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Potential of blue carbon as a natural climate solution

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Key Points

- Blue carbon ecosystems ('BCE' - mangrove forests, tidal marshes, seagrass meadows) are gaining international recognition as a natural climate solution (NCS) to help meet climate change mitigation and adaptation targets.
- BCE potentially have a ~36-185 million hectares global distribution (~the size of Mexico) and store 33 billion tonnes of carbon, while providing important adaptation co-benefits (such as coastal protection, fisheries/biodiversity enhancement).
- Protecting existing BCE could avoid emissions of 304 (141-466) Tg CO_2e pa and large-scale restoration could remove an extra 841 (621-1,064) Tg CO_2e pa by 2030, which is equivalent to ~3% of annual global greenhouse gas emissions.
- The potential contribution that blue carbon can make to nature-based solutions in the future will depend on societal actions. Restoring BCE at scale should be a key focus of the UN Decade of Ecosystem Restoration (2021-2030), with opportunities for developing robust information on the scope of small-scale restoration projects at the global or country scale.
- Emerging blue carbon markets should aim at incorporating the value of co-benefits into financial frameworks (including via impact investment, blended finance) to assist with the investments required for restoration and conservation.

Abstract

Nature climate solutions (NCS) recognise the importance of conservation and restoration of ecosystems for climate change mitigation and adaptation. A global focus on blue carbon ecosystems (‘BCE’ - including mangrove forests, seagrass meadows, and tidal marshes) as NCS arises because of their capacity for rapid carbon sequestration, long-term storage, avoiding emissions of disturbed ecosystems, and valuable adaptation co-benefits, such as coastal protection, and fisheries enhancement. BCE are calculated to store >30,000 Tg C globally across approximately 185 M ha, with conservation of BCE potentially avoiding emissions of 304 (141-466) Tg CO_2e pa, while large-scale restoration could drawdown an extra 841 (621-1,064) Tg CO_2e pa - collectively amounting to ~3% of global emissions. Most opportunities relate to mangroves, while opportunities from NCS approaches based on tidal marsh and seagrass require better understanding of their global distribution, losses and recovery. It is unlikely that destruction of BCE will come to a full stop, or that all destroyed BCE can be restored; however, innovative engineering solutions and strategic planning for coastal protection from sea level rise with green infrastructure offer opportunities to create or restore BCE. Investments required to fully restore BCE can exceed returns from carbon market financing, but gaps could be met by monetizing and financing for co-benefits, particularly coastal protection. We conclude that BCE are a cost-effective and scalable option for climate change mitigation and adaptation while enhancing human wellbeing.

Keywords: blue carbon, NDCs, climate change, restoration, ecosystem service, wetland, mangrove, seagrass, tidal marsh, protection

Introduction

Climate change mitigation is a grand societal challenge focused on achieving the goals of the Paris Agreement to limit the global average temperature to below 2°C, preferable to 1.5°C, above pre-industrial levels (Article 2, Paris Agreement, 2015). To achieve this target, signatory countries must develop a national climate plan, also called 'Nationally Determined Contributions', aiming for long-term climate change mitigation and adaptation . These actions include efforts to reduce emissions as well as efforts to remove excess greenhouse gasses (GHGs) from the atmosphere. In this context, nature climate solutions (NCS [1,2]) have emerged as a preferred option to achieve both these goals. NCS, including carbon sequestration through ecosystem management and reforestation, have the potential to be more cost-effective and scalable than technological options [3,4], such as direct air capture, geological sequestration and biochar production. Many of these technological options have yet to be deployed at large scale, and face significant economic, social and environmental barriers [5].

NCS were originally based overwhelmingly on terrestrial ecosystems (referred to as 'Green Carbon'), with coastal and ocean-based opportunities being largely ignored [6]. However, over the past decade research has shown the globally relevant role of coastal vegetated ecosystems in carbon drawdown, broadly referred to as 'Blue Carbon' (to denote a link to coastal ecosystems) (Figure 1). Blue carbon ecosystems ('BCE') refers to coastal habitats (particularly seagrass meadows, mangrove forests, tidal marshes, and potentially seaweed beds) that contribute significantly to carbon drawdown due to their intense greenhouse gas removal, their long-term permanence of carbon removed and the large carbon stocks accumulated, which support large potential emissions when disturbed [7] (Figure 1). Blue carbon strategies propose the conservation and restoration of these ecosystems as a strategy to mitigate and adapt to climate change [8].

BCE are widespread, highly productive habitats that rank among the most valuable ecosystems on Earth. They support diverse marine and coastal communities across multiple trophic levels by providing key functions such as food and habitat provision while supporting human wellbeing, including artisanal fisheries, nutrient cycling and coastal protection against erosion and sea level rise, which influence the livelihoods of millions [9–11]. BCE occupy approximately 0.5% of the sea floor, but contribute > 50% of global carbon burial in the oceans [12–14]. Their area stretches as a thin ribbon along the world's shorelines, from the upper intertidal zone down to about 50 m depth, depending on underwater light penetration. The high carbon sequestration rates of BCE derive from three main reasons; their ability to produce, capture and preserve carbon. They are highly productive ecosystems that produce large amounts of carbon ('autochthonous carbon'), and their position at the land-sea interface and high trapping capacity allows them to accrete sediment and accumulate carbon produced by other ecosystems ('allochthonous carbon') [8]. They have the ability to accumulate and preserve this carbon in forms that resist microbial attack, which is due to a variety of reasons, but mainly their water-logged, anaerobic conditions that limit decomposition [15].. Although BCE are exposed to natural disturbances such as cyclones and flooding, the general absence of fires in BCE provides more confidence on the permanence of carbon stores over climatically-relevant centennial to millennial timescales compared to terrestrial habitats [8,16]. Unfortunately, about 50% of BCE global extent has been lost across the globe [8], leading to emissions of greenhouse gases from the carbon stores they supported [17]. Hence, halting the loss of BCE, and reversing losses through restoration will help avoid emissions from disturbed BCE and will restore their carbon sequestration role [6].

