^{-1}$ in the 20$^{th}$ Century, though this has reduced to approximately 0.1-0.2% from 2000 onwards (Friess et al., 2019). Tidal marshes have been lost at 0.28% yr$^{-1}$ in the 21$^{st}$ Century (Campbell et al., 2022), and seagrasses are threatened across much of their global range, with losses of at least 5,602 km$^2$ since 1880 (Dunic et al., 2021). Habitat loss contributes to climate change through the emissions of stored carbon; in the 21$^{st}$ Century, global mangrove loss led to the emissions of 26.3 Tg CO$_2$e yr$^{-1}$ (Hamilton & Friess, 2018), while tidal marsh loss released 16.3 Tg CO$_2$e yr$^{-1}$ to the atmosphere (Campbell et al., 2022).

Lost areas of habitat provide an opportunity for new blue carbon accumulation through habitat restoration. In coastal wetlands, the term 'restoration' encompasses a range of management actions. They can generally be defined as the planting of seedlings (*mangroves*, Zimmer et al., 2022; *tidal marshes,* Sparks et al., 2013; *seagrasses,* van Katwijk et al., 2016) or the encouragement of natural regeneration, often through the removal of environmental stressors or the reintroduction of hydrological flows (*mangroves,* Lewis, 2005; *tidal marshes,* Garbutt & Wolters, 2008; *seagrasses,* Bouderesque et al., 2021), and/or the broadcasting of seeds and propagules on high tides (e.g., Orth et al., 2020). Coastal restoration activities are attracting substantial recent interest and funding (UNEP-WCMC, 2022), and restoration is a management action which is now the basis of several blue carbon projects (Friess et al., 2022a).

In this review we synthesize experiences of the restoration of mangroves, tidal marshes and seagrasses for blue carbon outcomes. Specifically, we outline: (i) the global scale biophysical potential for the restoration of blue carbon ecosystems; (ii) estimates of blue carbon benefits following coastal habitat restoration; and (iii) current challenges and constraints to the effective restoration of blue carbon

3

ecosystems. We aim for this to be a broad synthesis of the potential benefits of restoration and challenges to their implementation, and refer the reader to in-depth or systematic reviews on specific topics where appropriate.

## Large-scale scope of restoration for blue carbon

Due to centuries of coastal habitat loss, a large extent globally of areas could be reverted to their original state, with concomitant gains in blue carbon. Macreadie et al. (2021) estimate this area to be approximately 30 million ha (with a 95% confidence interval of 17.5-41.6 million ha), with seagrasses accounting for 57% and mangroves accounting for 37% of this potential. Tidal marshes have the lowest global restoration potential, probably because current land uses (such as urban development in estuaries) preclude conversion back to tidal marsh. If the restoration of blue carbon ecosystems could be conducted successfully at this scale, it could result in the removal of 841 Tg $CO_2$e per year from the atmosphere, equivalent to ~3% of global fossil fuel emissions (Macreadie et al., 2021). Studies for mangroves specifically suggest that 8,120 km$^2$ that has been deforested since 1996 might be biophysically restorable (Worthington & Spalding, 2018); resulting in 365 million tonnes of carbon gains once fully restored.

Such large potential carbon gains align with several international initiatives promoting large-scale coastal habitat restoration, such as the Trillion Trees Initiative and the Bonn Challenge (Lovelock et al., 2022), though interest and implementation from these initiatives in blue carbon ecosystems could still increase further to match that of terrestrial ecosystems (Waltham et al., 2020). Countries have also identified large-scale opportunities for blue carbon restoration. For example, Indonesia has a target of restoring 600,000 ha of mangroves by 2024, with carbon gains as a key driver of this policy (Sidik et al., 2023). While studies suggest that only ~186,600 ha (Worthington & Spalding, 2018) to 193,367 ha (Sasmito et al., 2023) are biophysically suitable for restoration in Indonesia, it shows the commitment that some countries have for blue carbon restoration. Other countries have set blue carbon restoration targets within their Nationally Determined Contributions to the Paris Agreement by 2030; in 2020–2021 Belize committed to restoring <4,000 ha of mangroves, Haiti committed to increasing its mangrove area to 19,500 ha, and Samoa committed to increasing its mangrove area by 5% (Friess, 2023).

## Carbon benefits of restoration

4

The achievement of meaningful coastal restoration targets is expected to have important impacts on blue carbon cycling and storage (Figure 1). Restoration will have positive impacts on above-ground and below-ground biomass pools, with carbon stocks increasing by 2-800 times, depending on ecosystem and setting (Stagg & Mendelssohn 2010; Sasmito et al., 2019; Oreska et al., 2020; Rosentreter et al., 2021; Iram et al., 2022; Shah et al., 2022; Kelsall et al., 2023). Similar positive impacts are expected on the soil carbon pool (generally referring to the top 1 m of sediment), though patterns are more mixed due to legacy carbon from prior land uses. Positive impacts on carbon fluxes are generally expected. For example, methane emissions would be expected to be as much as 4 times lower than degraded sites or those under other land uses, particularly if the replaced habitat was dominated by freshwater, due to the influence of salinity on methanogenesis (e.g., Cotovicz Jr et al., 2024).

