Carbon dioxide removal through mangrove forest (re-)establishment: Key drivers, uncertainties and challenges for long-term success

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Abstract

Mangrove (re-)establishment, or mangrove reforestation and afforestation, is increasingly recognized as a nature-based carbon dioxide removal (CDR) approach with the potential to contribute to climate change mitigation strategies, while delivering substantial ecological and societal co-benefits. However, their effectiveness as long-term CDR strategies depends on a sequence of interdependent decisions and management actions spanning project design, implementation, and monitoring. This review synthesizes evidence on important key factors impacting carbon sequestration outcomes in mangrove reforestation and afforestation and identifies key challenges for robust quantification of CDR potential. Successful mangrove (re-)establishment begins with the integration of stakeholder perspectives, governance frameworks and local ecological conditions into site selection. Failure to account for these dimensions can reduce carbon sequestration potential, as well as storage permanence, and compromise project success. Long-term success further depends on sustained management interventions, including survival monitoring, pest control, tree thinning, and supplementary planting. Across the (re-)establishment pathway, monitoring, reporting, and verification (MRV) remains a critical challenge, particularly due to the limited availability of pre-restoration baseline data of carbon stocks and fluxes. This deficiency introduces substantial uncertainty for estimating net carbon dioxide removal. Although mangrove restoration can achieve sequestration rates of up to 41 Mg CO_2 ha^{-1} yr^{-1} under favorable conditions, strong spatial and temporal variability underscores the need for continuous long-term datasets across diverse tropical regions. We highlight the importance of transdisciplinary collaboration to strengthen empirical evidence, improve process understanding and enhance MRV frameworks. While unlikely to offset hard-to-abate emissions at the scale required as a standalone solution, mangrove reforestation and afforestation represent a valuable component of diversified CDR strategies and provide significant benefits for biodiversity, coastal protection, and local livelihoods.

1. Introduction

Climate change and measures to mitigate its repercussions are among the greatest challenges of modern society. In 2015, 194 countries and the EU signed the historic Paris Agreement with the collaborative aim to limit global average temperature rise to well below 2°C and pursue efforts to further keep temperature increase to a maximum of 1.5°C above pre-industrial levels. To reach this goal, the formulated target would be to continuously reduce greenhouse gas (GHG) emissions by 39 Gt CO_2 equivalent per year to reach climate neutrality or "net zero CO_2 emissions" by 2050 (Smith et al., 2024). However, emission reduction scenarios indicate that the reduction of greenhouse gases alone will not be sufficient to reach this goal, particularly because of industries and industry processes fundamental to modern society, where complete emission reduction is difficult and costly to achieve. For these remaining "hard-to-abate" emissions, the active removal of GHGs via carbon dioxide removal (CDR) measures is commonly considered necessary to annually remove 8.9 Gt of atmospheric CO_2 equivalents by 2050 with a cumulative reduction of 190 Gt CO_2 equivalent (Smith et al., 2024). CDR measures have been estimated to currently remove ~2 Gt of CO_2 annually from the atmosphere (Oschlies et al., 2023). Accordingly, the implementation of CDR needs to expand drastically, and scenarios indicate that large-scale CDR measures have to be deployed by 2030 and scaled rapidly until 2050. Options for the implementation of CDR encompass a broad range of approaches that involve technical and natural means for CO_2 removal, and it is widely agreed that a broad portfolio of CDR measures will be necessary to even out environmental, technical and societal implications that a large-scale deployment might bring. Here, marine-based CDR approaches are so far widely underrepresented, but they are recently gaining more attention, as they may provide opportunities and at times advantages over terrestrial approaches regarding scaling, competitive spatial use and environmental and societal co-benefits.

Coastal ecosystems with dense vegetation, such as mangrove forests, seagrass meadows and salt marshes, sequester and store significant amounts of carbon, referred to as "Blue Carbon." These Blue Carbon ecosystems play a crucial role in natural carbon sequestration, and due to their high carbon storage capacities, measures for their active management and (re-)establishment to perform large-scale CDR are being considered. The majority of Blue Carbon ecosystems contain four primary carbon stocks: (1) aboveground biomass, (2) belowground biomass (roots and rhizomes), (3) sediment (particulate) organic and inorganic carbon and (4) dissolved organic and inorganic carbon present in the sediment porewater. Among coastal Blue Carbon ecosystems and terrestrial forest types, mangrove forests exhibit the highest carbon stocks and exceptionally high carbon uptake rates (Donato et al., 2011; Mcleod et al., 2011; Kauffman, Giovanonni, et al., 2020). Furthermore, mangrove forests provide an array of ecosystem services beyond CDR and carbon storage to local populations, such as protection during storm events, coastline stabilization, provisioning of resources, such as wood and food, and the basis for ecotourism (Worthington et al., 2020).

In this study, we aim to provide an interdisciplinary outline of actions and steps to consider for the real-world implementation of CDR through the reforestation and afforestation of mangrove ecosystems. The basis is a comprehensive, interdisciplinary process chain assessment for mangrove reforestation and afforestation, detailing relevant steps and considerations for implementation in a flow diagram which was developed as part of the "CDRatlas" project. The process chain was developed through a series of expert workshops covering environmental, technical, and societal prerequisites, risks and opportunities for mangrove reforestation and afforestation. We identified six core steps that are essential to the reforestation or afforestation of mangrove forests for CDR: (1) the assessment of environmental and legal feasibility and societal desirability, (2) the preparations for upcoming activities, (3) the execution and maintenance of reforestation or afforestation action, (4) the accounting of additional ecosystem services, (5) the determination of carbon stocks and fluxes for the calculation of CDR potentials and (6) the monitoring, reporting and verification of the CDR footprint (Figure 1). Following these core steps, we will outline relevant considerations, prevailing knowns and current unknowns that are relevant for the safe and sustainable implementation. The detailed process chain can be viewed in Figure S1.

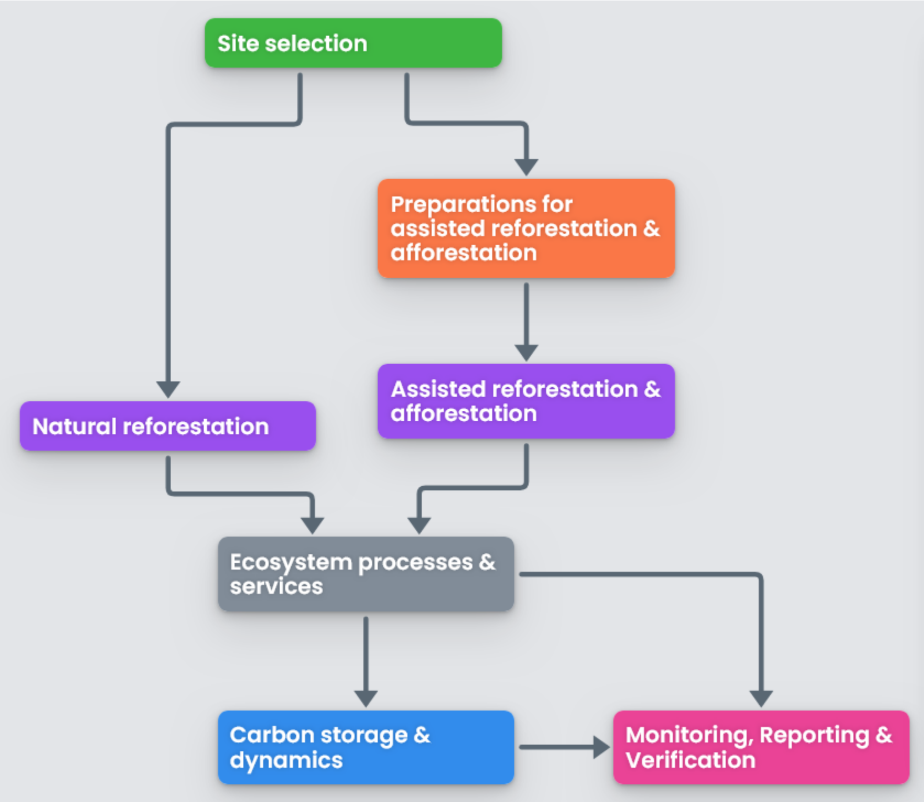

Figure 1 | Main steps for mangrove (re-)establishment that were defined within the process chain. Overarching themes have been defined for the main steps of mangrove (re-)establishment. These themes cover "site selection based on environmental and societal considerations" (green), "project preparations" (orange), "execution and maintenance" (purple), "ecosystem co-benefits" (grey), "carbon uptake and storage" (blue) and "monitoring, reporting and verification" (pink).

2. The assessment of environmental and societal feasibility

In general, two overarching questions need to be addressed prior to the (re-)establishment of mangrove forests in a given area: the environmental feasibility of the measure with regards to the environmental prerequisites, including environmental settings and species requirements, and the societal feasibility and desirability of the measure with regards to the societal needs, conflicts and constraints, including legal, economic, political and ethical aspects from antrophocentric and ecocentric viewpoints.

2.1. Environmental Feasibility of Mangrove Reforestation and Afforestation

The environmental feasibility of mangrove reforestation and afforestation is determined by a wide range of habitat conditions, including hydrodynamics, water parameters, sediment dynamics, climatic patterns and pollution risks. These factors influence not only the survival rate of planted saplings, but also the health and resilience of a new mangrove forest and, in turn, its carbon sequestration and storage potential, making their careful assessment crucial for reforestation or afforestation efforts (e.g., Beselly et al., 2025).

2.1.1. Hydrodynamics

Suitable hydrodynamic conditions are crucial to ensure resilient mangrove forests with sustained high rates of carbon sequestration and storage. Key factors include the tidal regime and wave exposure, which influence sediment dynamics, nutrient availability and plant stability.
Tidal regimes significantly influence the survival and growth of mangroves by driving the frequency and duration of inundation, as well as the exchange of sediment and nutrients. Mangroves typically

occupy the upper intertidal zone, thriving in areas regularly inundated by tidal waters between the mean sea level (MSL) and mean high water level (MHW) (Rogers and Krauss, 2019). For instance, *Avicennia marina* is commonly found between MSL and MHW in open estuaries in Australia (Clarke and Myerscough, 1993). As a result, distinct zonation patterns of the mangrove flora are commonly found along the tidal gradient, irrespective of season, emphasizing the role of surface elevation in their distribution (Ma et al., 2020). Further, different Mangrove species are adapted to specific zones along the intertidal stretch. Thus, effective reforestation or afforestation and management of mangrove forests have to take the intertidal zonation, the species-specific adaptations and distribution patterns into account.

Furthermore, wave exposure and currents also impact the feasibility of mangrove reforestation and afforestation. High wave energy can cause shoreline erosion and sediment resuspension, hindering seedling establishment and potentially uprooting full-grown trees. Thus, high-energy wave environments generally inhibit mangrove establishment, making sheltered intertidal zones more favorable for long-term forest planting activities (Alongi, 2008). Additionally, sheltered environments with moderate wave energy are known to promote sediment retention and organic carbon (OC) burial in coastal blue carbon ecosystems (Mazarrasa et al., 2018). Although direct studies in this field are limited, mangrove forests are typically found in low-energy coastal environments characterized by areas with relatively small wave heights and low energy. A study in a lagoon system in Florida with mangrove communities formed by the three species *Rhizophora mangle*, *Avicennia germinans* and *Laguncularia racemosa* identified critical wave thresholds at 4.2 ± 0.4 cm and 8.0 ± 0.5 cm for 50th and 80th percentile wave heights, respectively (Cannon et al., 2020). This means that if 50% of waves occurring in an area exceed heights of 4.2 cm or 20% of occurring waves exceed heights of 8.0 cm, the probability of mangroves being able to establish and survive in the area is reduced to below 50%. Overall, successful mangrove reforestation or afforestation is strongly linked to site-specific physical conditions, underscoring the importance of carefully evaluating local tidal and wave regimes to ensure long-term ecosystem stability and functionality.