There is ample scope for both conservation and restoration of BCE, as only about 1.5% of the global extent of BCE are included in marine protected areas [18]. Restoration of BCE at the scale required to recover much of the lost habitat area is potentially feasible for mangroves and tidal marshes, but challenging for seagrass meadows [19], with a 10-fold growth in the number of documented restoration projects of BCE since the turn of the century [20]. The restoration of about 1,487 km^2 of mangroves in the Mekong Delta destroyed by the US Air Force during the Vietnam war [21] remains one of the largest ecosystem restoration projects ever undertaken [20]. The planted forest in Can Gio Mangrove Forestry Park, has accumulated 152 Tg of CO_2e, greater than the total C emissions of Vietnam for the year 2013 (130 Tg CO_2e) across the 447 km^2 of mangroves restored at this site, suggesting the total Mekong mangrove restoration area to have removed an equivalent to about three-times Vietnam's 2013 greenhouse gas emissions [22]..

An important aspect of NCS that differs from other, more technological climate change mitigation solutions is that in addition to sequestering carbon they also generate a range of other ecosystem services that contribute to local human wellbeing (Figure 2). These ecosystem services include nutrient removal [23], fisheries enhancement [24] and coastal protection [25] (Figure 2). For example, the establishment of mangrove protected areas has been associated with long-term gains in fisheries production [26]. Knowledge of ecosystem services, such as evidence of protection from the 2004 Indian Ocean Tsunami [27], has driven massive mangrove planting programs in South and Southeast Asia. Mangroves are also increasingly valued for their contribution to protecting communities from the impacts of tropical storms and cyclones [28,29]. The full suite of functions and services that BCE provide alongside carbon sequestration increases the adaptive capacity for communities to cope with natural hazards and climate change, while helping achieve multiple UN Sustainable Development Goals. As such, the conservation and restoration of BCE can be considered a win-win solution that contributes to both climate change mitigation and adaptation, while enhancing coastal livelihoods [30], thereby being highly valued by coastal communities [31].

In this review, we discuss the global potential of blue carbon contribution to NCS to climate change mitigation and adaptation, focusing on the opportunities of large-scale restoration. We begin by exploring the global estimates of blue carbon stocks and outlining the potential of large-scale restoration of these ecosystems, and discuss the main constraints and feasibility. We subsequently describe the role of BCE on climate change mitigation and adaptation and their co-benefits, and how this information could guide future efforts for restoration strategies and plans for climate change mitigation and adaptation. Finally, we review the role of BCE in helping countries achieve their international emission reduction targets and identify future steps to improve blue carbon estimates on distribution extent, stocks, sequestration, and mitigation potential.

Global blue carbon stocks

Robust information on geographical extent and estimates of carbon storage are key to including BCE in national and global climate mitigation accounting. Current efforts to map BCE show that they encompass ~36 - 185 million hectares (due to large uncertainties in the distribution of seagrass meadows [32,33] and tidal marshes [34]) within the world's coastal zone [32,34,35]. One major caveat associated with the global estimates provided by our study is that all data supporting this review have been extracted from existing literature, which in most cases did not provide error propagation or uncertainty estimates. To overcome this, we estimated upper and lower interval bounds based on available information (i.e., spatial data or country-level estimates; further details are described in the SI) for soil and biomass carbon stocks (Table S1). Based on available information, BCE could potentially hold ~8,970 – 32, 650 Tg C in their soils and biomass. A full description of existing datasets and those used in this review is available on the Supplementary Material.

Mangrove forests occur across ~13.7 million hectares within tropical, subtropical and warm temperate coastal zones [35] (Figure 3) and are the best studied (in terms of publication number) and mapped BCE. Mangrove forests have been the focus of global and regional studies that helped scientists to better understand the ecology, distribution, co-benefits and climate change impacts in the ecosystem [35–46]. Mangrove forests potentially hold ~70% of their carbon in the soil [37], with global estimates ranging from 1,900 Tg C [42] to 8,400 Tg C [41] for the top meter of soil based on previous studies (Table S1). Considering that mangrove forests also store carbon within their living biomass, carbon stocks in this pool could vary from 1,230 Tg C [37] to 3,900 Tg C [47]. However, it is important to highlight that different carbon pools have different turnover times with fresh plant litter rapidly decomposing over years, stabilized soil organic matter persisting for thousands of years to centuries, and poorly defined intermediate carbon pools with turnover times of years to centuries [13,48]. Here, we used the data available on Hutchison et al. [44] and Simard et al. [47] for biomass stocks, and Atwood et al. [45] and Sanderman et al. [38] for soil stocks, as our lower and upper bound estimates, respectively (Supplementary Material Methods, Tables S1-S3). In this case, Indonesia (soil ranging from ~830-1,780 Tg C; biomass ranging from ~570-1040 Tg C), Brazil (soil ranging from ~235-514 Tg C; biomass ranging from ~98-313 Tg C) , Australia (soil ranging from ~83-464 Tg C; biomass ranging from ~113-115 Tg C) and Malaysia (ranging from ~200-365 Tg C; biomass ranging from ~95-255 Tg C) accounted for ~40-50% of global soil and biomass carbon stocks within mangrove forests, regardless of the dataset used to estimate carbon stocks (Tables S2-S3), with Indonesia alone holding more than 20% of this total (Figure 4, Tables S2-S3). It is estimated that mangrove soils alone have lost 30.4 – 122 Tg C due to land-use changes that occurred between 2000 and 2015, with Indonesia, Malaysia and Myanmar contributing to over 75% of these losses [38].

Recent efforts have focussed on understanding the distribution of blue carbon stocks in mangrove forests, but seagrass meadows and tidal marshes still lack robust and spatial-explicit estimates of soil carbon within their geographical extents. However, it is important to highlight recent efforts to map the habitat extent and soil carbon at local and regional scales for seagrass [49–51] and tidal marshes [34,42]. In addition, the Coastal Carbon Research Network, which has been created by the Smithsonian Environmental Research Centre in 2018, is developing publicly available interactive maps of existing soil core data [52].