**Figure 1.** *Restored blue carbon stocks (above- and belowground, AGB and BGB) tend to be 2–800 times higher than in degraded/converted/bare sites, and methane ($CH_4$) fluxes can be 4 times less than degraded/converted/bare sites, depending on habitats and age. For example, a) seagrasses, bare versus (vs) restored; (Oreska et al., 2020); b) Mangrove, converted/degraded vs restored (Sasmito et al., 2019; Rosentreter et al., 2021); c) Tidal marshes; converted/degraded vs restored (Kelsall et al., 2023; Iram et al., 2022; Stagg and Mendelssohn 2010; Shao et al., 2022); Refer to the original references for details; Positive values indicate an increase and negative values indicate a decrease.*
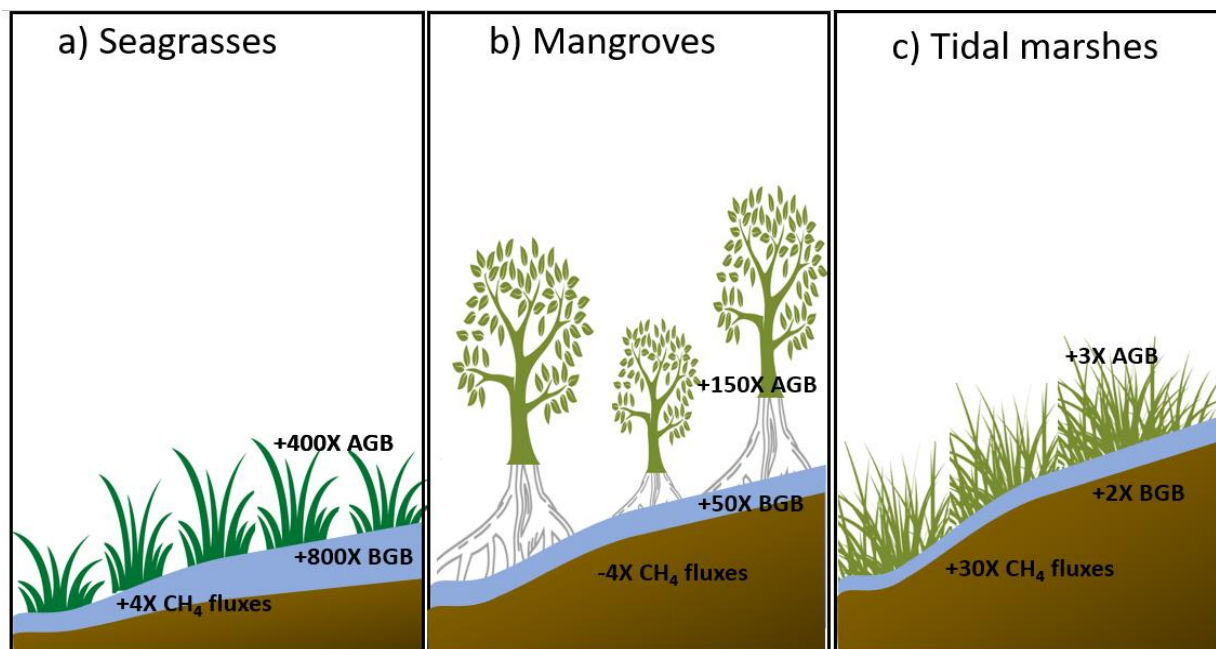
The following describes the major stock and flux components separately for each blue carbon ecosystem, and is indicative of the magnitude of stock and flux change that can be expected following restoration. For in-depth reviews of general carbon stocks and fluxes for different blue carbon ecosystems, we direct the reader to systematic reviews such as Fourqurean et al. (2012; seagrass stocks), Johannessen (2022; seagrass carbon accumulation rates), Sasmito et al. (2019; carbon stock change and GHG flux change with mangrove restoration), Mason et al. (2023; carbon accumulation rates and GHG fluxes for tidal marshes), and Taillardat et al. (2020; GHG fluxes for multiple blue carbon ecosystems). For further information on explicit geomorphic and ecological drivers of blue carbon stocks and fluxes, we direct the reader to a recent review by Kirwan et al. (2023).

*Mangroves*

Mangrove restoration efforts vary in the degree of human intervention, and include ecological engineering, monoculture plantations, afforestation, and ecological mangrove restoration that promotes natural regeneration (Ellison et al., 2020). The latter is encouraged through hydrological modifications including construction or reconfiguration of tidal creeks, culverts, sediment additions to change elevation, and tidal reintroduction (Lewis, 2005). Restoration projects are increasingly incorporating carbon benefits, and chronosequence studies are being used to understand blue carbon dynamics with restoration age (Osland et al., 2012; Andreetta et al., 2016; Marchand, 2017; Walcker et al., 2018; Wang et al., 2021; Azman et al., 2023) alongside systematic literature reviews (Sasmito et al., 2019; Rivera-Monroy et al., 2022) to estimate carbon outcomes from restoration.

Total ecosystem carbon stocks in restored mangroves generally increase with site age. For example, studies in southeast Australia indicated a strong relationship between carbon and site age (Carnell et al., 2022), with stocks in older restored forests (17 and 35 years) averaging ~115 MgC ha$^{-1}$, compared to only ~50 MgC ha$^{-1}$ in younger forests (13 years) (Carnell et al., 2022). In a restoration chronosequence study in Vietnam, total ecosystem carbon stocks also increased with stand age (2 to 27 years old) from ~201 to ~519 MgC ha$^{-1}$ (Pham et al., 2017). However, specific carbon pools may respond differently to restoration; meta-analyses suggest that mangrove biomass carbon increases for 15 years at a rate of 4 MgC ha$^{-1}$ yr$^{-1}$ after restoration, though mixed patterns are observed in soil carbon stocks to 1 m depth (Sasmito et al., 2019).