2.1.2. Water Parameters

The chemical and physical properties of water, particularly temperature, salinity and nutrient availability, are critical in preserving the long-term viability of mangrove forests.

Mangrove forests thrive in an average sea surface temperature (SST) range between 24°C and 28°C, which allows maximal productivity and reproductive success (Alongi, 2015; Field, 1995). According to Giri et al. (2011), most of the global mangrove coverage is situated in tropical regions with mean annual SSTs > 24°C. At higher latitudes, SSTs below 15°C have been identified as a critical thermal threshold for mangrove distribution (Cavanaugh et al., 2014). Studies on *A. marina* have shown that seedling growth is strongly reduced at temperatures of 17°C (Steinke and Naidoo, 1991). Together with annual ranges of air temperature, these temperature limitations explain the absence of mangroves in higher latitudes and their sensitivity to cold events. Rising SSTs linked to climate change are facilitating the poleward expansion of mangroves into temperate salt marshes (Cavanaugh et al., 2014; Saintilan et al., 2014). However, excessive warming may increase the frequency of marine heatwaves, which can lead to physiological stress, especially in combination with other factors, such as high salinity or reduced freshwater input (Sippo et al., 2018). In extreme cases, elevated SSTs (>30°C) combined with drought and hypersalinity have been associated with mass mangrove dieback events, such as those documented in northern Australia in 2015 and 2016 (Duke et al., 2017).

Salinity is a less critical environmental factor influencing mangrove distribution and growth, and physiological responses in mangrove ecosystems. Mangroves are adapted to saline conditions, typically thriving in environments with salinity levels between roughly 0 and 35, corresponding to freshwater and seawater conditions. In saline environments, mangroves generally perform better than in freshwater systems. Their energy-intensive adaptations for internal water regulation and salt excretion reduce competition from freshwater-dependent vegetation. Consequently, mangrove populations are very rare in freshwater habitats, where faster-growing plants lacking such costly adaptations generally dominate. Although mangroves can also tolerate salinities on the far opposite end of the spectrum, excessively high salinity (hypersalinity) in the porewater can impose significant stress. Elevated salinity levels above 35 have been shown to reduce stomatal conductance, limit photosynthetic efficiency and decrease biomass production (Dangremond et al., 2015). This reduction is attributed to decreases in stem hydraulic conductivity and an increased vulnerability to xylem cavitation under high salinity stress (Ball & Pidsley, 1995; Nguyen et al., 2015).

Nutrient availability is another factor that can greatly impact the growth patterns of mangrove trees and the overall resilience of mangrove ecosystems. Mangroves typically grow in nutrient-poor sediments (Alongi, 2018; Lovelock et al., 2006; Reef et al., 2010), with nitrogen and phosphorus being the most limited nutrients among them. Fertilization experiments demonstrated that alleviating nutrient limitations can lead to increased growth, with nitrogen or phosphorus additions resulting in increased aboveground biomass and altered biomass allocation patterns compared to unfertilized mangrove stands (Feller et al., 2007). However, excessive nutrient loading, often from agricultural or urban runoff, can result in eutrophication, oxygen depletion, and altered microbial decomposition processes (Hayes et al., 2017; Schile et al., 2017). Additionally, nutrient-enrichment can result in a shift in the above-to-belowground ratio towards lower belowground biomass (Gillis et al., 2019). Such reduced root growth may hamper structural integrity, compromising the resilience and stability of (re-)established stands (Salmo et al., 2013; Gillis et al., 2019).

2.1.3. Sediment Dynamics

Sediment dynamics and composition fundamentally determine the carbon storage capacity and ecosystem stability in mangrove forests, irrespective of being natural, reforested, or afforested. The primary processes and characteristics relevant for CDR are coastal geomorphology, sedimentation and resuspension rates, and sediment grain size distribution.

Mangrove colonization success and persistence have been found to be linked to rates of the prevailing retention and erosion of coastal sediments (Woodroffe et al., 2016). The accumulation of sediment deposits forms the primary substrate on which mangrove forests can best establish and develop. Sedimentation rates in mangrove ecosystems have been reported to range between 0.6 and 60 mm year^{-1}, sustaining constant sediment deposition and a slow increase in forest floor elevation (Van Santen et al., 2007; Chaudhuri et al., 2019; Smoak and Patchineelam, 1999). Through this, mangrove species more tolerant to flooding slowly expand their suitable habitat on the seaward forest side, whilst less flooding-tolerant mangrove species can establish at the landward site. High sedimentation and rising surface elevation are furthermore factors positively correlated with increases in carbon accumulation rates in mangrove sediments (Lovelock et al., 2014). With globally rising sea levels, coastal hydrodynamics may shift over time, potentially rendering areas that once supported healthy mangrove forests unsuitable for their persistence. When changes in hydrodynamic conditions cause erosion rates to exceed sediment accretion rates, coastal sediments erode and the elevation of mangrove forests slowly declines. This reduction in surface elevation can increase inundation stress for mangrove species with limited tolerance to prolonged flooding and likely reduce carbon accumulation rates over time (Lovelock et al., 2014).

Hydrodynamic patterns and flow velocities in coastal systems are largely determined by the prevailing slope. Therefore, the slope of the intertidal zone has been identified as a useful indicator to determine suitable coastal conditions for mangrove forests (Cannon et al., 2020). Gentle slopes below ~1 m rise in elevation over 100 m are more favorable for mangrove establishment as they allow for better sediment deposition, promoting seedling establishment, while steeper slopes lead to increased bed shear stress and strongly reduced sediment stability, reducing the settlement success of seedlings (Gijsman et al., 2024). Further, the slope gradient of the upper intertidal zone can also affect lateral root spread, especially in species adapted to low-energy depositional environments.

2.1.4. Weather and Climate

Climatic conditions significantly influence the productivity and resilience of mangrove forests. Key factors include temperature, precipitation patterns and extreme weather events, all of which shape the overall viability of these habitats. Gouvêa et al. (2022) utilized global modeling to predict mangrove distribution and biomass. Their models suggest that suitable conditions for mangrove establishment and growth are associated with mean air temperatures above ca 12 °C, with optimal biomass production occurring at approximately 26 °C. These results align with other observations that mangrove productivity peaks between 12 to 15 °C and 25 to 30 °C, depending on the species (Hatchings and Saenger, 1987; Inoue et al., 2022). *Kandelia obovata* and *Bruguiera gymnorrhiza* have they respective maximum net photosynthetic rates at 30 °C and 28 °C, respectively (Wu et al., 2026). However, heat stress ≥ 36 °C has been observed to gradually lower photosynthetic activity and thereby reduce CO_2 assimilation rates up until ˜46°C, where CO_2 assimilation rates cease for *R. stylosa* (Cheeseman et al., 1997). In addition, prolonged exposure to temperatures ≥ 34 °C can cause physiological damage, affecting overall plant health and survival (Alongi, 2018). Nevertheless, the occurrence of mangrove populations in some of the world's hottest coastal environments, including along the coasts of the Red Sea and parts of Australia's Northern Territory, suggests that some species may possess higher thermal tolerance than is currently documented. At lower temperatures, cold stress imposes significant constraints on mangrove survival rates. A globally modelled, species-independent evaluation of mean air temperatures in the coldest month identified a threshold just below 12 °C (Gouvêa et al., 2022). However, large differences can be observed between the genera individually. For instance, *Avicennia* species have been observed to tolerate mean monthly air temperatures as low as 8 °C, whereas *Rhizophora* species have their cold threshold at roughly 13 °C (Quisthoudt et al., 2012), defining the latitudinal distribution limits of mangroves on a global scale.

Precipitation patterns dictate freshwater inflow, thus affecting sediment salinity gradients and nutrient delivery to coastal ecosystems. In mangrove forests, precipitation patterns regulate the balance between brackish and saline conditions. According to a global modeling study by Gouvêa et al. (2022), mangrove forests generally require a minimum annual precipitation of approximately 120 mm, with optimal aboveground biomass production observed at around 730 mm per year. However, globally applicable generalizations on the influence of precipitation on mangrove distribution remain difficult, as not only precipitation determines the freshwater availability in a region. For example, precipitation is a minor limiting factor for mangrove distribution in parts of Australia (Gouvêa et al., 2022). In contrast, it strongly constrains growth and distribution extent in semi-arid regions like the Persian Gulf (Martínez-Díaz and Reef, 2023). These findings are supported by historical observations indicating that most mangrove ecosystems are located in areas with 1100-3000 mm of annual rainfall, with the highest species richness present at ~2000 mm (Duke et al., 1998). In general, precipitation requirements can vary by species and region. For instance, *Avicennia marina*, one of the most widespread mangrove species, shows substantial ecological plasticity, with populations established across wide ranges of rainfall and salinity (Martínez-Díaz and Reef, 2023). In areas with low rainfall, such as the Arabian Peninsula, reduced precipitation can lead to increased sediment salinity. This, in turn, poses physiological stress on mangroves and limits tree height, reproductive success and, on the community level, species richness (Osland et al., 2016; Ward et al., 2016). Under such conditions, mangroves may experience stunted growth and forests are vulnerable to transformation into hypersaline mudflats if freshwater inputs decline further (Alongi, 2008; Krauss et al., 2008).

Extreme weather events, such as hurricanes and typhoons, pose significant risks to mangrove forests and may cause defoliation, tree uprooting and large-scale sediment displacement. While some mangrove ecosystems are resilient and demonstrate post-disturbance recovery, repeating extreme weather events can prevent full recovery, leading to long-term structural changes and reduced ecosystem services, including long-term carbon storage and ecosystem stability (Sippo et al., 2018; Amaral et al., 2023). The study by Ward et al. (2016) further emphasizes that the resilience of mangrove forests to cyclones is influenced by factors such as forest structure and prior disturbance history, and Rahman et al. (2024) more recently indicate that functional trait distinctiveness of mangroves is a determining factor for the resilience of mangrove stands.

Ongoing climatic changes add complexities for evaluating the suitability of particular coastal regions for reforestation, afforestation and long-term viability of mangrove forests. While mangroves can withstand moderate levels of disturbance events, the increasing frequency and intensity of such events due to climate change can exceed resilience thresholds, posing significant impacts on mangrove distribution, structure and associated processes and services. Modelled risk indices for mangrove forests under different future scenarios and in situ observations, indicating that increased cyclone activity, stronger storms and hail events can substantially impact these ecosystems and corresponding ecosystem processes (Lima et al., 2023; Hülsen et al., 2025). Further studies suggest that current and future disturbance regimes, induced by climate change, can reduce ecosystem resilience, leading to degradation or loss (Ferreira et al., 2024). As previously discussed in this section, droughts and heatwaves can lead to increased sediment salinity and water stress, resulting in reduced productivity and even tree die-off. In southeast Brazil, extreme drought events have caused significant mangrove dieback, highlighting the potential vulnerability of these forests to prolonged dry conditions (Gomes et al., 2021). Whilst drought events are likely to have predominantly negative impacts on mangrove forests, precipitation increases can potentially have positive and negative effects on ecosystem processes of mangrove forests. On one hand, increased rainfall can reduce salinity stress and increase sediment moisture, promoting mangrove growth and productivity (Jennerjahn et al., 2017). On the other hand, increased flooding and extreme storm events can cause smothering of the aerial root systems of mangroves by large quantities of deposited sediments, leading to higher tree mortality (Adams and Rajkaran, 2021). Furthermore, extreme precipitation can result in flash floods, causing large-scale sediment erosion events along estuarine banks, which can lead to a loss of habitat for mangroves.

Taking these dynamic habitat conditions into consideration is essential for successful reforestation or afforestation and management of mangrove forests (Dunlop et al., 2025), as all of them directly influence the dynamic equilibrium of the ecosystem, its ecosystem processes and ultimately its carbon sequestration potential.