Considering their mapped [33,53] and potential [32] distribution, seagrass meadows would potentially cover 16 - 165 million ha globally, with these estimates derived from aggregating mapped seagrass areas and modelling the ocean area suitable to support seagrass growth, respectively, resulting in an estimated soil carbon stock varying from ~1,732 - 21,000 Tg C (Supplementary Material). Such order-of-magnitude range is associated with the large uncertainties in seagrass mapping, the difference in methods used in each study and the variability between individual seagrass beds. For example, McKenzie et al. [33] and UNEP-WCMC-Short [53] have compiled existing datasets of mapped seagrass meadows globally to estimate and map seagrass extent, respectively, with McKenzie et al. [33] estimating with low confidence that seagrass meadows could occupy 27 million ha [33]. On the other hand, Jayathilake & Costello [32] used global occurrence records of seagrass and environmental variables to model the potential distribution of seagrass meadows globally at 165 million ha. Despite high variation on numbers, Australia, Indonesia and the United States of America are within the countries with the largest distribution of seagrass meadows, regardless of the dataset used in this review (Table S4).

Currently, we still face a great uncertainty on seagrass mapping, with Jayathilake & Costello [32] and Howard et al. [54] highlighting the lack of data on the eastern and western coast of South America, Africa, Indian Ocean and Indo-Pacific region, and we are only beginning to discover vast meadows of deep-water seagrasses (depths > 50 m) [55–58]. If considering the upper and lower bounds for seagrass distribution extent (Tables S1 and S4) [32,53] and the global estimates of sediment carbon stocks available in Atwood et al. [51], we estimate that carbon stocks could vary from ~3,760 Tg C to 21,000 Tg C with Australia (~37-2,320 Tg C), United States of America (~6-1,700 Tg C) and Indonesia (~41-1,320 Tg C) accounting for ~2-25% of the total carbon stocks within seagrass meadows (Figure 4, Supplementary Material). Australia is one of the few countries in the world that has a full inventory of carbon stocks within BCE, including seagrass meadows, which shows that this ecosystem would hold ~762-1051 Tg C [59]. Using Australia as an example, we can see that using Jayathilake & Costello [32] and UNEP-WCMC-Short [53] seagrass extent layers have some limitations that could potentially underestimate and/or overestimate carbon inventories. This highlights that global maps and estimates are a first step towards estimating blue carbon stocks, however, countries that invest on BCE mapping and carbon sampling will have a better evaluation of its carbon storage capacity, required to support policy actions.

In contrast to mangrove forests, global distribution maps of tidal marshes remain incomplete. Previous studies suggested that the global area of tidal marshes was in the range of 2.2-40 million ha [12,60]. The most recent and comprehensive study to date, estimates ~5.5 million ha across forty-three countries, however, this estimate is likely conservative and incomplete [34]. For example, there are known areas of tidal marshes in Canada, Russia, South America and Africa that are not reflected on current maps, so are not included in the global area estimate. In addition, despite their importance, we are still lacking a global and spatially explicit map of soil carbon for tidal marshes. Based on the global tidal marsh distribution mapped by Mcowen et al. [34], the global soil map [61] and mean carbon stocks (Tables S1 and S5), we estimated that potential blue carbon stocks in tidal marsh soil could vary from ~862-1,350 Tg C (Table S1 and S5) with United States of America (354-640 Tg C), Australia (112-

223 Tg C) and Russia (180-277 Tg C), holding approximately 77-86% of the global soil tidal marsh carbon stocks (Figure 4, Table S5).

To better understand the contribution of BCE to the global carbon cycle, improved global scale mapping of BCE, particularly seagrasses and tidal marshes, is required. Furthermore, it is important to highlight that the range of carbon stock estimates for each ecosystem and carbon pool (biomass and soil) provided in this Review reflect the large uncertainties associated with global blue carbon estimates (for example, linked to poorly constrained total seagrass and tidal marsh habitat extent and the large variability in carbon storage among blue carbon habitats across and within countries). Although large uncertainties remain (such as propagation and uncertainty estimates for each country and global values), the ranges provided in our Review encompass the known variability in blue carbon ecosystem extent and stocks across countries (Table S1). Other significant research gaps include understanding the spatial variability - and drivers of such variability - for key carbon parameters. For example, while there have been improvements in understanding how and why soil carbon stocks vary globally in tidal marshes [62] and mangroves [63,64], global drivers of seagrass carbon stock remain largely unexplored. Less still is known about patterns and drivers of carbon fluxes (carbon burial rates, atmospheric and/or lateral greenhouse gas fluxes) beyond particular sites or regional studies [65,66]. Better understanding of these factors will improve attempts to estimate and model BCE contributions to global carbon cycling, and enhance our capacity to predict the greenhouse gas benefits of restoration activities at national scales [67].

The scope for large scale restoration

Delivering the full restoration potential requires returning these ecosystems to their historical extent, which in many cases remains unknown, but can be derived from consulting historical sources, including paintings [20]. In some cases, urbanization has modified or reclaimed mangrove forests or tidal marshes to an extent that restoration is not possible because it is either economically, legally or logistically unfeasible. However, wetland conversion to urban area represents a relatively minor component of the area lost (for example, accounting for just 3% of mangrove loss globally between 2000 and 2016 [68]) although might have been much greater during urban growth in the 19th and 20th centuries. In many regions, blue carbon habitats have largely been claimed for rice paddies (in Asia), aquaculture ponds (Asia and Central America), and pasture lands (temperate tidal marshes) [69,70], a conversion that was initiated centuries ago With appropriate engineering and consideration of land tenure issues, such lands are more feasible for restoration once they fall out of production, although successful tidal marsh and mangrove restoration also depends on sediment supply in minerogenic systems, as well as current and future soil elevation. Mapping converted areas that can be returned to the original habitat has shown that there is the potential for > 800,000 ha globally to be biophysically suitable for restoration back to mangrove forests [71], if land tenure and other socioeconomic issues can be resolved. Likewise, restoring tidal flows can lead to rapid tidal marsh restoration, with tidal marsh restoration successfully expanding tidal marshes even in some of the most densely urbanized cities in the world, such as New York City, USA [20], although restoration failures have also occurred, providing opportunities to learn [20,72–74].