6

Although aboveground biomass can regenerate rapidly, carbon storage belowground takes substantially longer to replenish, with shallow soil carbon pools restocked more quickly than deeper layers due to organic matter and allochthonous inputs. A study in Thailand of hydraulic restoration and planting following conversion to shrimp ponds reported that soil organic carbon concentration increased over four years and stocks increased from 110 to 160 MgC ha$^{-1}$ in two years, based on measurements to 5 cm soil depth (Matsui et al., 2010). In created mangroves along a 20-year chronosequence in Southwest Florida, soil organic matter and total carbon increased with site age in the upper 10 cm of soil and were estimated to need 18–28 years to reach natural equivalence (Osland et al., 2012). Total carbon in the upper 10 cm soil layer was 30 g kg$^{-1}$ in the created wetland compared to 144 g kg$^{-1}$ in the natural wetland, and 13 g kg$^{-1}$ and 88 g kg$^{-1}$ in the lower 10–30 cm layer in the created and natural wetlands (Osland et al., 2012). Other studies have observed soil organic carbon increases with restoration activities (Zhang et al., 2012; Dung et al., 2016; Pham et al., 2017; Sasmito et al., 2019; Ratul et al., 2022; Thura et al., 2023). Restoration activity can also influence soil carbon returns; a global review found that restoration was more effective at accumulating carbon in the top meter of soil over 40 years compared to afforestation projects (Song et al., 2023). In some cases, carbon stocks in restored mangroves can exceed those in natural systems, but differences in these comparisons may be attributed to local hydogeomorphological controls on carbon storage (Kusumaningtyas et al., 2022).

While most research has focused on changes in carbon stocks over time, we lack a clear understanding of how carbon sequestration and greenhouse gas fluxes respond to mangrove restoration. Limited evidence from Southeast Australia showed that sequestration rates do respond to site age, at ~3 Mg C ha$^{-1}$ yr$^{-1}$ in older forests compared to ~1.5 Mg C ha$^{-1}$ yr$^{-1}$ in younger forests (Carnell et al., 2022). Methane fluxes decreased with age, being ~5 times higher in the younger forest compared to older forests, with rates of 5.8 mg CH$_4$ m$^{-2}$ day$^{-1}$ (Carnell et al., 2022). Flux measurements are important to understand the climate change mitigation potential of mangrove restoration; in an analysis of mangrove restoration offset potential in North Sulawesi, carbon mitigation was estimated at -27.6 Mg CO$_2$-e ha$^{-1}$ yr$^{-1}$ (Cameron et al., 2019). Scaled up to the national level in Indonesia, the country with the largest mangrove area globally, this potential could represent up to an offset potential of 16.56 million Mg CO$_2$-e ha$^{-1}$ yr$^{-1}$ (Cameron et al., 2019).

*Tidal marshes*

There is global interest in restoring tidal marshes to recover lost ecosystem functions and services (Hagger et al., 2022; Macreadie et al., 2017, Kellway et al., 2020). Tidal marshes are often restored in areas with high salinity (>15 ppt), and can be colonized by a range of plant species adapted to those conditions (Craft, 2007; Negandhi et al., 2019). Several techniques are involved in restoring tidal marshes (Billah et al., 2022; Craft, 2022; Mason et al., 2023) that can enhance carbon sequestration (Macreadie et al., 2017; Kellway et al., 2020). Common techniques include the placement of dredge material on subsiding marshes or construction of new marshes by filling open water areas (Costa-Pierce & Weinstein, 2002; Madrid et al., 2012). Hydrology, including salinity, tidal flow, and water level conditions, are major drivers influencing tidal marsh productivity and the success of restoration. Hydrological restoration involves diverting river water (Baustian et al., 2023) and de-embankment of existing dikes or enlarging culverts (Karberg et al., 2018; Wolters et al., 2005). Tidal reinstatement is used for restoring marshes on marginal agricultural lands in coastal areas (Kelleway et al., 2020; Lovelock et al., 2022), reintroducing tidal connections through the installation, removal or modification of water regulation structures.

The time needed for tidal marsh restoration to achieve ecological equivalence to natural marshes can vary between 3 to >15 years (Billah et al., 2022; Broome et al., 2019) but significant increases in carbon stocks (up to 3 times; Shao et al., 2022) can be reached as early as four years (O'Connor et al., 2020). Annual aboveground carbon production was 200 to 1,700 g C m$^{-2}$ in constructed wetlands of the United States (from placement of dredged material) at 2 to 3 years old (Madrid et al., 2012). Restored wetlands in China also observed aboveground biomass stocks near 167 g C m$^{-2}$, which were three times higher than degraded wetlands (Shao et al., 2022). The total carbon soil stock (to 1 m) of restored marshes can be about 51.86 Mg C ha$^{-1}$, approximately two times higher than degraded wetlands (Shao et al., 2022).