2.1.5. Pollution

In addition to natural habitat conditions, anthropogenic pressures need to be carefully evaluated before starting reforestation or afforestation efforts in a given area. Especially, environmental pollution poses significant threats to the feasibility of mangrove afforestation and reforestation. Various contamination events and sources, including discharged sewage, agricultural runoff, oil spills, as well as toxic trace metals or plastic debris, alter biogeochemical processes and degrade ecosystem health.

Surface runoff from agricultural areas and discharged sewage introduce elevated loads of nutrients, particularly nitrogen and phosphorus, into mangrove ecosystems. Moderate nutrient input may temporarily enhance mangrove growth. However, excessive nutrient enrichment can lead to algal overgrowth on aerial roots, which reduces gas exchange with the atmosphere, weakening the trees and making them more susceptible to disease and dieback (Alongi, 2015; Schaffelke et al., 2005).

Excessive nutrient input inhibits fine-root formation while stimulating the growth of leaves and shoots (Reef et al., 2010). This shift in biomass allocation toward aboveground tissues increases transpiration rates and may reduce resilience to freshwater limitation (Lovelock et al., 2009). Further, pesticide and heavy metal residues discharged into mangrove stands may indirectly affect tree health and growth by altering microbial communities crucial for nutrient cycling (Reef et al., 2010; Erazo and Bowman, 2021).

Oil contamination, originating from tanker or drilling platform accidents, severely affects mangroves by coating aerial roots and leaves, thereby obstructing gas exchange, photosynthesis and toxin uptake, which impair physiological functions and lead to potential widespread tree death (Naidoo, 2025). Further, oil films on the sediment surface affect diverse sediment processes, and oil residues can persist in sediments for years and even decades, with chronic toxicity inhibiting tree regrowth and recovery of affected mangrove stands (Duke, 2016).

Levels of trace metals, namely cadmium, chromium, copper, lead, mercury and zinc, in the environment increase due to human activities. Although mangroves have comparatively high tolerance levels to heavy metal concentrations, when accumulating in mangrove sediments and plant tissues, they can destroy photosynthetic pigments and disrupt enzymatic processes (MacFarlane and Burchett, 2001; Huang and Wang, 2010). The severity of the observed effects varies among species, time of exposure and development stage of the plants, with most frequently stunted growth of roots and aboveground biomass (Yan et al., 2017).

Plastic debris affects mangroves physically and chemically. Large plastic items can smother pneumatophores (aerial roots), reduce soil aeration and damage young seedlings. Microplastics introduce chemical additives and may be ingested by benthic organisms, disrupting food webs and microbial sediment processes and thereby imposing stress on mangrove trees and reducing forest resilience (Maghsodian et al., 2022; Zhou et al., 2024).

In conclusion, anthropogenic pollution poses a major threat to the dynamic ecological equilibrium of mangrove forests. Apart from natural habitat conditions, anthropogenic impact may significantly affect the suitability of environmental conditions for mangroves. While continues pollution is a considerable threat, irregular pollution events, such as oil spills, may cause considerable damage to existing mangrove ecosystems and make otherwise favorable habitat conditions unsuitable for long periods. Therefore, the proximity to and likelihood of such potential pollution events should be taken into account when considering a long-lasting storage of CO_2 via mangrove-based CDR.

2.2. Legal Feasibility of Mangrove Reforestation and Afforestation

The legal feasibility of mangrove reforestation or afforestation measures is shaped by various national and international regulations, policies on habitat modification and land ownership structures. Taken together, these determine the possibilities for implementation of mangrove-based CDR and influence the success and sustainability of such projects.

2.2.1. Land Ownership

Land tenure plays a critical role in mangrove (re-)establishment feasibility, as ownership rights determine access and management authority (Lovelock and Brown, 2019). Mangrove areas may fall under government jurisdiction, communal ownership and private landholdings, each requiring different legal approaches for reforestation or afforestation efforts (De Vries and Rudiarto, 2023). In some cases, unclear land titles or competing claims can complicate project implementation, necessitating stakeholder engagement and legal clarification (Slobodian and Bogaz, 2019). Establishing legal frameworks for land-use agreements and community participation is essential to ensuring the long-term success of such efforts (Zimmer et al., 2022; Sasmito et al., 2023). Overall, addressing land tenure complexities is vital for the effective (re-)establishment and sustainable management of mangrove ecosystems.

2.2.2. Laws and Regulations on Mangrove Planting

Due to their ecological importance and potential role in climate change mitigation, many countries have established legislation for the regulation of the planting (and conservation) of mangroves. For instance, in Zanzibar, the legal framework for mangrove management has developed over time, integrating both traditional practices and formal regulations to address challenges in sustainable mangrove forest management (Mohamed et al., 2023). Regulations may require permits or environmental impact assessments before large-scale planting efforts can proceed, ensuring that such activities do not adversely affect existing ecosystems (Slobodian and Bogaz, 2019). Additionally, some jurisdictions have designated mangrove forests as protected areas, restricting interventions to ensure natural regeneration. International agreements, such as the Ramsar Convention on Wetlands or the Convention on Biological Diversity, further emphasize the need for sustainable mangrove management, including reforestation and afforestation, highlighting the global recognition of the ecological value of mangrove forests (Spalding et al., 2010). Together, these laws and policy instruments form a multi-layered governance framework that is essential for guiding effective, ecologically sound, and socially inclusive mangrove (re-)establishment initiatives.

2.2.3. Regulations on Habitat Modification for Mangrove Growth

(Re-)establishing mangrove forests in degraded or unsuitable habitats may involve modifying site conditions, such as restoring natural tidal exchange patterns or raising the substrate elevation to former levels. These modifications are subject to environmental laws governing land-use changes and coastal engineering activities, which are crucial for ensuring that (re-)establishment efforts do not inadvertently harm existing ecosystems (Datta et al., 2012). In many regions, introducing external materials, such as sediment or artificial structures, requires governmental approval to prevent unintended ecological consequences, as these interventions can significantly alter local hydrology and sediment dynamics (Lewis, 2005). Compliance with these regulations ensures that reforestation and afforestation activities align with broader conservation goals and minimize negative environmental impacts, promoting the long-term success of (re-)establishment projects {Citation}. Furthermore, integrating scientific knowledge with regulatory frameworks enhances the effectiveness of mangrove reforestation and afforestation by aligning ecological objectives with legal requirements (Friess et al., 2016). Overall, a robust legal framework that governs habitat modifications is essential for the sustainable management of mangrove ecosystems.

2.2.4. Legal Framework on Carbon Accounting

The legal framework for carbon accounting is a complex system under continuous development that integrates international agreements, national regulations and voluntary standards to ensure accurate and transparent reporting of GHG emissions and removal. The Paris Agreement, particularly Article 6, establishes mechanisms for international cooperation through carbon markets, facilitating the transfer of emission reductions and CDR between countries to achieve climate goals (Stua et al., 2022). Complementing this, the ISO 14064 standards provide organizations with guidelines for quantifying, monitoring, reporting and verifying GHG emissions and removals, supporting participation in both regulated and voluntary programs (ISO 14064-1, 2018). The Greenhouse Gas Protocol Corporate Standard, developed by the World Resources Institute and the World Business Council for Sustainable Development, offers a globally recognized framework for organizations to measure and report GHG emissions and removals. In the European Union, the Corporate Sustainability Reporting Directive (CSRD) mandates extensive ESG disclosures, including detailed carbon accounting, for large companies and those listed on EU-regulated markets, aligning with international efforts to standardize carbon reporting. Overall, these frameworks collectively contribute to a more unified and effective approach to carbon accounting, essential for tracking progress toward global climate objectives.

2.2.5. Carbon Credits

"Carbon credits" are commonly defined as tradable certificates representing a reduction in emissions or removal of one metric ton of carbon dioxide or its equivalent, generated through activities such as reforestation, land-use management interventions, or more technical CDR approaches. Carbon credits form the foundational unit of both compliance and voluntary carbon markets, enabling emitters to finance mitigation activities outside their own value chains. Despite their widespread use, the environmental integrity of carbon credits depends heavily on the additionality, permanence, and verifiability of the underlying project. Credits that lack robust baseline methodologies, transparent monitoring protocols, or independent third-party verification risk overstating their climate benefits and may not correspond to genuine GHG reductions or removals (Probst et al., 2024). Moreover, challenges such as reversal risks (e.g., fire, disease, or future land-use change), leakage, and inconsistent accounting standards can undermine the credibility of carbon-crediting mechanisms and complicate integration with national and international climate policy frameworks (Romm et al., 2025). Ensuring that carbon credits contribute meaningfully to climate-change mitigation, therefore, requires stringent safeguards, including standardized emission-accounting rules, long-term monitoring commitments, and governance structures that prevent double-counting and ensure real, durable climate outcomes (Haya et al., 2023). For nature-based solutions such as mangrove reforestation and afforestation, understanding how carbon credits are generated, verified, and regulated is essential, as the financial viability and long-term sustainability of such projects often rely on the integrity and credibility of the crediting instruments they produce, as discussed later on.

2.2.6. Grey Carbon Market

The term "grey carbon market" refers to carbon-trading mechanisms that involve carbon credits with questionable environmental integrity or that may not lead to genuine emission reduction or GHG removal. These markets often operate in the voluntary carbon market (VCM), where oversight is less stringent compared to compliance markets, raising concerns about the credibility and additionality of certain offset projects (Kreibich and Hermwille, 2021). Many voluntary offsets lack robust methodologies or long-term monitoring, leading to low-quality credits that may overstate climate benefits (West et al., 2020; Van Dam et al., 2024). In some cases, carbon credits are issued for avoided deforestation or land-use projects that would have occurred without the incentive, thus creating a "non-additional" offset that fails to contribute meaningfully to climate change mitigation (Kollmuss et al., 2008). Furthermore, the potential for double-counting and lack of harmonized verification standards undermines trust in the market and complicates integration with international carbon accounting systems (Schneider et al., 2019). This practice undermines the effectiveness of carbon markets and poses challenges for policymakers seeking to ensure that carbon-trading yields real environmental outcomes. Addressing the issues within the grey carbon market requires enhanced transparency, science-based baselines, third-party audits and stronger regulatory oversight to ensure that offsets deliver legitimate climate benefits (Schneider et al., 2019).

Understanding and navigating these legal aspects and general frameworks is fundamental to the feasibility of mangrove reforestation and afforestation. By adhering to regulatory requirements and addressing land ownership complexities, reforestation and afforestation initiatives can achieve both ecological and legal sustainability.

2.3. Societal Desirability of Mangrove Reforestation and Afforestation

Mangrove reforestation and afforestation present compelling cases for ecological, economic and social benefits, making such measures highly desirable. However, land-use conflicts, livelihood opportunities and various ecosystem services must be carefully assessed to maximize positive outcomes.

2.3.1. Land-Use Conflicts

One of the primary challenges in mangrove forest (re-)establishment is balancing competing land-use interests. Coastal areas where mangroves thrive are often subject to development pressures from agriculture, aquaculture, tourism, or urban expansion (Friess et al., 2016). Notably, shrimp-farming has been a major driver of mangrove deforestation, particularly in Southeast Asia and Latin America, resulting in severe ecosystem degradation (De Lacerda et al., 2021). For example, in Thailand, aquaculture expansion has been identified as the principal cause of mangrove loss over several decades (Chaiklang et al., 2024). Addressing these conflicts requires integrated coastal zone management, stakeholder engagement and effective regulatory enforcement to promote sustainable coexistence between mangrove forests and other land uses (Friess et al., 2016; Jiang et al., 2023). Overall, tackling land-use conflicts through coordinated and inclusive management approaches is essential for the long-term success of mangrove (re-)establishment and conservation.