However, the scope for global-scale coastal wetland restoration is constrained by multiple socio-economic constraints, especially in countries where a large proportion of the restorable habitat is on small agricultural land holdings where restoration efforts could conflict with the livelihoods and food security of local communities [75,76]. Restoration efforts in Southeast Asia highlight the impact of socio-economic constraints on wetland restoration; of the total area that is biophysically suitable for mangrove restoration in Southeast Asia, only 5.5-34.2% is ultimately restorable after various socio-economic (such as livelihoods, food security, and land rights) and operational constraints (including deforestation risk, site accessibility, proximity to seed sources) are considered [77].

Seagrass restoration, however, is comparatively costly and has a lower success rate compared to other marine ecosystems (such as mangrove forests, tidal marshes, coral reefs, oyster reefs) [19,78]. Furthermore, it is important to consider that usually carbon stocks in seagrass biomass and marsh grass leaves are not included in carbon inventories since its residence time varies from only a few months to a few decades, and is therefore, irrelevant to climate change mitigation. Around 29% of the known seagrass global extent has disappeared since the 1940s at a mean rate of 1.5% per year, with large scale losses reported in USA, Australia, New Zealand and Europe [79]. Yet, there is potential to restore areas formerly occupied by seagrass if the pressures that led to the loss, dominated by deteriorated water quality with eutrophication [80], are removed [81]. Natural recolonization is, however, a very slow process. For example, more than 70 million seeds of eelgrass (*Zostera marina*) have been distributed in the western Atlantic coastal lagoons since 1999 that resulted in the recovery of 3,600 ha [82,83]. However, improved seagrass restoration practices are required to enhance the success of previous restoration efforts, which have been largely unsuccessful to date [84].

Seagrass restoration can benefit from a variety of tools and methods (for example, buoy-deployed seeding, dispenser injection seeding, artificial in-water structures and community involvement) that enhance the cost effectiveness, efficiency and scalability of restoration activities [81]. The restoration of seagrass meadows can catalyse natural recovery processes [23,78,85], along with the recovery of the associated ecosystem services including carbon sequestration, which can otherwise be too slow if left to natural recovery, as demonstrated for seagrass meadows in Virginia, USA coastal waters [23]. Restoration of seagrass meadows is complex but possible, and future efforts should combine actions to improve the suitability of the habitat to allow for natural recovery processes in seagrasses, in addition to state-of-the-art methods that enhance the success of active restoration programs [83].

Blue carbon losses from human activities

Quantifying the extent of BCE losses has been hampered by a lack of understanding of their historical and current spatial extent, in particular for seagrasses and tidal marshes. The lack of global time-series for tidal marsh area, comparable to those existing for mangroves [35,43,70], during the satellite record is surprising as they can also be detected using remote sensing tools. However, tidal marsh mapping can be challenging due to their spatial, temporal, and spectral complexity, particularly when bordering other grassland and wetland types [86,87], requiring high spatial resolution of the satellite products, which has only recently been achieved. Uncertainties on past and present seagrass area and distribution are, however, difficult to resolve, as detecting seagrass meadows underwater in satellite images, and separating them from other communities, such as macroalgae, remains challenging [88]. Furthermore, the lack of remote sensing data before the 1970s precludes a robust assessment of global BCE extent prior to the worldwide establishment of coastal settlements.

Despite these difficulties, global losses of tidal marshes have been estimated at ~1-2% per year [89], and seagrass loss rates since the 1940s have been estimated at 1.5% per year [79], but more recently, conservation actions have resulted in the deceleration and reversal of declining trends in seagrasses in some regions [90]. Remote sensing methods for terrestrial forests have been adapted for mangroves, allowing for a country-by-country estimate of ecosystem change from 2000 to 2016 [68]. Globally, annual loss rates of mangroves have decreased sixfold from 0.99% in the 1980s to an average of 0.16% per year between 2000 and 2012 [36]. Although mangrove loss has slowed considerably, loss rates are still substantial in places like Southeast Asia, which contains ~50% of global mangrove forest area. Some

Southeast Asian countries such as Myanmar experienced loss rates as high as 0.5% yr^{-1} between 2000 and 2016 (with 44,485 ha deforested between 2000 and 2016) [68].

The major drivers of BCE loss vary by ecosystem and region, but generally include physical modification (including ecosystem change, drainage), pollution, non-native species, and climate change [91]. In most cases, the primary drivers of BCE loss are indirectly driven by socioeconomic factors that centre around coastal development, energy, food, recreation/tourism, and infrastructure expansion. For example, mangrove forests continue to be converted to expand aquaculture and agriculture (such as rice and oil palm), often to meet national food and economic security targets. Commodity production accounted for 47% of global mangrove loss in the early 21st century, with 92% of commodity production occurring in Southeast Asia alone [68,69]. Similarly, extensive use of fertilizers on agricultural fields and the subsequent eutrophication of coastal ecosystems has been identified as a significant driver of large-scale seagrass and tidal marsh loss [79,80,92].

The effects of climate change (including increased temperatures, sea-level rise, increased frequency and severity of storms and heatwaves) on BCE loss has been increasing over time [68,93,94]. For example, increases in the occurrence and extent of marine heatwaves have led to recent and extensive losses (36-80% loss in seagrass cover) in temperate and subtropical seagrasses in the Chesapeake Bay (USA), the western Mediterranean, and Western Australia's Shark Bay [95–98]. Some mangrove forests, such as those in the Gulf of Carpentaria, Australia [99], have experienced dieback from factors associated with extreme El Niño events (for example, elevated temperatures, reduced inundation, and drought). Furthermore, recent cyclones have also led to widespread damage and losses in mangrove forests [16,100,101].