Annual fluxes are often measured and combined to assess net carbon (or $CO_2$ equivalence) benefits (Baustian et al., 2023). Above and belowground net primary production rates of restored marshes in the U.S. can increase by four times (depending on sediment treatment) compared to degraded sites (Stagg & Mendelssohn, 2010). Soil carbon accumulation rates vary in restored tidal marshes in comparison to natural marshes, with some studies indicating the rates are lower than those of natural marshes (Broome et al. 2019, Kelsall et al., 2023, Table 1), whereas other studies indicated soil carbon accumulation rates of restored salt marshes can be nearly twice as high as reference marshes (Poppe & Rybczyk, 2021, Mason

8

et al., 2023). The direction of this relationship may be somewhat independent of restoration, and more influenced by biophysical processes and plant species composition (Mason et al., 2023).

**Table 1.** *Indicative carbon abatement benefits (g C m$^{-2}$ yr$^{-1}$) from soil carbon accumulation rates (from 10–30 cm depths) and greenhouse gas fluxes of tidal marsh restoration (see references for details on marsh ages, geomorphic settings etc.). For a global review of carbon fluxes of restored salt marshes see also Mason et al. (2023).*

| Carbon flux | Tidal Marsh | | References (Location) |
| --- | --- | --- | --- |
| | Restored | Natural | |
| Soil carbon accumulation (g C m$^{-2}$ yr$^{-1}$) | -42 to -125 | -91 to -329 | Baustian et al., 2017, 2020 (USA); Hansen & Nestlerode, 2014 (USA); Craft et al., 1999 (USA); Cammen, 1975 (USA) |
| $CO_2$ flux (g C m$^{-2}$ yr$^{-1}$) | +345 to +2,907 | +664 to +2,141 | Burden et al., 2013 (UK); Iram et al., 2022 (Australia); Negandhi et al., 2019 (Australia); Poffenbarger et al., 2011 (USA); Shiau et al., 2019 (USA) |
| $CH_4$ flux (g C m$^{-2}$ yr$^{-1}$ | +0.03 to +1.5 | -1.2 to +0.2 | Adams et al., 2012 (UK); Burden et al., 2013 (UK); Iram et al., 2022 (Australia); Kroeger et al., 2017 (global); Negandhi et al., 2019 (Australia); Poffenbarger et al., 2011 (USA); Al-Haj & Fulweiler, 2020 (global); Shiau et al., 2019 (USA) |

Anoxic soil conditions promote greenhouse gas emissions, reducing the magnitude of carbon abatement benefits (Emery & Fulweiler, 2017). Emissions of $CO_2$ or $CH_4$ from restored tidal marshes are highly variable and influenced by the legacy of restored soil (Iram et al., 2022), hydrology, vegetation (Derby et al., 2022), and elevation levels (Negandhi et al., 2019). Emissions are also strongly influenced by salinity regime (Poffenbarger et al., 2011; Kroeger et al., 2017). Restored tidal marshes are considered most effective in providing carbon abatement because the microbial communities are influenced by salinity and tidal exchange, thus resulting in reduced greenhouse gas fluxes (Kroeger et al., 2017; Negandhi et al., 2019). Restored tidal marshes had $CO_2$ and $CH_4$ fluxes (Adams et al., 2012; Burden et al., 2013; Iram et al., 2022; Negandhi et al., 2019; Table 1) within the range of reference marshes and global averages for natural saltmarshes (Kroeger et al., 2017; Poffenbarger et al., 2011; Rosentreter et al., 2021). However,

9

high precipitation events cause freshening, thus buffering salinity and reducing the carbon abatement potential of restored salt marshes during such events (Negandhi et al., 2019).

*Seagrasses*

Seagrass restoration has been conducted for many decades (van Katwijk et al., 2016, Ward & Beheshti 2023). Although the scale of restoration lags compared to other coastal ecosystems, newer techniques in seed-based restoration (Tan et al., 2020) and mimicry of emergent traits (Temmink et al., 2020) can facilitate large-scale planting and increase restoration effectiveness (van Katwijk et al., 2009). Several seagrass restoration guidelines focus on seagrass restoration at regional (e.g., Western Indian Ocean Region) or national (e.g., Sweden, Kiribati, UK and Ireland) scales (e.g., UNEP, 2020; Gamble et al., 2021). Additionally, a specifically designed restoration guideline that incorporates carbon benefits has been developed under the Verified Carbon Standard (VCS), VM0033 Methodology for Tidal Wetland and Seagrass Restoration.

Although some successful seagrass restorations have been documented (van Katwijk et al., 2016; Tan et al., 2020, Orth et al., 2020), few report carbon stock recovery and sequestration in restored meadows. Those that have been assessed for carbon recovery primarily occur in three regions: the temperate coastal bays of Virginia in the mid-Atlantic, U.S., the subtropical/tropical Gulf of Mexico, U.S., and temperate southern Australia (Table 2). These measurements show that: (i) restored meadows have comparable or higher carbon burial rates and surface sediment carbon stocks compared to mature meadows (McGlathery et al., 2012; Marbà et al., 2015; Thorhaug et al., 2017; Orth et al., 2020); (ii) estimates can be made regarding timing, trajectory, spatial patterns, and sources of blue carbon recovery following seagrass restoration (McGlathery et al., 2012; Greiner et al., 2013; Greiner et al., 2016; Oreska et al. 2017a; Oreska et al., 2017b); and (iii) there are impacts of disturbance and natural recovery on carbon (Marba et al., 2015; Thorhaug et al., 2017).