2.3.2. Co-Benefits and Livelihood Perspectives

Mangrove forests provide a range of ecosystem services in addition to carbon dioxide removal that are crucial to coastal populations. On one side are those less recognized for contributing directly to economic securities, such as enhancement of water quality by filtering pollutants, trapping sediments and reducing nutrient loads, thereby mitigating eutrophication and improving coastal ecosystem health (Alongi, 2002). The dense and complex root structures of many mangrove species additionally stabilize shorelines by anchoring sediments and, thus, preventing erosion, which will particularly become increasingly important upon sea-level rise (Karimi et al., 2022). Furthermore, mangrove forests buffer coastal communities against extreme weather events such as storm surges, floods and tsunamis, significantly reducing damage and loss of life during natural disasters (Das and Vincent, 2009). On the other side, mangrove forests provide benefits directly linked to the economic well-being of coastal communities, such as their role as nurseries for numerous commercially important fish and shellfish species, sustaining regional fisheries and contributing to long-term food security (Nagelkerken et al., 2008) and provisioning of other resources, such as timber, fruits, or medicinal plants. If those resources are harvested sustainably, they can provide communities with essential materials while preserving ecological functions (Walters et al., 2008). Tourism centered around mangrove ecosystems, such as eco-tourism and guided boat tours, can provide sustainable income, particularly in regions with well-preserved habitats (Spalding and Parrett, 2019).

Additionally, the reforestation and afforestation of mangrove forests generate local employment along the (re-)establishment steps and processes, including roles for skilled planting assistants, monitoring staff and nursery operators (Valenzuela et al., 2020). The grey carbon market (see section 2.2.6) can potentially add financial viability, as carbon certification enables communities to trade carbon credits, creating long-term income avenues, while incentivizing conservation (Locatelli et al., 2011). Furthermore, when mangrove (re-)establishment initiatives are aligned with local economic contexts and supported by inclusive governance, they can become powerful tools for poverty reduction and community resilience (Valenzuela et al., 2020), while fostering local stewardship and contributing to sustainable management of the socio-economic system itself (Primavera and Esteban, 2008; Datta et al., 2012; Camacho et al., 2020). Overall, integrating mangrove reforestation and afforestation with local livelihoods ensures that mangrove management is both societally sustainable and economically beneficial. In sum, the multiple ecosystem services provided by mangrove forests underscore their value not only for biodiversity and environmental integrity but also for socio-economic resilience.

3. Mangrove Forests (Re-)Establishment: Natural and Assisted Approaches

Mangrove (re-)establishment efforts can generally be classified into natural and assisted approaches. Natural approaches rely on limited human intervention to restore habitat suitability in areas that receive sufficient seed and propagule influx from connected mangrove forests, thereby enabling natural regeneration. In contrast, assisted approaches involve more intensive human intervention once suitable habitat conditions have been established. These interventions may include raising seeds and propagules in nurseries before transplanting them as seedlings, as well as the direct planting of seeds, propagules, or seedlings to enhance establishment success and reduce the time required for (re-)establishment. Such measures are fundamental in areas where seed and propagule input from connected mangrove forests is insufficient to support natural regeneration.

3.1. Natural Mangrove Forest (Re-)Establishment

Natural mangrove forest establishment relies heavily on ecological processes that enable the colonization of degraded or newly available habitats. Two critical factors that drive the success of natural regeneration are habitat suitability and connectivity to existing mangrove populations (Numbere, 2018; Lewis et al., 2019). Habitat suitability is often compromised or threatened by human activities and needs to be restored and protected in the long term. Hydrodynamic conditions, such as the strength and direction of tidal currents, can influence how far propagules travel and whether they settle in appropriate habitats (Van Der Stocken et al., 2019).

3.1.2. Environmental Suitability and Habitat Improvement

The successful natural reforestation, as well as assisted reforestation and afforestation of mangrove stands, relies heavily on habitat suitability. Key factors such as salinity, tidal inundation, substrate type and hydrodynamics must be thoroughly assessed to determine whether the conditions are conducive to mangrove establishment (Juanico and Salmo, 2015; Gijsman et al., 2024). In situations where these natural conditions are suboptimal, habitat improvement measures, such as sediment stabilization, nutrient augmentation, or hydrological modifications, may be necessary to enhance the site's viability (Ellison et al., 2020; Lewis, 2005; Osland et al., 2020). For instance, porewater salinity and tidal regimes are crucial factors, as many mangrove species are adapted to specific salinity ranges and tidal inundation frequencies (Juanico and Salmo, 2015). Similarly, substrate type, such as the presence of mud or sand, affects root anchorage and nutrient availability (Ellison et al., 2020). If these conditions are not naturally suitable, habitat improvement measures may be implemented to enhance habitat suitability, such as sediment stabilization by introducing materials that retain moisture or increase sediment deposition rates (Lewis, 2005). Hydrological modifications, such as restoring tidal flow or adjusting water salinity, are often necessary to create suitable conditions for seedling survival (Pérez-Ceballos et al., 2020). The restoration of tidal hydrodynamics is particularly vital, as it enables the natural flushing of sediment and nutrients (Lovelock et al., 2022). In many coastal regions, human activities, such as land reclamation and infrastructure development, have altered tidal patterns and sediment dynamics, making it difficult for mangroves to (re-)establish naturally (Ellison, 2021). By assessing environmental conditions and implementing targeted habitat improvement measures, mangrove reforestation or afforestation efforts can be rendered more successful.

3.1.1. Connectivity to existing mangrove populations

Successful mangrove (re-)colonization of suitable habitats additionally depends on the ability of seeds and propagules to travel via tidal and ocean currents (Othman, 1994), as most mangrove propagules are dispersed through hydrochory: buoyant seeds and seedlings are carried by water currents over significant distances. Connectivity facilitates the dispersal of propagules, seeds and seedlings from source populations to areas in need of regeneration. Natural connectivity to healthy mangrove source forests ensures the continuous flow of genetic material, which is crucial for the adaptability of mangrove populations to changing environmental conditions (Zimmer et al., 2022). However, habitat fragmentation can disrupt this connectivity, impeding natural regeneration processes (Thrush et al., 2008). Protection of source forests from degradation can help to preserve the integral connectivity, facilitating natural reestablishment, while ecological services of the connected mangrove forests, such as coastal protection, carbon sequestration and supported biodiversity, are simultaneously sustained (Kodikara et al., 2017). Therefore, maintaining connectivity not only supports natural mangrove forest (re-)establishment but also enhances the overall health and sustainability of coastal environments, while contributing to the sustainability and resilience of coastal socio-ecological systems overall.

3.2. Assisted Mangrove Reforestation and Afforestation

In cases where a natural (re-)establishment of mangrove forests is improbable, assisted reforestation or afforestation becomes an option (e.g., Zimmer et al., 2022). By actively intervening in areas where natural regeneration is unlikely, ecosystem processes and services can be restored. Implementing conservation measures that mitigate threats is essential for facilitating natural forest (re-)establishment. This process involves active habitat modification to improve environmental parameters and facilitate ecosystem recovery. Thus, successful mangrove reforestation and afforestation require a comprehensive understanding of site-specific conditions, including hydrology, sediment composition and historical land-use. Ensuring appropriate hydrological conditions is crucial, as improper water flow can hinder seedling establishment and growth (Dale et al., 2014). Additionally, selecting species well-adapted to the local environment and micro-climate enhances survival rates of seedlings and young trees (McKee et al., 2007). Engaging local communities is vital (Grimm et al., 2024); their involvement provides valuable insights into historical site conditions and ensures sustainable management practices after (re-)establishment (Ravaoarinorotsihoarana et al., 2023). Involving local communities fosters stewardship and ensures that (re-)established forests are maintained and utilized sustainably (Ravaoarinorotsihoarana et al., 2023). Implementing such a collaborative approach not only aids in (re-)establishment but also supports local livelihoods. Furthermore, understanding specific ecological requirements and challenges allows for the development of tailored strategies that increase the likelihood of successful reforestation or afforestation (Zimmer et al., 2022; Gatt et al., 2024). Therefore, a well-planned and executed holistic approach to assisted reforestation or afforestation action not only restores ecological integrity but also provides socio-economic benefits.

3.2.1. Technical Readiness of Mangrove Reforestation and Afforestation

Technical readiness for mangrove afforestation and reforestation is high compared to other CDR approaches. Unlike many novel CDR methods, such as ocean alkalinity enhancement (OAE) or enhanced weathering, mangrove reforestation and afforestation benefit from decades of field experience, codified best practices and increasingly robust institutional and technical infrastructure (Su et al., 2021). Although early efforts often relied on low-technology planting and simple hydrological interventions in rural settings, large-scale programs have since been implemented by governments and NGOs, e.g., in Malaysia, Indonesia and Vietnam, with progressively improved nursery systems, site preparation protocols and monitoring frameworks (Hai et al., 2020; Sitthi et al., 2025). The relative maturity of assisted mangrove forest (re-)establishment is important for its role in climate change mitigation portfolios. While (re-)established stands may take decades to fully match natural mangrove forests in terms of biomass and ecosystem complexity, their contribution to carbon sequestration begins soon after establishment, making them a near-term, low-risk CDR option. Recent modelling work suggests that converting abandoned aquaculture ponds back into mangrove forests could sequester up to 84 Mt CO_2-equivalent (CO_2e) globally, highlighting both the technical feasibility and climate change mitigation potential of scaling reforestation and afforestation (Jiang et al., 2025). Although reforestation and afforestation projects are guided by decades of empirical evidence, allowing for predictable cost estimation, risk mitigation and integration into carbon-accounting schemes (Su et al., 2021), reducing the uncertainty of deployment compared to less-developed methods, large-scale pilot implementations are still needed to validate carbon sequestration and storage potentials, long-term environmental safety, scalability and the effectiveness of MRV (monitoring, reporting, verification) frameworks (Choudhary et al., 2024; Lovelock et al., 2024). In conclusion, mangrove reforestation and afforestation are technically mature, cost-effective and ecologically beneficial methods for removing and sequestering atmospheric CO_2. Its readiness level, scalability and co-benefits position it as a high-priority, no-to-low-risk nature-based solution.

3.2.2. Species Selection

Selecting appropriate mangrove species for reforestation and afforestation according to local environmental conditions is crucial to ensure the success and sustainability of such efforts, since mismatches between selected species and site conditions can lead to high mortality rates and project failure. The selection process should consider both species-specific traits and the regional availability of propagules. Environmental factors such as hydrology, sediment composition and salinity levels play a significant role in determining species suitability for a considered site. For instance, *Rhizophora apiculata* thrives in mid- to high-intertidal zones with soft, muddy substrates (Duke, 2006). Conversely, *Sonneratia alba* and *Avicennia alba* are better adapted to colonize newly formed mudflats due to their tolerance of fluctuating salinity and dynamic conditions (Balke et al., 2011). *Avicennia marina* exhibits high salt tolerance, making it suitable for (re-)colonizing high-salinity environments (Silva and Amarasinghe, 2021). Thus, understanding species-specific environmental requirements and traits is essential for matching species with site conditions. In addition, functional traits of selected species should align with reforestation or afforestation objectives. For example, if carbon sequestration is a primary goal, species like *Kandelia obovata*, enhancing the sediment carbon content more effectively over the long term than, e.g., *Sonneratia apetala* (He et al., 2020), would be most suitable. Combinations of functionally distinct species seem to be more efficient in carbon storage (Rahman et al., 2021) and other ecosystem processes (Rahman et al., 2024) than diverse assemblages of functionally similar species. Furthermore, the availability of healthy propagules is a practical consideration; utilizing locally sourced propagules can enhance genetic diversity and adaptability to local conditions (Hai et al., 2020). Engaging local communities and stakeholders in the selection process can also provide valuable insights into historical species distributions and site conditions, further informing appropriate species selection (Zimmer et al., 2022). Selecting species that do not meet the goals of the project can undermine the intended benefits, such as carbon sequestration or biodiversity enhancement. Therefore, a thorough assessment of site conditions, species traits and the objectives of the reforestation or afforestation measure is imperative.