While the complexity of mangrove dynamics at their poleward limit may make it difficult to show causality [102], increasing air temperature has been related to the poleward range extension of mangroves in many instances [103–107]. For example, in Western Port, Australia, mangrove encroachment was related to increasing annual maximum temperature, at sites with lower elevations and closer to known mangrove-tidal marsh boundaries [104]. Unfortunately, this expansion has generally come at the expense of tidal marshes as they become displaced by mangroves. Such replacement is associated with increases in blue carbon as a carbon-poor herbaceous ecosystem is replaced by a woody ecosystem [108,109].

Sea-level rise combined with coastal squeeze (i.e., habitat loss in the intertidal area due to anthropogenic structures or actions, which prevent the landward migration of coastal habitats that would otherwise occur in response to sea level rise) and reduced sediment loads to coastal systems from the damming of rivers threatens the future persistence of many mangrove and tidal marsh ecosystems [39,110–112]. Although the effects of sea-level rise on BCE loss might not begin to truly manifest for several decades, it is predicted that 5-30% of coastal wetlands can be submerged by 2080 if upslope migration pathways, through lateral accommodation space, are precluded [104,113,114]. Conversely, moderate rates of sea-level rise can be beneficial to the burial of BCE soil carbon, in particular geomorphic settings, or locations - such as much of the southern hemisphere - which have experienced stable relative sea level over recent millennia [62], constraining the soil space available to accumulate sediments by BCE.

Restoration of blue carbon ecosystems

The protection and restoration of BCE has the potential to add substantially to climate mitigation efforts and Nationally Determined Contributions (NDCs). Under business as usual operations, it is expected that blue carbon ecosystems (BCE) will continue to decline due to ongoing global losses, but if protection is prioritised, then blue carbon trajectories should stabilise [115] and even reverse to lead to recovery [20]. Rehabilitation and restoration of BCE will increase the contribution of blue carbon to NCS, as will planning for sea level rise to maximise accommodation space for BCEs under sea level rise.

Despite difficulties in measuring the extent of ecosystem loss in BCE, Griscom et al. [116] estimated the potential for avoided emissions at 304 (141-466) Tg CO_2e yr^{-1} through the protection of these systems, while large-scale restoration could drawdown an extra 841 (621-1,064) Tg CO_2e yr^{-1} by 2030 through further avoided emissions and additional carbon sequestration from growth (mangroves only) and soil carbon sequestration (Figure 5). For example, the maximum mitigation potential from avoided coastal impacts has been estimated for mangrove forests [116], showed that countries in Southeast Asia would have the highest mitigation potential at up to 65 Tg CO_2e yr^{-1} (Figure 5). Considering the estimated global annual emissions from fossil fuels for 2019 and 2020 [117] (36,400 and 34,100 Tg CO_2e, respectively), this equates to potential abatement of ~3% of global emissions (0.5-08% from protection, and 2.3-2.5% from restoration, further details on this estimate are available in the Supplementary Material). Although mangroves and seagrasses contribute equally to avoided emissions potential through the protection of existing habitat (Figure 5), mangroves contribute the largest proportion to mitigation potential from the restoration of disturbed or lost habitats [116] (Figure 5). The substantial contribution of mangroves to mitigation potential through restoration compared to seagrasses and tidal marshes, in part likely reflects our broader understanding of ecosystem loss, and thus the opportunities for restoration in mangroves compared to the other systems. Since we do not understand the full extent of habitat loss for seagrasses and tidal marshes, the potential for seagrass and tidal marsh restoration to contribute to climate mitigation efforts estimated by Griscom et al. [116] are more uncertain. Estimated maximum extent of available habitat for restoration in Griscom et al's [116] analysis ranged from 0.2 to 3.2 million hectares for tidal marshes, and 0.8 to 25.4 million hectares for seagrasses. In contrast, restorable habitat for mangroves is much more resolved with estimates between 9 and 13 million hectares [71]. Such large differences in the estimated restorable habitat for tidal marshes and seagrasses cause potential avoidable GHG fluxes and additional carbon sequestration to vary by orders of magnitude.

An important area for future research is therefore to better estimate where future opportunities exist for BCE restoration. Griscom et al. [116] estimated the restoration potential to climate mitigation based on the estimated total extent of ecosystem loss; however, it is not realistic to assume that all BCE that have been lost can be restored. In cases such as the conversion of mangroves to a port or the conversion of tidal marsh to urban development, it is improbable that these lands can be converted to their natural state. For example, it has been estimated that of the 973,640 ha of mangroves lost between 1996-2016, 17% is not restorable due to urbanization and erosion [71], and possible due to restoration costs. On the other hand, the creation and expansion of new BCE through engineering solutions (including modification of hydrodynamics to restore historical tidal exchange) and plantings (such as mangrove plantations) that allow BCE to grow where historically they were absent, has the potential to add to NDC (assuming plantations do not occur on or convert other BCE), while deriving additional benefits like coastal defence against sea level rise [118]. Thus, future studies should aim to estimate and map the available BCE area that can be restored to its natural state as well as identify potential new areas for the expansion of BCE and the associated carbon fluxes linked to blue carbon projects. Furthermore, the time taken for restored ecosystems to achieve carbon stocks that are equivalent to natural ecosystems, and data on the trajectory of carbon accumulation with time since restoration is limited, with an opportunity for future studies to answer this fundamental question in blue carbon question [119]. Whether restoration and conservation measures are effective for long-term survival depends on a number of contributing factors. To ensure success, restoration policies should be based on globally accepted best practice, with particular attention to site suitability, while taking into account local conditions and the management policies that are relevant to the local community [73,120–124].