In Virginia, where seagrass meadows have been restored via seed broadcasting since 2001 (McGlathery et al., 2012; Orth et al., 2020), carbon stocks in the top 5 cm of sediment in the meadow were twice that of adjacent bare sediments after nine years (278.9 compared to 138.7 g C m$^{-2}$; McGlathery et al., 2012) and 1.3x greater than that of younger (1-5 year old) areas of the meadow (Orth et al., 2020). Carbon burial rate was significantly higher in 10-year-old meadows (36.68 g C m$^{-2}$ y$^{-1}$) compared to four-year-old

10

meadows and unvegetated areas (no net accumulation), with a five-year lag period after seeding before carbon burial rates doubled. Sequestration rates were expected to reach functional equivalence to mature meadows by year 12 after seeding based on projections of continued increases in shoot density (Greiner et al., 2013). This observed and expected increase in carbon sequestration occurred in parallel to a fining of the sediments and increases in shoot density with age. However, twelve years after restoration, the distribution of $C_{org}$ concentrations at the meadow-scale was driven by hydrodynamics, rather than the age of the seagrass within the meadow, with higher $C_{org}$ concentration further into the meadow away from the bare subtidal edge due to current attenuation (Oreska et al., 2017a). Stable isotope analysis indicated that sources of sediment organic matter in the restored seagrass meadows were distinct from those of bare sediment, and were on average composed of ~40–50% from seagrass and 46–56% from benthic macroalgae and/or seston produced *in situ*, with only ~3–10% coming from macroalgae or adjacent *Spartina alterniflora* salt marshes in restored meadows spanning four to 13 years old (Greiner et al., 2016; Oreska et al., 2017b).

In southwest Australia, carbon burial rates in restored meadows also increased with age and reached functional equivalence to mature meadows 18 years after planting (mean carbon burial rate $26.4 \pm 0.8$ g C $m^{-2}$ $y^{-1}$; Marbà et al., 2015). In the Gulf of Mexico, seagrass meadows restored in eight previously disturbed areas of seagrass loss had higher 20-cm-deep sediment carbon stocks (mean $C_{org}$ stocks $38.7 \pm 13.1$ Mg $ha^{-1}$) than impacted barren or always barren sediments, with highest carbon stocks in older restored beds. These values were comparable to or higher than natural seagrass meadows, suggesting that seagrass restoration can reduce carbon losses from previous disturbances to seagrass meadows (Thorhaug et al., 2017). Mean organic carbon gains in the top 20 cm of sediment in restored sites, compared to impacted (now barren) sites, were estimated as 20.96 Mg $C_{org}$ $ha^{-1}$; however, carbon accumulation rates varied significantly by site and restoration age (but not species), with intermediate-aged seagrasses (4–15 years old) having the highest carbon accumulation rates compared to young (<3 years old) or old (>15 years old) restored seagrasses (Thorhaug et al., 2017).

A complete understanding of the greenhouse gas offset potential of seagrass restoration requires a full inventory of carbon fluxes. The first complete inventory for a restored meadow (in Virginia) estimated net offsets of 0.42 t $CO_2$e $ha^{-1}$ $yr^{-1}$, for which the financial benefit based on carbon sequestration would only cover about 10% of restoration costs, suggesting additional ecosystem services should be assessed for incentivization (Oreska et al., 2020). Recent multidisciplinary modeling approaches have combined

11

extensive knowledge of species' growth and the $C_{org}$ benefits through restoration (Duarte et al., 2013; Reynolds et al., 2016), which can demonstrate the long-term potential of seagrass restoration to capture and store carbon.

One challenge in understanding carbon dynamics following seagrass restoration is geographical biases in data collection and availability, with restoration projects mainly documented in the temperate and subtropical coastlines of North America, Europe, East Asia, and Australia (van Katwijk et al., 2016), and blue carbon benefits only recorded in studies in the U.S. and Australia (Table 1). This is despite key academic and management interest in restoring tropical seagrasses for climate change mitigation (Rifai et al., 2023). Similarly, most studies focused on the restoration of temperate species, with a single study assessing $C_{org}$ benefits of restoration of tropical species (Table 2). In tropical regions, several seagrass species co-occur within one meadow, adding further complexity to restoration. The traits of tropical species are highly variable, from fast-growing colonizing species to long-lived persistent species, with different responses to disturbance (Kilminster et al., 2015). Multispecies restoration, such as a combination of the five species that include all species traits (colonizing, opportunistic, and persistent), has been shown to have the highest restoration potential in Indonesia (Asriani et al., 2018; Williams et al., 2017), and the combined restoration of colonizing and persistent species resulted in high organic carbon stocks in the Gulf of Mexico across 15–16 years and 38–43 years since restoration (Thorhaug et al., 2017).