3.2.3. Desired Trait Characteristics

Proper selection of mangrove species according to their functional response to environmental conditions and effect on ecosystem processes traits can improve the success of reforestation and afforestation projects relative to the initially defined objectives and goals (Pimple et al., 2022): For instance, salinity tolerance (response trait) and root structure (effect trait) differ across species (Reef and Lovelock, 2015; Osland et al., 2017). *Rhizophora* spp., with their extensive prop roots, are particularly effective in dissipating wave energy and reducing coastal erosion, making them ideal for stabilizing shorelines (Pérez-Ceballos et al., 2020). On the other hand, *Avicennia spp.* can colonize a wider range of sediment types and salinity levels and play an important role during early stages of mangrove (re-)colonization, particularly in disturbed or newly formed habitats (Othman, 1994; Giri et al., 2011). Species with high growth rates, broad salinity tolerance or strong root systems not only provide physical protection to coastlines but also promote ecosystem resilience by enhancing biodiversity and supporting the carbon storage capacity of mangrove forests (Alongi, 2015; Othman, 1994). The strategic selection of mangrove species based on their traits ensures that reforestation or afforestation projects meet specific ecological goals and requirements for desired ecosystem services, such as carbon storage (Li et al., 2025). Besides selecting specific trait expressions to match particular requirements for ecosystem services, high functional distinctiveness of mangrove communities improves multiple ecosystem properties and processes (Rahman et al., 2024). Thus, carefully selecting species according to their traits can help mitigate the effects of climate change and improve coastal resilience.

3.2.4. Propagule Availability

The availability of appropriate planting material is a fundamental component of successful mangrove reforestation or afforestation. When selecting propagules from source populations, the risk of overcollection and the maintenance of intraspecific genetic diversity must be considered. Distance and accessibility to source populations influence the viability of propagules during human-mediated transport, with longer distances often resulting in reduced success due to unsuitable or stressful conditions during the transport or storage of propagules. While transporting propagules from distant sources may be necessary in some cases, this must be done with careful planning to ensure their survival during transit and avoid exacerbating the depletion of already vulnerable populations. Unsustainable harvesting practices can deplete natural populations, leading to local extinctions or the loss of local genetic diversity, which compromises the health and resilience of small and fragmented mangrove stands (Mantiquilla et al., 2021). Genetic diversity allows mangrove populations to thrive under highly stressful environmental conditions (Nizam et al., 2022), hence populations with greater genetic variability are more likely to survive and thrive in the face of environmental challenges (Kennedy et al., 2022). Thus, sourcing propagules from multiple, genetically diverse populations reduces the risk of inbreeding and strengthens the ability of the (re-)established mangrove stand to recover from disturbances.

3.2.5. Logistical Considerations

Effective mangrove reforestation or afforestation requires meticulous logistical planning to address the infrastructure, labor and resource management needs that ensure the success of (re-)establishment efforts. First, the availability of suitable nurseries and storage facilities is a prerequisite paramount for propagule acclimatization and maturation. Nurseries have to provide the appropriate salinity and substrate conditions necessary to promote seedling vigor, survival and proper root development, particularly when conditions at the project site are suboptimal (Bayraktarov et al., 2016). The establishment of community-based nurseries offers added value by promoting local engagement and income, which can lead to greater long-term success through sustained support by the local community (Datta et al., 2012; Nguyen et al., 2016). Furthermore, a well-developed transportation network is integral for minimizing stress and mortality during the transfer of propagules. Careful planning of the transportation process, including the use of moist storage, temperature regulation and measures to prevent infections by pathogens or parasites, helps mitigate transplant shock, particularly in species that are sensitive to desiccation and environmental stress (Schmitt and Duke, 2015; Dumroese et al., 2024).

4. Carbon stocks and fluxes and the determination of CDR potentials

To determine the CDR potential of a mangrove reforestation or afforestation efforts at a site, a core set of data on changes in carbon stocks, greenhouse gas emissions, and carbon fluxes is required. In the following sections, we describe each of these components in detail, synthesize the available evidence from the scientific literature, and identify the key knowledge gaps that currently limit robust assessments of CDR potential.

4.1. Stored carbon

The carbon stored in mature mangrove forests is distributed in one of two carbon pools within the ecosystem, which together form the total carbon stock: (I) aboveground and belowground biomass carbon and (II) sediment carbon. Carbon stocks within these pools are differently stable over time. The more stable the sediment carbon stock can last for millennia under intact mangrove forests. The less stable the carbon bound in the living tree biomass can last for the lifetime of the mangrove trees in the range between multiple decades and more than 100 years, depending on the species (Verheyden, 2004; Saintilan et al., 2014). The mangrove carbon stock is estimated at 3,751 Mg CO_2e ha^{-1} (Donato et al., 2011). This results in approximately 55 Pg CO_2e based on the latest global mangrove area estimates of 14,735,899 ha by Bunting et al. (2022). Estimates suggest that approximately 70% of the carbon stored within mangrove forests is buried in sediments and 30% is incorporated in living tree biomass, with about 20% allocated to above-ground and about 10% to below-ground biomass (Hamilton and Friess, 2018). The sediment carbon stock in mangrove forests is primarily composed of particulate organic carbon (POC) from autochthonous sources, such as root growth and the burial of fallen leaves and woody plant parts, or allochthonous sources, such as organic matter from seagrass meadows, macroalgal beds, or the pelagic. Inorganic carbon, mainly in the form of calcium carbonate ($CaCO_3$), is incorporated into the sediment through the remains of corals, mollusk shells and diatoms. Although inorganic carbon can originate from the shells of mangrove mollusks, they usually make up a relatively small portion. When large quantities of inorganic carbon are present in mangrove sediments, it is likely allochthonous material (Saderne et al., 2019).

Carbon stored in mangrove forests is subjected to constant change, with the carbon stock being the net balance of continuously ongoing fluxes exchanging carbon between the pools within the system, other neighboring systems and the atmosphere through a multitude of processes. Carbon accumulation within mangrove forests occurs via two processes: (1) carbon dioxide is sequestered through photosynthesis and incorporated system-internally into plant biomass, or autochthonous carbon and (2) autochthonous carbon shed from the trees in the form of leaves or flowers and carbon washed in by the tides from surrounding habitats, or allochthonous carbon, accumulates gradually in mangrove sediments. Carbon release processes are much more numerous in mangrove systems. Animals and microbes feeding on organic material inside mangrove forests and their sediments respire parts of this consumed matter in the form of CO_2 or other greenhouse gases (GHG) directly back into the atmosphere, whilst degrading other parts into POC and dissolved organic carbon compounds (DOC). The degraded POC, together with other POC, such as leaf litter and woody detritus, and particulate inorganic carbon, such as carbonate minerals, can be flushed out of the mangrove forest by the tides. Similarly, DOC and dissolved inorganic carbon (DIC) can be transported by tidal water movements to neighboring ecosystems. This leads to a continuous cycling of autochthonous carbon within the mangrove ecosystem and an influx of allochthonous carbon and outflux of autochthonous carbon from and to the surrounding habitat. Understanding the dynamic nature of carbon movement between these pools and how it shapes the size of the carbon stocks in the different pools is crucial for evaluating the stability and efficiency of mangrove ecosystems as long-term carbon storage.

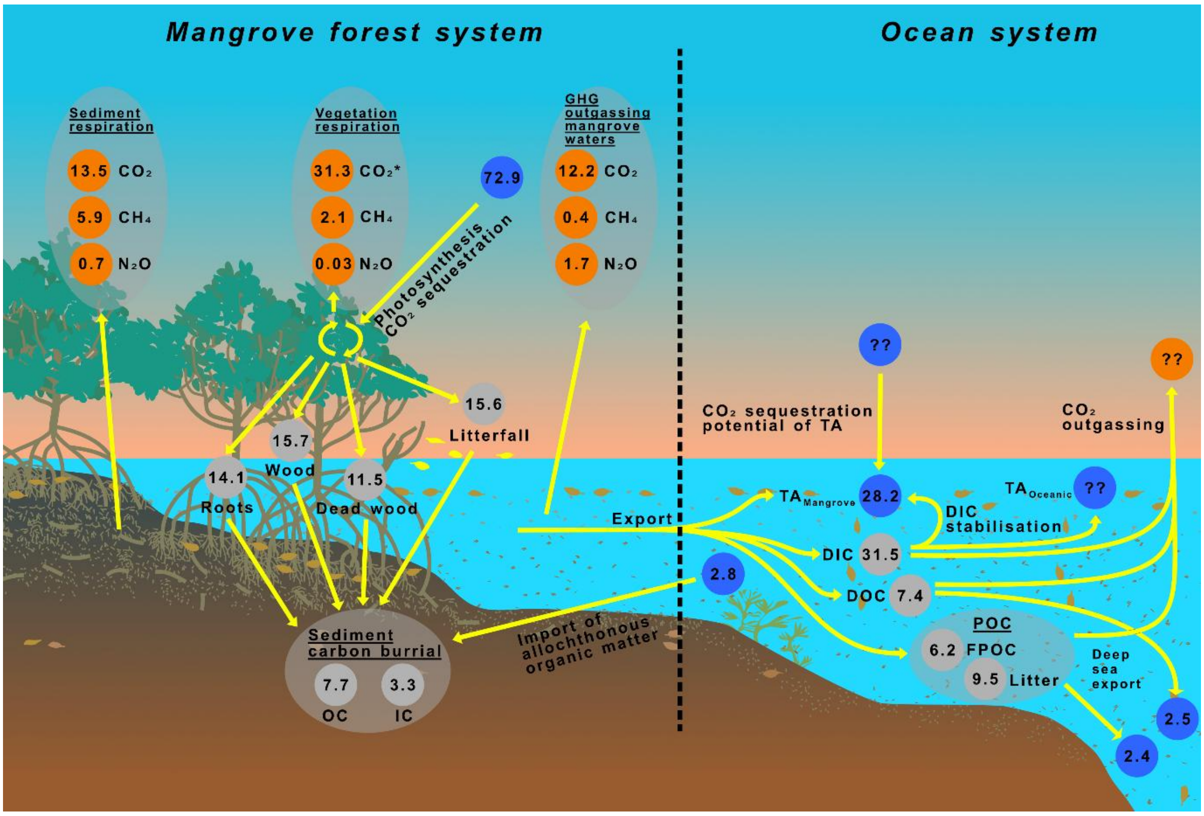

Figure 2 | Global mangrove carbon and greenhouse gas flux exchange (in teragrams of carbon dioxide (equivalent) per hectare of mangrove forest, per year). *Blue circles are sequestrations, orange circles are releases, and grey circles are transformations. OC, organic carbon; IC, inorganic carbon; DIC, dissolved inorganic carbon; DOC, dissolved organic carbon; POC, particulate organic carbon; FPOC, fine particulate organic carbon; TA, total alkalinity; CO_2, carbon dioxide; CH_4, methane; N_2O, nitrous oxide. CO_2*, the respiration value listed here is the combined respiration of mangrove flora and fauna, as no meaningful values for independent flora respiration could be retrieved from scientific literature.*

4.2. Greenhouse Gas Fluxes

Mangrove forests are dynamic ecosystems that continuously exchange greenhouse gases (GHG), such as carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O), with the atmosphere, influencing their net carbon storage rates and, thus, their role in the global carbon cycle. Mangrove forests contribute significantly to global greenhouse gas budgets, with their emissions and sequestration patterns shaped by both natural processes and anthropogenic influences. On average, mangrove trees in mature forests sequester 72.7 Mg CO_2 ha^{-1} yr^{-1} through photosynthesis, often referred to as gross primary production (GPP), with the extremes ranging from 35.6 to 129.6 Mg CO_2 ha^{-1} yr^{-1} (Adame et al., 2024; Alongi, 2025; Gou et al., 2023; Zheng & Takeuchi, 2022). Carbon from the sequestered CO_2 is eventually incorporated into the mangrove's tissues as litter, wood and roots (Table S1). Mangrove species, tree ages, freshwater availability and porewater salinity have been identified as factors driving variation in sequestration rates (Zheng and Takeuchi, 2022; Adame et al., 2024). Although plants take up CO_2 through photosynthesis, their tissues also release CO_2 through respiration to produce energy (Busch et al., 2013). To correct GPP for this CO_2 release, vegetation respiration needs to be subtracted. Such respiration data are mainly acquired using eddy covariance tower systems, which detect CO_2 just above the forest canopy level. This calculated respiration of CO_2 from such systems is often referred to as ecosystem respiration (ER), as it includes the CO_2

released from the flora during dark periods, the fauna and the microbial communities in the sediments (Adame et al., 2024). By subtracting the CO_2 from mangrove sediments, the respiration of flora and fauna can be estimated from ER. The average CO_2 release by mangrove flora and fauna is estimated at 31.3 Mg CO_2 ha^{-1} yr^{-1} (Adame et al., 2024; Alongi, 2022; Gou et al., 2023), balancing the GPP. Results from the few available comparative studies indicate that CO_2 sequestration rates slow down with increasing forest age in both natural and planted mangrove stands (Estrada and Soares, 2017; Sahu and Kathiresan, 2019). This research further revealed that CO_2 sequestration rates are strongly dependent on mangrove species, which has potential implications for CDR-focused mangrove reforestation and afforestation (Sahu and Kathiresan, 2019). However, more studies across planted mangrove are needed to strengthen these findings.