Practical aspects of restoration

While conservation of remaining BCE is a highly cost effective way of reducing GHG emissions, restoration requires significant investments and is, therefore, less cost effective [116]. However, both conservation and restoration contribute to adaptation to climate change, and therefore restoration activities can offer opportunities to develop market-based mechanisms that take advantage of existing frameworks for carbon offsets [125]. For example, the application of Verified Carbon Standard (VCS) Methodology for Tidal Wetland and Seagrass Restoration (VM0033) has opened up the opportunity for projects to gain financial help through carbon offsets. Projects can be developed in partnership with those wanting to purchase offsets, whether in the public or private sector, with the opportunity to make additional contributions from other financial streams [30]. In Cispatá Bay, Colombia, 11,000 hectares of mangrove forest are being protected and restored, which will reduce potential emissions by 17,000 metric tons CO_2 within two years [126]. Conservation International, along with Colombia community partners, intend to use the carbon value generated through the project to contribute to a long-term sustainable financing strategy for the region, and Apple© is investing into the project as part of their emissions reduction strategy [126]. Alongside the growing experience of restoration has come the refinement of the VCS methodology, so that projects that follow these procedures should be successful as experience grows. In many cases, restoration can be effective using simple methods such as natural or purposeful re-seeding or replanting that can be very cost-effective.

Restoration of BCE provide other ecosystem service benefits (besides carbon drawdown) that support coastal communities and their economies [9,127] and many open up opportunities for developing payment for ecosystem service (PES) schemes that are complementary to carbon management. PES can provide flexibility in financing and a broader outlook on ecosystem service provision. Moreover, ecosystem service co-benefits from BCE restoration are supported by a wide range of policy goals, for example contributing to multiple of the Sustainable Development Goals, Convention on Biodiversity Aichi targets, Ramsar Convention, UNESCO (Biosphere reserves, Natural World Heritage sites) and others, marine protected areas (MPAs), as well as community goals for development [128,129]. Achieving the restoration scale required to deliver maximum climate change mitigation benefits from BCE (Figure 5), needs strong governance, and support from governments, beneficiary industries, corporations and communities, all committed with a global effort propelled by the UN Decade of Ecosystem Restoration [130].

MPAs and other listed sites can be used to help achieve the Aichi and Durban targets, as well as Sustainable Development Goals (SDG). The Aichi Target 11 was intended to protect 10% of marine areas within national jurisdictions by 2020 and this target has been further embedded under Target 14.1 of the United Nations SDG, while the Durban Action Plan called for an expansion to 30% of global marine protected area [131,132]. A common feature of MPA establishment is to protect and conserve biodiversity, rather than carbon, meaning that some proposed locations for new MPAs might not be able to serve both goals. An economic analysis of the development of MPAs to reach their future targeted expansions concluded that although benefits exceed costs, regardless of the degree of human impact in an area, expansion into more biodiverse low-usage areas would be the most cost effective [133]. In terms of climate change mitigation, use of the same criteria to locate MPA's expansion might not yield the same benefit as projects can only claim carbon credits if they can verify that the carbon stored there would be at risk [134]. In this context, established MPAs might not meet requirements of 'additionality' criteria of existing blue carbon methodologies [135] - that is, projects are generally unable to claim credits for carbon abatement achieved through conservation or restoration actions that are not already prescribed by existing legislation or management requirements. This is an area of current policy debate and development, with approaches for dealing with additionality likely to vary among jurisdictions and accounting mechanisms [136].

Relevance to Nationally-Determined Contributions (NDCs)

Nationally Determined Contributions (NDC) to the Paris Agreement (2016) encourage nations to decide what activities they will undertake to lower their GHG emissions. Currently, blue carbon commitments within NDCs are limited. Recently it has been estimated that 64 countries have included a reference to coastal and marine ecosystems in terms of climate adaptation and mitigation in their NDCs. The majority of these countries list NDC commitments related to mangroves (45), with fewer (10) for seagrass [125]. Tidal marshes are not specifically mentioned, although many countries NDC refer generally to wetlands or commits to using the IPCC Wetland Supplement [137], which provides guidance for estimating GHG emissions and removals for some activities associated with mangroves, seagrass and tidal marshes [125]. A few countries such as the Bahamas and Belize have ambitious, quantified targets for BCE in their NDCs, but many countries still lack any measurable targets [138].

The IPCC Wetland Supplement [137] provides methodologies and default emission factors such that nations can include 1) emissions from a limited range of management activities for conservation and 2) removals through restoration activities. To date, only three countries have reported GHG emissions and removals associated with land-use change of coastal wetlands (Australia, United Arab Emirates and USA). Market-based methodologies are linked to the underlying IPCC guidance, and focus on generating carbon credits for conservation and restoration of coastal wetlands [67,120,135] However, their use is still limited with few pilot projects, mostly within mangroves [120]. The Livelihoods Carbon Funds are partnering with communities in India (The Sundarbans Mangrove Restoration Project), Indonesia (Yagasu) and Senegal (with Senegalese NGO Océanium) to restore and protect mangrove forests. In the Sundarbans 16 million trees, in Indonesia18 million trees, and in Senegal 8,000 hectares have been planted, with the carbon credits generated being used to help repay the project's costs [75]. Together, these initiatives are predicted to sequester 2.7 million tons of CO_2 over 20 years. On a smaller scale, Tahiry Honko (Madagascar) and Mikoko Pamoja (Kenya) have been among the first projects where Plan Vivo has certified blue carbon credits [120]. In Tahiry Honko the conservation and restoration of over 1200 ha of mangrove forest generates over 1300 carbon credits per year, where half of the funds generated goes to help support the local communities. The Mikoko Pamoja project helps conserve 117 ha and has established 10 ha of mangrove forest [120]. In 2019, the project sold 1912 credits with 65% of the revenue going to support community development projects.

A number of countries can benefit from the inclusion of their coastal ecosystems in NDCs. In terms of adoption, this might include small island states with low emissions and small land area relative to the length of coastlines and/or shallow shelf, which could support extensive blue carbon habitats (for example, in Madagascar and Solomon Islands), as well as countries (such as USA and Canada) where there are large opportunities for restoration or habitat creation (including landward retreat with sea level rise). In terms of climate change mitigation, the main beneficiaries would be those countries with extensive, carbon-rich, BCE that have experienced high rates of loss [37]. For example, in Indonesia, conversion of mangroves to shrimp ponds accounts for up to 20% of GHG emissions associated with land-cover change [139]. However, many nations do not have the necessary data to assess either the extent of BCE or carbon stocks, which is particularly the case for tidal marsh and seagrass ecosystems.