While seagrass restoration can restore carbon sequestration processes (Table 2), many studies have not demonstrated additionality – the additional carbon generated as a result of an intended management action, which is a prerequisite for the calculation of carbon credits. Most restoration projects measure success based on metrics related to habitat attributes (e.g., shoot density) rather than the return of ecosystem services (Orth et al., 2020). For studies that do measure changes in carbon, some approaches (such as radionuclides to determine soil carbon accumulation rates) require years of sediment accumulation, and natural seagrass sediment dynamics such as resuspension and mixing can prevent the estimation of sediment and carbon accumulation rates (Lafratta et al., 2020). Other methods of measuring sediment accumulation to quantify carbon sequestration in seagrass meadows have proven difficult, such as modifications to the Rod Surface Elevation Table method to apply it to seagrass meadows, due to problems with measuring surface elevation change below the water surface, and maintaining feldspar marker horizons (Potouroglou et al., 2017; but see Ewers Lewis & McGlathery, 2023). Quantification of changes in carbon dynamics are important to measure because not all restored

meadows accumulate substantial volumes of additional carbon. Meadows exposed to high hydrodynamic energy may lack sediment accumulation, which can cause sediment mixing and erosion to be present decades after restoration/recovery (Lafratta et al., 2020). Finally, uncertainties surrounding the influence of restoration on emissions of $CH_4$, $N_2O$, and $CO_2$, represent a global knowledge gap (Oreska et al., 2020), and accounting for emissions may result in low carbon benefits for some seagrass restoration projects.

**Table 2.** *Summary of specific studies investigating seagrass carbon benefits through habitat restoration.*

| Location(s) | Bioregion and climate | Species | Carbon benefits | References |
|---|---|---|---|---|
| Virginia Coast Reserve Long Term Ecological Research site, Virginia coastal bays, U.S.A. | North Atlantic, temperate | *Zostera marina* | Sediment carbon content doubled in top 5 cm after 9 y (bare=138.7 g C m$^{-2}$, restored 9 y =278 .9 g C m$^{-2}$); Carbon storage (including sediment and vegetation) was 3x that of bare sediments after 9 y; Carbon burial rate doubled after a 5-year lag period at start of restoration; Carbon burial rate was 36.68 g C m$^{-2}$ y$^{-1}$ after 10 y compared to non-depositional in unvegetated; | Aoki et al., 2021; Greiner et al., 2013, 2016; McGlathery et al., 2012; Oreska et al., 2017a, 2017b, 2020; Orth et al., 2020; Reynolds et al., 2016 |
| Gulf of Mexico; Biscayne Bay, Florida, and Corpus Christi Bay, Galveston Bay, and Predator Island Laguna Madre, Texas, U.S.A. | Tropical Atlantic, tropical | *Thalassia testudinum, Halodule wrightii, Halophila decipiens, Syringodium filiforme* | Mean sediment carbon stocks to 20 cm in restored sites that had previously been seagrass were 38.7 ± 13.1 Mg ha$^{-1}$, comparable or higher than natural meadows; Mean carbon gains to 20 cm =20.96 Mg $C_{org}$ ha$^{-1}$ Carbon burial rates were higher in restored beds of intermediate-age (4-15 y) compared to young (<3 y) or old (> 15 y) | Thorhaug et al., 2017 |
| False Bay, South Australia | Southern Australia, temperate | *Posidonia australis* | Radionucleotide dating failed due to low depositional and high hydrodynamic environment, leading to no estimates for carbon burial rates Carbon stocks showed a general trend of | Lafratta et al., 2020 |

14

| | | | being highest in intact seagrasses, followed by recovered seagrass, and non-recovered ($5.7 \pm 1.2$, $4.5 \pm 0.7$, and $3.3 \pm 0.3$ kg $C_{org}$ m$^{-2}$, respectively), but were not significantly different | |
| --- | --- | --- | --- | --- |
| Oyster Harbor, Western Australia | Southern Australia, temperate | *Posidonia australis* | Carbon burial rates increased with age and were functionally equivalent to mature meadows after 18 years ($26.4 \pm 0.8$ g C m$^{-2}$ y$^{-1}$) | Marbà et al., 2015 |

15

## Challenges for successful restoration

Regardless of restoration technique, blue carbon benefits will only be realized if restoration is conducted successfully. All restoration techniques ultimately require correct biophysical conditions within the site to allow successful planted or natural seedling establishment. While data availability on restoration outcomes is poor (Eger et al., 2022; Gatt et al., 2022), the global track record in restoring blue carbon ecosystems is mixed. For example, most mangrove projects in Sri Lanka showed 0% survival, with only 3 sites >50% (Kodikara et al., 2017). Similarly in Colombia, only 24% of projects were highly successful (Rodríguez-Rodríguez et al., 2021). For seagrass projects along the U.S. west coast, ~60% were unsuccessful (Ward & Beheshti, 2023). Restoration, whether through planting or the encouragement of natural regeneration, is challenging because seedling mortality is generally high even in natural intertidal systems (e.g., van Regteren et al., 2020; Sloey et al., 2022), and seedlings are often not planted in biophysically suitable conditions. Restoration attempts in suboptimal locations occur primarily due to socio-economic and governance reasons (Friess et al., 2022b; Figure 2). This could include conflicting land tenure claims in suitable areas, pushing restoration activities into lower intertidal areas with fewer tenure conflicts, or ambitious short-term planting targets that require large extents of plantable areas to be found at short notice (e.g.,, Wodehouse & Rayment, 2019).