Mangrove forests not only sequester CO_2 from the atmosphere, but also release greenhouse gases. For the release of CO_2, CH_4 and N_2O within mature mangrove systems, two main pathways are known: (1) the release of all three gases by microbial respiration of organic matter in the sediments and waters and (2) the release of CH_4 and N_2O by mangrove tissues. GHG fluxes from microbial respiration in mangrove sediments are influenced by factors, such as tidal regimes, sediment composition, temperature, pH, salinity, sediment redox potential, availability of oxygen, carbon and nitrogen sources, that in turn shape microbial communities and their activities (Allen et al., 2007; Alongi, 2014; Romero-Uribe et al., 2022). Sediment respiration of GHG needs to be accounted for in net CO_2 removal calculations. Across tropical areas, the CO_2 release from mangrove sediments is on average 13.5 Mg CO_2 ha^{-1} yr^{-1} (Adame et al., 2024; Alongi, 2022; Bouillon et al., 2008; Wang et al., 2016). The CH_4 fluxes of mangrove sediments mirror the net balance between methanogenic and methanotrophic microbial communities present in mangrove sediments. Although the methanotrophic microbes can oxidize parts of the methane produced by methanogenic microbes, mangrove sediments release approximately 0.2 Mg CH_4 ha^{-1} yr^{-1} into the atmosphere (Adame et al., 2024; Allen et al., 2007; Alongi, 2020; Liao et al., 2024; Wang et al., 2016). This equates to 5.9 Mg CO_2e ha^{-1} yr^{-1} when recalculated with the CH_4-to-CO_2 conversion factor of 27 as stated in the Sixth Assessment Report of the IPCC (Lee et al., 2023). N_2O fluxes from mangrove ecosystems are smaller compared to CH_4 fluxes, yet also significant. Studies have shown that mangrove sediments release as much as 0.0026 Mg N_2O ha^{-1} yr^{-1} (Allen et al., 2007; Hershey et al., 2021; Liao et al., 2024), or roughly 0.7 Mg CO_2e ha^{-1} yr^{-1} when converted with the IPCC factor of 273 g CO_2 g^{-1} N_2O (Calvin et al., 2023). The release is especially influenced by the availability of nitrogen in the sediments, as artificial nitrogen input through sewage or fertilizer can dramatically increase the release of N_2O into the atmosphere (Kreuzwieser et al., 2003; Wang et al., 2016).

GHGs are released not only at the sediment–atmosphere interface, but also from the water surface, as sediment-derived GHGs can be transported into the water column via porewater exchange, while microbial communities simultaneously respire organic matter within the water column. On a global scale, 12.2 Mg CO_2 ha^{-1} yr^{-1} are released at the surface of water bodies within mangrove forests, such as creeks and estuarine waters (Alongi, 2022; Cabral et al., 2025; Call et al., 2019; Chen et al., 2026; Sippo et al., 2016). Global averages for the release of CH_4 and N_2O from mangrove waters are estimated at 0.4 CO_2e ha-1 yr-1 (Alongi, 2022; Cabral et al., 2025; Call et al., 2019; Fu et al., 2025; based on the IPCC AR6 - non-fossil Methane conversion factor of 27) and 1.7 Mg CO_2e ha-1 yr-1 (Cabral et al., 2025; Fu et al., 2025; Maher et al., 2016; Murray et al., 2018; based on the IPCC AR6 - Nitrous oxide conversion factor of 273). This release of GHG is strongly linked to aerobic and anaerobic decomposition processes in the sediments, which release porewater highly saturated with GHG into the creeks and other water bodies within the mangrove forests and the decomposition of organic matter directly in the water bodies (Bouillon et al., 2008; Chen et al., 2026).

Additional CH_4 and N_2O can be released directly by the mangroves through their tissues. Such emissions can reach levels of 2.1 Mg CO_2e ha^{-1} yr^{-1} for CH_4 and 0.033 Mg CO_2e ha^{-1} yr^{-1} for N_2O

(Gao et al., 2021; Zhang et al., 2022; Liao et al., 2024). The mechanisms and drivers of these fluxes are poorly understood but are suspected to reflect a combination of mangrove vegetation serving as a transport pathway for GHGs by facilitating diffusion out of the soil and the respiration of microbial communities on the outer tissues of mangroves (Liao et al., 2024). The impact of mangrove reforestation and afforestation on GHG respiration needs further investigation, as no articles investigating this topic could be found to the best of the authors' knowledge until today. Although newly (re-)established mangroves likely have increased rates of atmospheric CO_2 sequestration through photosynthesis during the initial growth periods of mangrove saplings (Alongi, 2020), data are urgently needed on the changes of GHG fluxes back into the atmosphere in maturing mangrove forests. The balance between GHG storage and emissions is likely to shift over time as mangrove ecosystems mature, as their release is linked to the organic matter stocks in mangrove sediments, which increase over time (Rosentreter et al., 2018; Song et al., 2023). Therefore, understanding GHG flux dynamics is crucial for constructing accurate carbon budgets for (re-)established mangrove forests and assessing the long-term climate change mitigation potential of these systems (Alongi, 2020). Accurate quantification of these fluxes is essential to inform management strategies and the effectiveness of mangrove reforestation or afforestation projects in storing carbon and mitigating climate change.

4.3. Particulate Carbon Dynamics

Mangrove forests are dynamic ecosystems that engage in continuous carbon exchange with adjacent environments through tidal movements, influencing their carbon stocks. These exchanges involve both particulate organic carbon (POC) and particulate inorganic carbon (PIC) sourced from aboveground biomass and sediment organic matter, thus linking mangrove-derived carbon with adjacent systems through coastal export (Alongi, 2020; Bouillon et al., 2008). POC is separated into fine particles and macro particles, such as leaf litter. In the Asia-Pacific region, fine particulate POC export varies widely, with maximum exports reported in Vietnam at >2 Mg C per hectare mangrove forest and year (ha^{-1} yr^{-1}), while minimum exports were observed in western Taiwan at approximately 1 g C ha^{-1} yr^{-1} (Adame and Lovelock, 2011). These variations are influenced by factors such as tidal range, geomorphology and climatic conditions (Rovai et al., 2018). Globally, fine particulate organic carbon (FPOC) export from mangrove forests is estimated at 1.7 Mg C ha^{-1} yr^{-1} or 6.2 Mg CO_2e ha^{-1} yr^{-1}, emphasizing the significance of lateral carbon exchanges from mangrove ecosystems (Alongi, 2022; Bouillon et al., 2008; Reithmaier et al., 2023). For macro-particulate litter exports, data are scarce, but global estimates are 2.6 Mg C ha^{-1} yr^{-1} or 9.5 Mg CO_2e ha^{-1} yr^{-1} (Lugo and Snedaker, 1974; Boto and Bunt, 1981; Adame et al., 2024). The fate of this exported POC after entering coastal waters is difficult to determine as it is strongly linked to environmental factors, such as the strength and direction of currents, wave and tidal regimes. There is some evidence, however, that decay processes transform a large fraction of the exported mangrove POC into dissolved carbon forms and are integrated into the biomass of other systems (Cabral et al., 2024). From very few studies, estimates of the portion of POC that is incorporated into the sediment of other marine systems, such as the deep sea, reach roughly 15%, amounting to 0.7 Mg C ha^{-1} yr^{-1} or 2.4 Mg CO_2e ha^{-1} yr^{-1} (Maher et al., 2018). While the net fluxes of POC from mangrove forests have been extensively documented, comprehensive studies on how much autochthonous POC is exported versus how much allochthonous POC is imported and eventually buried as part of the mangrove sediments are not well studied. A global assessment found that in marine-influenced mangrove sediments, marine-derived organic carbon constituted approximately 38% of the total organic carbon (Bouillon et al., 2008). In estuarine mangrove systems, terrestrial organic carbon accounts for about 34% of the total sediment organic carbon (Bouillon et al., 2008). Combined with carbon burial rates of averaged 2.1 Mg C ha^{-1} yr^{-1} or 7.7 Mg CO_2e ha^{-1} yr^{-1} globally (Alongi, 2022; Breithaupt & Steinmuller, 2022; Cabral et al., 2025;

Jennerjahn, 2021; Mcleod et al., 2011), this results in a burial rate of approximately 0.8 Mg C ha^{-1} yr^{-1} or 2.8 Mg CO_2e ha^{-1} yr^{-1}, when keeping variations in decay rates of organic matter from allochthonous and autochthonous sources aside. Specific data on PIC fluxes into and out of mangrove forests are even scarcer. Data available on PIC burial rates in mangrove sediments allow to estimate it at approximately 0.9 Mg C ha^{-1} yr^{-1} or 3.3 Mg CO_2e ha^{-1} yr^{-1} on a global scale (Saderne et al., 2019). The accumulated inorganic carbon is likely carbonate material from adjacent marine environments, as only a few mangrove fauna species are capable of large-scale $CaCO_3$ production (Macreadie et al., 2019; Saderne et al., 2019). However, comprehensive studies quantifying PIC fluxes are lacking in the literature, highlighting a gap in the current understanding of carbon fluxes in mangrove ecosystems (Santos et al., 2021). Understanding these POC and PIC fluxes is essential for constructing accurate carbon budgets in fully established and (re-)establishing mangrove forests.