Increasing the inclusion of blue carbon within NDCs can be supported in numerous ways (Figure 6). First, the use of existing [67,134] or new frameworks can aid in the identification of blue carbon restoration opportunities at national or subnational scales. In Australia, for example, this process allowed prioritisation of leading opportunities which are now being included within the domestic carbon accounting framework [67]. Resourcing of field data collection and the development of new tools, including spatial data products (such as Jones et al. [140] and Holmquist et al. [141]), is integral to understanding the scope and scale of such opportunities at the national scale. Second, the improvement of standardised protocols for the accounting of GHG benefits [135], and accessibility to low-cost data collection approaches such as remote sensing and affordable field sensors (such as Maher et al. [142]) will enhance feasibility and uptake of carbon crediting projects. Finally, further demonstration projects are needed (but see Wylie et al. [120]), to showcase successes and failures across a broader range of geographic settings, land use and ecosystem types, to build datasets, insights and awareness for future restoration projects [119].

Other considerations

There have been unsuccessful or poorly planned attempts at long-term restoration, which have utilised, non-native or inappropriate species, monoculture without consideration of plant traits and functional diversity and/or unsuitable sites for planting, including planting over other BCE [73,123,143]. The capacity of many BCE plant species to recolonise via vegetative growth and/or transportation of propagules and seeds on the tide, means that passive revegetation techniques (such as improvements in tidal connectivity and water quality, and/or alterations to hydrodynamic energy) can be preferred in many circumstances [67].

Positive steps can be taken to mitigate sea level rise through restoration projects that include a mix of species and at increased densities and scale, while taking into account local hydrology, the potential for landward expansion of mangroves and tidal marshes and relevance to the socio-economic factors affecting the local communities [39,114,144,145]. Global warming can also have an impact; for example, increasing temperatures can result in changing gas fluxes in BCE that lead to enhanced emissions [97,121,146]. The effect of temperature increase on greenhouse gas emissions in seagrass meadows has indicated the potential negative effects of climate change. In 2010/2011 an unprecedented heatwave led to a 22% loss of seagrass cover in Shark Bay (Australia), resulting in an estimated increase in national CO_2 emissions from land use change by 4-21% per annum [97]. Also, using an experimental approach, it has been shown for the first time that increasing temperature up to 37 °C led to a four-fold increase in methane fluxes if compared to the community held at 25 °C from Red Sea seagrass sediments [146]. Increase in temperature is likely to lead to a range of different responses in BCE including an increase or decrease in primary productivity, faster decomposition of organic matter and changes in BCE spatial distribution, but there are still many uncertainties regarding the nature and extent of these changes and how they might differ with latitude and environmental setting [112,121]. Conversely, enhanced removals can occur when mangroves expand into tidal marsh or invasive species colonise tidal marsh and seagrass meadows [147,148147,148]. Biological invasions of this type will require their own type of management actions based on site-specific conservation strategies [104,149].

Future pathways

Restoring and managing BCE for climate change mitigation and adaptation is exciting and feasible [28] - with capacity to drawdown an extra 841 Tg CO_2e pa - but it is complex, as it must take into account a range of issues including social factors, existing policy, variation in geomorphological settings and human impacts, and the impacts of climate change. For example, monitoring and management of restored areas is needed to ensure their permanence, but many projects do not take into consideration sea level rise and/or socio-economic or other human-related impacts in their initial planning [120,121,123]. These complexities are not insurmountable, but they do require further research and planning, including reconsideration of how international policies can better reward long-term carbon drawdown through BCE (as opposed to rewarding cheaper, shorter-term abatement). As we progress toward bringing losses of BCE to a halt, by removing the pressures that led to their losses (for example, ⅓ of seagrass loss in Europe was due to disease, poor water quality and coastal development, de los Santos et al. [90]) and protecting BCE, the goal should shift to one of deploying a sustained effort to restore BCE back to their historical extent [20].

The development and uptake of blue carbon financing mechanisms have been supported in recent years through national and international initiatives, comprising government and non-government groups [125,149]. The incorporation of blue carbon within NDCs in the IPCC Paris Agreement has seen a number of countries explore domestic mechanisms to promote blue carbon projects and include emissions abatement in their national accounts [67,128]. Within the voluntary market, projects have utilised existing forestry-based mechanisms (such as Gold Standard and Plan Vivo) to quantify carbon credits in mangroves [150], while new mechanisms, such as the VCS methodology VM0033 described above, have also been developed specifically for seagrass and tidal wetlands including tidal marshes and mangroves [135,151]. Although such voluntary market projects have commenced in a number of countries, only a very small number have been verified to date [150,152]. Novel approaches such as blended financing (whereby a commercial project provides societal benefits plus financial returns to investors), and/or pairing with co-benefits such as reduction of insurance premiums related to coastal protection [153,154], can help to enhance uptake and progression to verification in coming years. Importantly, there remain divergent views on the application of market-based approaches to quantifying ecosystem service provision to finance environmental protection and restoration, including for coastal wetlands [155,156].

Scientific research has multiple roles in facilitating the development of blue carbon accounting frameworks (Figure 6). Ongoing, original research of carbon cycling in the coastal zone underpins our capacity to quantify and account for emissions abatement achieved at local through to national scales [59,157] . However, further research is required to fill substantial gaps in scientific understanding [112]. These include the need to address geographic biases in the coverage of blue carbon data (including imbalances between the developed - developing nations divide); develop robust information on the scope of small-scale restoration projects at the global or country scale, improved estimates of the influence of disturbances and interventions on greenhouse gas fluxes [119]; and the impacts of climate change and sea-level rise on BCE and the permanence of carbon sequestration [39]. Furthermore, future studies should focus on developing new spatial products with similar resolutions, particularly for seagrasses and tidal marshes, and also assess propagation and uncertainty estimates in global blue carbon estimates to constrain the large variability in the estimates provided in our Review. Therefore, providing more meaningful information for policy and other initiatives aiming to implement blue carbon into climate change mitigation strategies. At the project scale, demonstration is required to generate knowledge and data specific to natural, degraded, restored and created wetlands. Demonstration projects will also help to identify and ameliorate policy and governance roadblocks. These might include issues of land tenure, project boundaries and rights to carbon [158], economic barriers to uptake (such as, high transaction costs), demonstrating additionality, and issues with double-counting of carbon gains [67]. The carbon export and storage beyond the boundaries of BCE, including macroalgae, has been raised as an important but unquantified carbon sink [14]. However, future research needs to overcome scientific and policy barriers that currently preclude the inclusion of carbon export and sequestration beyond BCE extent in blue carbon accounting and abatement associated with management activities.