**Figure 2.** *Interlinked socio-economic, governance and biophysical constraints lead to low blue carbon restoration success.*
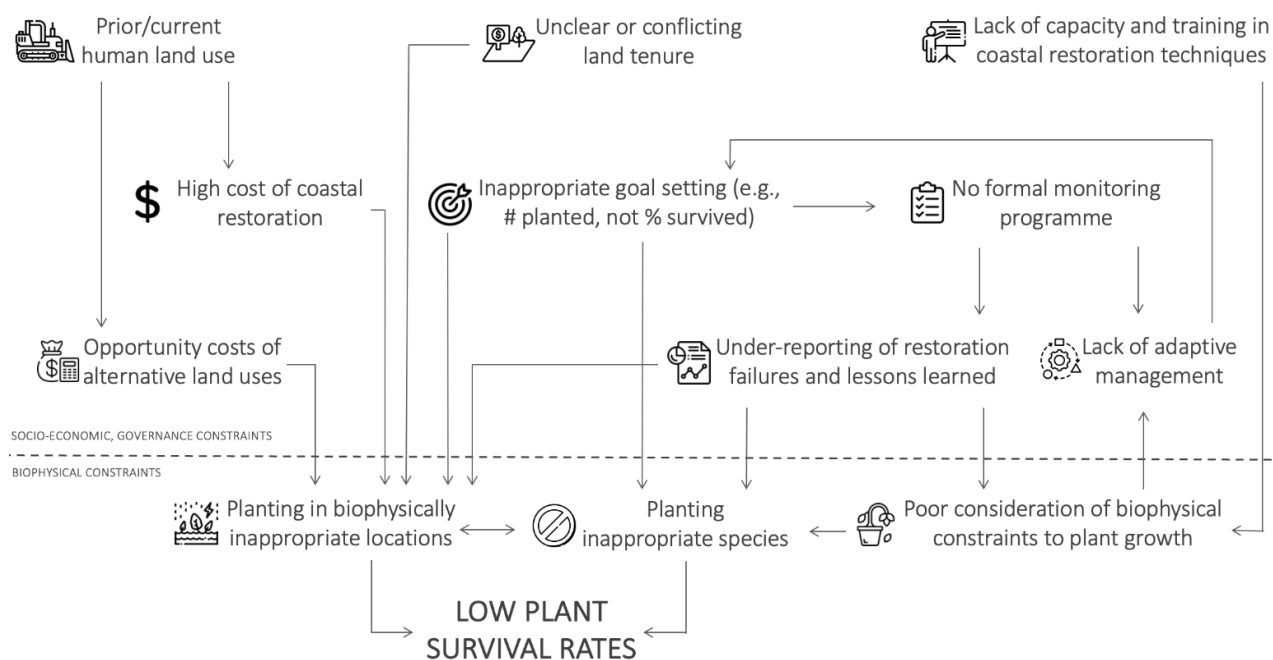
*Biophysical constraints to blue carbon restoration*

The mortality of seedlings will be high if biophysical conditions are unsuitable for establishment. Generally, this means the physical environment of a restoration must original ecohydrological conditions before disturbance. In mangroves and tidal marshes, this often relates to changes in the hydroperiod that are unsuitable wetland plants. For example, site elevations change substantially when aquaculture ponds are dug out, or when reclaimed areas are disconnected from further sediment input compared to the surrounding intertidal area. Other site conditions constraining wetland vegetation colonization include high sediment consolidation and low microtopography (Brooks et al., 2015).

Once physical factors constraining natural reestablishment are understood, a key management step in wetland restoration is the removal or alleviation of that biophysical stressor (Lewis, 2005). In intertidal projects these are often site-based engineering steps, such as manipulating site elevations through sediment addition (e.g., Staver et al., 2020), and strategic dyke breaching (e.g., Kiesel et al., 2020; López-Portillo et al., 2021) to allow adequate water flows. However, removing constraints to natural reestablishment can be particularly challenging in seagrass restoration, as this ecosystem is often affected by stressors that occur at a distance from the restoration site, such as eutrophication caused by organic pollution from the surrounding watershed (van Katwijk et al., 2016). Lack of freshwater input due to

17

impoundment and manipulation of the wider watershed has also been a constraint to successful coastal wetland vegetation establishment (Liu et al., 2021).

*Socioeconomic and governance constraints to blue carbon restoration*
While biophysical variables determine the successful establishment of individual plants, the reasons why restoration projects are placed in suboptimal conditions in the first place is often the consequence of broader socioeconomic and governance constraints. Specifically, current land use may prohibit restoration back to the original blue carbon ecosystem; coastlines reclaimed for urban development have substantially less restoration potential than other land uses, for example (Worthington et al., 2020).

For intertidal areas where human use has been abandoned, restoration (particularly in tropical regions) can be inhibited by unclear or conflicting land tenure. Clarifying the land tenure landscape along many coastlines involves substantial investigation and negotiation, which will take time that may not be available for a project, pushing restoration efforts to locations with fewer land tenure concerns, such as commons or state lands in the subtidal zone that are not suitable for plant growth (Lovelock & Brown 2019; Friess et al., 2022b). Land tenure can be an obstacle to restoration even in developed bureaucracies. For example, challenges are faced when negotiating permission from private landholders, persuading risk averse public stakeholders that restoration on their lands will be successful, or where ownership boundaries on the coast are still ambiguous (Bell-James et al., 2023).