4.4. Dissolved carbon dynamics

Porewater, or dissolved carbon (DC), plays a significant role in carbon cycling within coastal ecosystems. Due to porewater residence times ranging from days to months (Tait et al., 2017), this pool is not considered a stable carbon reservoir for CDR. However, the exchange of mangrove-derived porewater with adjacent marine waters constitutes an important pathway for CO_2 sequestration, storage and release beyond mangrove forest boundaries. Dissolved carbon in porewater occurs in three forms: dissolved organic carbon (DOC), dissolved inorganic carbon (DIC) and total alkalinity (TA). The dissolved carbon-containing compounds can be separated into the fractions of dissolved CO_2 and other dissolved compounds. The proportion of dissolved CO_2 to other dissolved carbon compounds remains obscured in available literature, as not all fractions are determined in scientific studies (Sippo et al., 2016). These other dissolved compounds can be divided into alkalis, such as carbonates, bicarbonates, or hydroxides, that can increase the water's pH or buffer the uptake of acidifying compounds and non-alkalis. The term "total alkalinity" (TA) is commonly used to refer to compounds that form proton-accepting compounds when dissolved in water. In waters with high TA, its buffering capacity can increase the dissolution of CO_2 from the atmosphere into these waters and stabilize it there, resulting in an effective net sequestration of CO_2, which is often overlooked. DOC in mangrove porewater originates primarily from the decomposition of autochthonous and allochthonous organic matter within sediments, but can additionally be derived from root exudation and the leaching of organic matter (Maher et al., 2013; Wölfelschneider et al., 2025; Xia et al., 2026). DIC is formed via aerobic and anaerobic remineralization of organic matter in the sediments (Maher et al., 2013; Reithmaier et al., 2023; Bouillon et al., 2007). Whereas aerobic respiration predominantly produces non-alkaline compounds, anaerobic pathways, including denitrification, sulfate reduction, manganese reduction, and iron reduction, produce both non-alkaline compounds and alkalis, commonly referred to as TA (Reithmaier et al., 2023). However, TA resulting from anaerobic respiration can only stabilize if the reduced remineralization products are removed, i.e., sulfide precipitated as pyrite from sulfate reduction or nitrogen outgassing (N_2) from denitrification (Hu and Cai, 2011). Dissolved CO_2 released from mangrove waters into the ocean amounts to approximately 11.3 Mg CO_2 ha^{-1} yr^{-1} (Sippo et al., 2016; Cabral et al., 2024; Chen et al., 2026). The fate of this exported CO_2 remains shrouded in uncertainties, as it is difficult to predict if it returns to the atmosphere or remains dissolved in the ocean waters stabilized by the exported alkalinity and transported to deep ocean waters, becoming part of the long-term global carbon stocks.
Global estimates approximate that about 2 Mg C ha^{-1} yr^{-1}, or 7.4 Mg CO_2e ha^{-1} yr^{-1}, of DOC are exported through tidal creeks into coastal waters from mangrove forests (Alongi, 2022; Bouillon et al., 2008; Reithmaier et al., 2023; B. Zhang et al., 2026). Roughly 50% of such porewater-derived DOC has been identified to be resistant to photo- and biodegradation (Kristensen et al., 2008). Once exported, about 34% of the total exported DOC eventually reaches the deep ocean as the refractory

fraction, where it remains stable for thousands of years, effectively storing carbon on long timescales (Maher et al., 2018). The non-refractory portion of exported DOC, on the other hand, can fuel microbial CO_2 production, leading to acidification of the coastal waters and partial CO_2 outgassing (Bogard et al., 2020).
The export rates of DIC from mangrove forests to adjacent coastal waters are globally even higher when compared with DOC exports. Global average DIC export from mangroves is estimated at 8.6 Mg C ha^{-1} yr^{-1} or 31.5 Mg CO_2e ha^{-1} yr^{-1} (Alongi, 2022; Cabral et al., 2025; Reithmaier et al., 2023; Sippo et al., 2016). Reithmaier et al. (2023) further indicate that this exported DIC may contribute to the acidification of coastal waters and promote subsequent CO_2 outgassing. In contrast, other studies have linked the export of mangrove-derived DIC to increased phytoplankton growth, indicating a very different fate (Cabral et al., 2024b). Such contradicting findings leave the fate of exported DIC from mangrove forests unclear and require further investigation.
Mangrove-derived TA exports into coastal waters are estimated to reach a global average of 810.000 mol ha^{-1} yr^{-1} (Alongi, 2022; Reithmaier et al., 2023; Saderne et al., 2019; Sippo et al., 2016). This TA primarily occurs as carbonate alkalinity (Reithmaier et al., 2023). Based on research on the sequestration potential of alkalinity in coastal waters, 1 mol alkalinity can sequester 0.79 mol of CO_2 under natural conditions (Fakhraee et al., 2023; Liu et al., 2025). Based on this, the TA exported from mangrove forests can facilitate the inhibition of dissolved CO_2 outgassing from the DIC pool or the sequestration of additional CO_2 taken up from the atmosphere into the ocean waters. In total, the exported TA of 1 hectare of mangrove forest per year is equivalent to 28.2 Mg CO_2. In contrast to exported DOC and the other fractions of DIC, TA can prevail for timescales of millennia in the ocean, making it a long-term sink for CO_2 (Middelburg et al., 2020).
Understanding the transport of DC in its different forms is crucial for evaluating the overall carbon budget of Blue Carbon ecosystems. The movement of DC through tidal flows highlights the connectivity between mangrove forests, estuaries and the open ocean, demonstrating that carbon storage by these ecosystems extends beyond their immediate location.

4.5. CO_2-capture and -storage potentials of (re-)established mangrove forests

Considering all carbon fluxes into and from mangrove forests and combining them with the potential fate of the exported modes of carbon, a mature mangrove forest has the potential of taking up and storing 40.8 Mg CO_2e ha^{-1} yr^{-1} (Figure 2). How this potential develops during the decade-long regrowth of a planted mangrove forest is not yet fully understood and comes with challenges that are not easily overcome. Especially, the time frame of a fully comprehensive study, over the course of 3 or more decades, is not easily realized. Across the tropics, carbon storage rates and stocks of plant biomass and the sediment vary regionally and with stand characteristics, such as tree density and size or species composition (Ahmed & Kamruzzaman, 2021; Alongi, 2012; Gillerot et al., 2018). Based on available data, evidence indicates that the increase in plant above and belowground biomass naturally plateaus when the trees in a mangrove forest reach their growth limits at ages between 20 and 30 years (Alongi, 2018; Song et al., 2023). A similar relationship between stand age and sediment carbon stocks could be revealed in a global comparison of mangrove reforestation and afforestation projects (Song et al., 2023). The one study integrating not only changes in biomass and sediment carbon stock to calculate sequestration potentials for replanted mangrove forests, but a wide variety of additional flux measurements, reaches a value of 11 Mg CO_2e ha^{-1} yr^{-1} 10 years after replanting (Cameron et al., 2019). It has to be mentioned here that some of the growth of one of the investigated sites in the study was extraordinarily low and the upscaling of the results remains uncertain. More basic estimates, only based on changes in biomass and sediment carbon stock, estimate a carbon dioxide sequestration potential of 28 Mg CO_2e ha^{-1} yr^{-1} over a time period of 40 years after replanting (Song et al., 2023).

Although such estimates are more robust and more readily obtainable, GHG release from mangrove ecosystems may offset more than 50% of the CO_2 sequestered by mangrove vegetation (Figure 2). Furthermore, it is estimated that more than half of the carbon released from mangrove forests through tidal waters is transported as dissolved organic and inorganic carbon (Mcleod et al., 2011), and 10% of the organic carbon flowing from land to sea is derived from mangrove forests on a global scale (Dittmar et al., 2006). Hence, we can deduce that substantial amounts of carbon are lost from mangrove ecosystems through tidal outwelling. The fate of this outwelled carbon yet remains relatively obscured, especially for particulate fractions. Most recent research provides evidence that outwelled total alkalinity is likely to cause a substantial drawdown and long-term stabilisation of CO_2 in ocean waters far beyond the borders of the mangrove forests, where it originates from (Saderne et al., 2019; Reithmaier et al., 2023). Both atmosphere-ecosystem carbon exchange and lateral carbon fluxes between ecosystems and adjacent coastal waters are substantially altered by mangrove (re-)establishment, and continue to change as mangrove forests develop and mature (Cameron et al., 2019).

How such potentials vary across mangrove systems worldwide and how accurately the more basic calculations compare to fully comprehensive studies needs to be evaluated when more data from areas with varying environmental impact factors become available in the future. To generate reliable data for a well-founded review in the future, a comprehensive monitoring scheme seems a fundamental requirement.

4.6. Extending the Carbon Dioxide Removal Potential of Mangrove Reforestation and Afforestation

A promising extension of mangrove-based CDR is to adopt cyclical management in which reforestation and afforestation are not a one-off endpoint but the start of ongoing biomass production, harvest and reuse in other CDR pathways or long-term storage. Once a mangrove stand reaches its purported climax or maturity, harvested biomass can feed into processes such as BECCS or biochar, and the reforestation cycle can then begin anew, thereby amplifying cumulative CO_2 uptake via photosynthesis while sediment carbon stocks remain. This strategy could transform mangrove forests from static carbon stocks into dynamic, renewable CO_2-removal systems. The Malaysian Matang Mangrove Forest Reserve has operated under a 30-year rotation scheme with intermediate thinning at 15 and 20 years, culminating in clear-felling for charcoal production (Goessens et al., 2014). In Matang, the density of *Rhizophora apiculata* trees declines from >4,00 stems ha^{-1} at year 15 to less than 2,000 stems ha^{-1} by year 30, partly due to thinning-intervention, illustrating predictable growth and yield cycles (Goessens et al., 2014). While the produced charcoal is used for medical, cosmetic, fuel purposes and leisure products, thus not contributing to long-term CDR, this approach illustrates societal acceptance of mangrove biomass-harvest under planned silviculture (Satyanarayana et al., 2021). However, the emissions cost of converting mangrove biomass to charcoal is nontrivial: about 152.80 Mg C ha^{-1} is estimated to be released during the charcoal production phase alone in the productive area of Matang (Wolswijk et al., 2022). Alternative uses, e.g., engineered biochar or long-term use of the wood for construction, would increase the net CDR potential of cyclic forestry. By coupling mangrove reforestation and afforestation with biomass harvesting and conversion to stable carbon products, the net CO_2 removed over successive cycles could exceed what is sequestered in a climax stage mangrove alone. However, Matang’s long-term yield trends have shown diminishing biomass productivity in more recent decades, indicating risks of overharvest or degradation (Khan et al., 2024). Although such cyclical approaches are not without risks, it highlights the potential of an innovative hybridization of nature-based and engineered CDR, e.g., BECCS or biochar. Linking mangrove reforestation and afforestation with such downstream CDR pathways offers an avenue to amplifying cumulative CO_2 removal beyond a static forest, if critically balancing the carbon lost via process emissions and diminished carbon storage in sediments against the additional uptake from cyclic growth.

5. Monitoring, Reporting and Verifying Mangrove Reforestation and Afforestation

Effective monitoring is crucial for evaluating the success of mangrove reforestation and afforestation efforts and ensuring the correct values of CO_2 removal are being reported. However, the process chain does not end with monitoring and reporting actions. The last crucial step is the verification of such reported data. By systematically tracking key ecological indicators, researchers can assess the long-term sustainability and effectiveness of mangrove reforestation and afforestation projects and CDR measures. Such efforts not only support the role of mangrove reforestation and afforestation projects as Nature-based Solutions for climate change-mitigation through carbon sequestration and storage, but also biodiversity and coastal protection as measures of climate change-adaptation.