The success of the blue carbon strategy over the decade since it was first put forward in 2009 [6] reflected in its growing uptake within NDCs and has led to an effort to ascertain additional pathways for climate change mitigation through ocean-based NCS [159]). A growing slate of options is arising, including carbon sequestration from alkalinity release with carbonate dissolution, organic carbon preservation in anoxic conditions within BCE [66,119,160], management of seaweed farms [161,162] and conservation of natural kelp forests [14] for carbon sequestration [161,162], management of sediments beyond BCE to avoid emissions, and rebuilding populations of large marine animals [119]. The success of these emerging blue carbon options requires robust scientific evidence along with accountability of their contribution to greenhouse gas mitigation, including meeting the requirements of additionality and permanence of this benefit. A research road map to increase the scientific evidence supporting blue carbon options was recently put forward [119]. There is an opportunity to further catalyse these actions through synergies between the concurrent UN Decades for Ocean Science (2021-2030) and Ecosystem Restoration (2021-2030) [130], as both robust scientific knowledge and more cost-effective technologies to further restoration efforts are needed. The potential for BCE to offset global emissions (~3% pa, or approximately the combined global emissions from landfill and wastewater sectors [163]), while also providing numerous additional ecosystem services for coastal adaptation and local livelihoods, along with the novel BCE components arising, justifies the attention blue carbon is currently receiving as a nature-based solution to climate change mitigation and adaptation.

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Competing interests

The authors declare no competing interests.

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Figure Legends

Figure 1: Description of blue carbon cycling and its publication timeline: (a) Key elements and processes in blue carbon cycling. 1: BCEs drawdown CO_2 from the atmosphere through

photosynthesis, contributing to net accumulation of organic carbon within plant biomass and sediments – termed 'autochthonous' carbon. 2: BCEs can also accumulate organic carbon from outside sources (such as via terrestrial runoff, plankton), termed 'allochthonous' carbon. Anoxic sediment conditions and the positioning of BCEs in depositional settings (at the interface between land and sea) make them ideal for carbon accumulation. Net carbon drawdown represents 'sequestration', while the pool of accumulated blue carbon is referred to as the 'stock'. 3: Environmental factors such as climate change and coastal development can diminish stored blue carbon stocks by making them more susceptible to microbial attack. Microbial respiration converts organic carbon into inorganic carbon, or greenhouse gases (such as CO_2). This figure was created using symbols from the Integration and Application Network *(ian.umces.edu/media-library)*. (b) Timeline showing the evolution of blue carbon notable papers (classified as 'ISI Highly Cited' papers with 'blue carbon' title as at 3 June 2021, plus one paper from 1981 that first reported the 'blue carbon'term), major moments in international climate change policy and significant events in Blue Carbon policy and management. (1: Smith [164], 2: Nellemann et al. [6], 3: McLeod et al. [13], 4: Pendleton et al. [91], 5: Wylie et al. [120], 6: IPCC [165], 7: Macreadie et al. [119], and 8: Verra method for blue carbon [166].

Figure 2: Co-benefits (or 'ecosystem services') provided by blue carbon ecosystems that contribute to the livelihoods and wellbeing of coastal communities. The figure shows an example from the Mapping Ocean Wealth Australia Program, which has quantified (Australian dollar - AUD) and mapped the value of Fisheries [167–169], Coastline protection [167], and Recreation [167,170] provided by BCE.

Figure 3: Global distribution of mangrove forests (Bunting et al. [35]), tidal marshes (Mcowen et al. [34]) and seagrass meadows (mapped [53] and modelled [32] distributions).

Figure 4: Potential distribution of blue carbon soil stocks (Tg C) per country: a) upper bound and b) lower bound, and blue carbon soil stocks per coastline length (Tg C/km): c) upper

bound and d) lower bound. A full description of the methods and data sources used is available in the Supplementary Material. Note: blue carbon stocks are along the coastline (not inland). Colour coding by country is for display purposes only.

Figure 5: Maximum mitigation potential at country level for avoided coastal impacts in mangrove forests [116], estimated annual loss rates (%) for seagrass [79], tidal marshes [89], and mangroves [36,68,116], global average soil carbon stocks (for mineral soils) per area (95% CI) [137], avoidable flux for avoided coastal wetland impacts pathway (Mg C ha^{-1}) [116], avoidable flux and additional sequestration rates (95% CI) for the coastal wetland restoration pathway (Mg C ha^{-1} y^{-1}) [116], potential area extent available to restoration (Million ha) [116], maximum mitigation potential from avoided emissions due to conversion (i.e., mineralization of carbon stocks in biomass and soils, and loss of carbon sequestration potential; Tg CO_2e y^{-1}) [116] and maximum mitigation potential from restoration (i.e., mineralization of carbon stocks in biomass and soils, and recovery of carbon sequestration potential; Tg CO_2e y^{-1}) extracted from previous studies [116]. Maximum mitigation potential for avoided emissions assumes that 100% of the carbon stocks are lost following habitat loss [116] (1 Mg ha = 1,000,000 ha; 1 Tg = 1,000,000 Mg). Uncertainty estimates in grey bars represent 95% confidence intervals Tidal marshes and seagrass meadows do not have information available on the potential for avoided emissions per country.

Figure 6: Roadmap for incorporating local, national and demonstration project data into carbon accounting frameworks and conservation strategies. This figure was created using symbols from the Integration and Application Network *(ian.umces.edu/media-library)*.