Inappropriate goal setting can also lead to poor restoration outcomes (Waltham et al., 2021), such as planting in inappropriate locations. Many projects set the number of seedlings planted as a goal. Planting over the short time scales mandated by funders may encourage managers to seek large open areas (such as undervalued tidal flats, *sensu* Chen & Lee, 2022) that provide sufficient space for planting but are not biophysically suitable for establishment (Lovelock & Brown, 2019; Wodehouse & Rayment, 2019). Ultimately, coastal restoration will be most successful when targets are based on robust knowledge of past and future changes that underpin suitable scenarios of environmental performance (Sheaves et al., 2021), which allows for long-term, iterative and adaptive management (e.g., Thom, 2000).

Finally, coastal restoration costs can be high compared to terrestrial restoration, due to costs such as construction actions for hydrological manipulation (Bayraktarov et al., 2016). Carbon accounting of a

rehabilitated seagrass meadow in the South Bay, Virginia, U.S. suggests that blue carbon credit generation may only offset ~10% of restoration costs (Oreska et al., 2020). Costs can sometimes be reduced by community labour for planting and other activities (e.g., Tan et al., 2020), though this is not often the major cost in a restoration project. As such, blue carbon projects that involve restoration are currently not breaking even, and future projects will likely require co-financing options such as tax incentives or philanthropic funding in addition to carbon credit sales (Friess et al., 2022a). Combined, socioeconomic and governance factors have important implications for the scale of restoration; the area biophysically suitable for mangrove restoration in Southeast Asia is estimated at >303,000 ha, but only 5.5–34.2% of this area was deemed restorable once financial, land use, and operational constraints were considered (Zeng et al., 2020).

*Spatial prioritization of blue carbon restoration*

Many restoration projects are opportunistic in their location (e.g., Ledoux et al., 2005) due to low land availability and opportunity costs. However, achieving ambitious restoration targets and meaningful blue carbon gains requires landscape-scale planning. Modelling can consider various biophysical, socioeconomic, and governance constraints when determining optimal restoration locations (e.g., Syahid et al., 2023), and has been widely used in predicting potential changes in the distribution of mangrove (Hu et al., 2020; Rodriguez-Medina et al., 2020; Sahana et al., 2022), seagrass (Bertelli et al., 2022), and tidal marsh (Boumans et al., 2002; Raw et al., 2021). Species distribution models have been used to determine areas for seagrass (Stankovic et al., 2019; Valle et al., 2011) and mangrove (Hu et al., 2020) restoration. These models predict the environmental distribution of species and can be projected into geographical space as habitat suitability, probability of species occurrence or favorability of species occurrence, which can be interpreted as restoration potential (Stankovic et al., 2019). The positive correlation of the models' habitat suitability prediction and the planting units of seagrass (Valle et al., 2015) demonstrated that usage of the model can eliminate "best professional judgment" for site selection (Stankovic et al., 2019). However, most models rely on complex relationships of the abiotic factors that shape a realized niche at the specific point in time, rather than the fundamental niche (Grech & Coles, 2010), which can provide greater potential extents and additional restoration sites, resulting in an increase in total seagrass coverage (Oreska et al., 2021). In addition, these models coupled with spatial drivers and models of blue carbon variability (e.g., Ewers Lewis et al., 2019) offer guidelines for the return of the ecosystem services through restoration across spatial and temporal scales.

Modelling can also be used to show how the realized/fundamental niche may be modified as physical conditions change with climate change, such as increasing inundation periods under sea-level rise. This has most commonly been applied to conservation projects, by assessing the sustainability of existing blue carbon resources and their projected distributions in the future (e.g., Rosencranz et al., 2019 for tidal marshes; Nguyen et al., 2022 for seagrasses and mangroves). Restoration suitability is implicit in these models when they predict landward migration into presently terrestrial areas, as those areas will need to be colonized by new intertidal vegetation. While less common, intertidal habitat restoration has also been specifically modelled under sea-level rise, with biogeormorphic models of a planned tidal marsh restoration project showing that newly colonizing vegetation will be able to keep pace with realistic rates of project sea-level rise, and resilience is particularly influenced by restoration method (Gourgue et al., 2022).

## Conclusions

Extensive habitat loss creates ample opportunity for the restoration of mangroves, tidal marshes and seagrasses. The evidence is clear that their restoration leads to substantial per hectare gains in blue carbon, and has led to rapid interest in blue carbon restoration for the achievement of climate change mitigation policy. While the realizable area of restoration globally is much smaller than its biophysical potential, future blue carbon restoration is likely to make a meaningful contribution to climate change mitigation, whilst providing a wide range of co-benefits to coastal communities around the world.

These important carbon gains and other co-benefits will only be unlocked, however, if success rates and scales of restoration increase substantially beyond what is currently achieved around the world. Restoration projects, particularly for mangroves and seagrasses, often struggle to meet their success criteria due to myriad biophysical, socioeconomic and governance constraints. Issues of land tenure and opportunity costs often push mangrove restoration projects into uncontested but suboptimal spaces in the lower intertidal or upper subtidal zone, or seagrasses are restored in areas where the main driver of ecosystem degradation (such as water pollution) has not been addressed. However, guidelines and examples exist to overcome such barriers, and improve the success of coastal habitat restoration and the carbon benefits they provide.

20

## Data Availability Statement

No primary data were generated for this manuscript.

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