5.1. Monitoring net CO_2 Sequestration

Accurate reporting of CO2 sequestration rates is a cornerstone of carbon dioxide removal (CDR) projects, ensuring transparency, accountability and the validation of human intervention. In nature-based solutions, robust reporting frameworks are essential to substantiate climate change-mitigation claims and inform adaptive management strategies (Keith et al., 2021; Calvin et al., 2023). Historically, terrestrial forest (re-)establishment projects have developed comprehensive reporting protocols to document carbon sequestration and storage. For instance, the Woodland Carbon Code in the United Kingdom provides standardized methodologies for measuring, reporting and verifying carbon storage rates, ensuring credibility and consistency in reported data (Eggleston et al., 2006). Similarly, although critically assessed recently, international mechanisms such as REDD+ (Reducing Emissions from Deforestation and Forest Degradation) have established guidelines for reporting emissions reductions achieved through forest conservation and sustainable management practices (Korhonen-Kurki et al., 2013). These frameworks emphasize the importance of standardized reporting to ensure the integrity of carbon offset projects (Bellassen et al., 2015). Mangrove reforestation and afforestation projects can draw parallels from these terrestrial frameworks, but must address unique challenges inherent to coastal ecosystems.
The dynamic nature of the mangrove habitat is driven by the tidal regime and corresponding gradients of salinity and sediment deposition, which complicates the standardization of reporting protocols (Krauss et al., 2008). One significant challenge in reporting for mangrove CDR projects is the potential release of stored carbon back into the atmosphere due to disturbances such as deforestation, land-use change, or extreme weather events (Lovelock et al., 2015). This risk underscores the importance of incorporating permanence considerations into reporting frameworks to account for the longevity and stability of carbon stocks. Moreover, potential conflicts with local communities and Indigenous peoples necessitate the inclusion of societal and environmental safeguards in reporting protocols to ensure equitable and sustainable project outcomes (Korhonen-Kurki et al., 2013; Silori et al., 2013). To overcome these challenges, mangrove CDR projects can adopt several strategies: (1) Developing tailored reporting frameworks that account for the unique characteristics of mangrove ecosystems, including integrating remote sensing technologies and ground-based observations to enhance the accuracy and reliability of reported data (Kuenzer et al., 2011). (2) Engaging local communities in the reporting process can provide valuable insights and promote stakeholder buy-in, thereby enhancing the credibility and sustainability of the projects (Pratihast et al., 2013). (3) Aligning reporting protocols with established standards, such as those outlined in the IPCC guidelines for land

use, land-use change and forestry, can ensure consistency and facilitate the integration of mangrove CDR projects into broader climate change-mitigation strategies (Romijn et al., 2015). In conclusion, while terrestrial reforestation and afforestation projects offer valuable lessons in developing robust reporting frameworks for carbon sequestration, mangrove afforestation and reforestation projects must address specific challenges related to their unique ecological contexts.

5.1.1. Baseline determination of Carbon Stocks and Fluxes

Establishing baseline data on carbon stocks and fluxes as a reference to assess the additionality of mangrove reforestation and afforestation is essential for determining the viability of the CDR measure. Without initial reference measurements, it is impossible to determine the extent of carbon sequestration, exchange and storage over time. Baseline data provide a crucial foundation for evaluating long-term changes in carbon storage and fluxes, allowing researchers to quantify the effectiveness of reforestation or afforestation efforts with respect to additionality.

Mangrove forests are among the most carbon-dense tropical ecosystems, storing carbon in both living biomass and, disproportionately, in their sediments. On average, approximately 30% of ecosystem carbon is stored in plant biomass and 70% in sediments, although the distribution between these pools varies with environmental conditions, stand structure, and species composition (Donato et al., 2011; Hamilton and Friess, 2018; Ahmed and Kamruzzaman, 2021). Carbon sequestration through photosynthesis is counterbalanced by greenhouse gas emissions from plant respiration and organic matter decomposition, while sediment carbon accumulation is facilitated by the burial of both autochthonous mangrove-derived organic matter and tidally imported allochthonous material. Biomass carbon and sediment carbon stocks generally plateau after 20-30 years of stand development (Song et al., 2023). This relationship with stand age is less prominent in sediment carbon stocks and requires more in situ sampling and laboratory analyses for reliable quantification (Alongi & Zimmer, 2024). Carbon stock changes can be more confidently attributed to mangrove (re-)establishment in previously unvegetated areas, whereas restoration of existing vegetated ecosystems complicates attribution. Because sediment carbon accumulation also occurs prior to restoration and may increase through enhanced burial of both autochthonous and allochthonous carbon following mangrove establishment, distinguishing carbon sources is essential for robust carbon accounting and CDR assessments (Thura et al., 2023).

A comprehensive assessment of the CDR potential of mangrove (re-)establishment further requires quantification of atmospheric greenhouse gas exchange and tidal carbon fluxes between mangroves and adjacent coastal waters. Tidal exchange exports dissolved and particulate carbon, which may either be respired back into the atmosphere or contribute to long-term CO_2 uptake (Bogard et al., 2020) and storage beyond mangrove boundaries through processes such as naturally driven ocean alkalinity enhancement and storage in deep sediment and water layers (Sippo et al., 2016; Maher et al., 2018). These fluxes exhibit high spatial and temporal variability, requiring frequent measurements for adequate estimation. Atmospheric emissions of CO_2, CH_4, and N_2O from mangrove forests potentially offset a large portion of vegetation-driven CO_2 sequestration. Likewise, substantial carbon is exported through tidal outwelling, primarily as dissolved organic and inorganic carbon, although the fate of particulate carbon remains poorly constrained (Dittmar et al., 2006; Mcleod et al., 2011). Recent evidence indicates that exported total alkalinity can promote sustained oceanic CO_2 uptake or bind otherwise outgassing dissolved CO_2 far beyond mangrove forest boundaries (Saderne et al., 2019; Reithmaier et al., 2023). Both atmospheric and lateral carbon fluxes are substantially altered by mangrove establishment and continue to evolve as forests mature, making them integral components of the overall CDR potential of mangrove ecosystems (Cameron et al., 2019).

Therefore, baseline data are critical for refining carbon balance estimates and improving management strategies for mangrove reforestation and afforestation. Tracking baselines of carbon stocks enables researchers to distinguish between carbon sequestration driven by tree growth and carbon storage influenced by external factors such as hydrology and sedimentation rates (Twilley et al., 1992). Without reliable reference points, it becomes challenging to differentiate geophysically driven carbon accumulation from changes induced by human interventions, i.e., additionality of CDR, potentially leading to inaccurate climate change-mitigation projections. Furthermore, establishing baseline data enhances the credibility of carbon-offset initiatives and international climate agreements by ensuring that reported carbon sequestration rates reflect actual ecosystem changes (Lovelock et al., 2011). This is particularly important for securing funding for mangrove forest (re-)establishment projects, as carbon sequestration estimates influence financial incentives under mechanisms such as REDD+ (Reducing Emissions from Deforestation and Forest Degradation). Moreover, integrating baseline data with ongoing monitoring helps identify ecological thresholds beyond which mangrove forests may lose their ability to serve as effective carbon sinks, providing valuable insights for adaptive management strategies. In conclusion, baseline data serve as the foundation for understanding the long-term impact of mangrove reforestation or afforestation on carbon dynamics. By establishing reference values for carbon stocks found in the living biomass and the sediments, import and export of particulate and dissolved carbon and greenhouse gas exchange of the system (Figure 2.), practitioners can accurately assess sequestration potential, storage capacity and stock stability. A comprehensive baseline framework ensures that mangrove conservation efforts are guided by reliable, evidence-based assessments, strengthening global climate change-mitigation initiatives.

5.1.2. Tree Growth and Aboveground Biomass Accumulation

Monitoring tree growth contributes to quantifying carbon sequestration rates of mangrove ecosystems in their aboveground biomass (AGB). Tree growth directly influences aboveground biomass accumulation and reflects carbon sequestration. Additionally, regular growth monitoring helps identify environmental drivers of growth rates and allows for optimizing site conditions for maximum AGB accumulation. Measuring tree height, stem diameter at breast height (DBH) and canopy expansion over time provides insight into biomass increase and long-term ecosystem development. In situ measurements of tree height and DBH of at least a representative subset of trees of all present species, as well as their abundance, will translate into species- and region-specific allometric equations to estimate the biomass of individual trees, which are aggregated to determine the AGB per unit area (Komiyama et al., 2005). While in-situ measurements are labor-intensive and time-consuming, they remain the most reliable means of obtaining accurate AGB data. Advanced technologies like LiDAR and drone imagery can supplement such traditional in-situ assessments and allow for more reliable extrapolations over large areas (Sanderman et al., 2018; Doodee et al., 2023). Belowground (i.e., root) biomass (BGB) is often deduced from AGB and DBH. While this method can introduce relatively high uncertainties to BGB estimates and, thus, carbon stocks and sequestration rates, it often remains the option of choice, as direct measurement of root biomass is labor-intensive and often destructive (Adame et al., 2017; Rovai and Twilley, 2021). Only with increasing research efforts on BGB producing more observation-based data can this hurdle be overcome in the future. Until then, estimates of BGB based on AGB and DBH need to be considered with caution when incorporated into the determination of CDR potentials for mangrove reforestation and afforestation efforts.

5.1.3. Sediment Carbon Stock

The vast majority of the carbon stored in mangrove forests is trapped and retained in their waterlogged, anoxic sediments. To evaluate their long-term contribution to climate change mitigation, it is essential to quantify sediment carbon stocks and monitor storage rates. Sediment carbon stocks in mangrove ecosystems are typically assessed through the extraction and analysis of sediment cores to a standard depth of 1 meter, which are sectioned either at regular intervals (Kauffman, Adame, et al., 2020), or according to visible changes in sediment structure or texture, to measure bulk density and carbon content and derive carbon stocks by multiplying carbon content with bulk density and summing over depth (Alongi, 2014). Techniques such as loss-on-ignition and elemental analysis (e.g., using CN analyzers) are commonly applied for determining organic carbon contents (Sanders et al., 2010). It is important to note that sampling designs should account for spatial heterogeneity and variability within a mangrove stand to ensure accurate estimations. The determination of storage rates of sediment carbon over time requires the quantification of sediment accretion rates. Two reliable methods for determining sediment accretion rates are the utilization of marker horizons (e.g., feldspar clay) or sediment elevation tables, which measure vertical changes in the sediment surface over time (Cahoon et al., 2002; McKee et al., 2007). Together, these field-based techniques provide a robust framework for evaluating sediment carbon storage upon mangrove reforestation or afforestation. For both methods, it is crucial to initiate them at the start of the (re-)establishment. Determining sediment chronologies in mangrove forests can also be conducted using radionuclide dating methods, such as the isotopes 210Pb and 137Cs. However, this approach presents several challenges, particularly within the upper layers of sediment where dynamic processes prevail. Bioturbation and physical mixing due to tidal forces are only two prominent issues associated with dating sediments originating from mangrove ecosystems. This mixing can homogenize isotope profiles, making it difficult to distinguish between different sediment layers and leading to potential overestimations of sedimentation rates (Smoak and Patchineelam, 1999). Both 210Pb and 137Cs isotopes can be remobilized within sediments due to chemical changes and porewater diffusion, leading to their vertical redistribution and further complicating the identification of distinct chronological markers (Appleby, 2002).

5.1.4. Particulate Carbon Fluxes

Without accounting for lateral fluxes of organic matter, be it dissolved (see below) or particulate, estimates of net storage rates may be incomplete or inaccurate (Macreadie et al., 2017). Particulate organic carbon (POC) and particulate inorganic carbon (PIC) fluxes are primarily assessed through a combination of hydrological measurements and laboratory analyses of water samples. Accurate measurement of POC and PIC fluxes requires careful consideration of temporal and spatial variability. Tidal dynamics, seasonal changes and anthropogenic influences can all impact carbon fluxes. Methodological challenges, such as ensuring representative sampling during different tidal phases and accounting for rapid changes in water chemistry, are critical for reliable assessments (Alongi, 2014). To measure POC fluxes, researchers often employ the Eulerian method, which involves collecting water samples at fixed stations within mangrove waterways over tidal cycles. These samples are filtered to separate particulate matter, and the collected material is then analyzed for organic carbon content using techniques such as loss-on-ignition or elemental analysis. This approach allows for the quantification of POC and temporal variations in relation to tidal movements (Twilley et al., 1992). Measuring PIC fluxes is more complex due to the dynamic nature of carbonate chemistry in mangrove environments. PIC is typically assessed by analyzing the carbonate content of suspended particulates in water samples, often using acid digestion followed by quantification of released CO_2. However, the accuracy of PIC measurements can be influenced by factors such as varying salinity, pH and the presence of organic matter, which all affect carbonate solubility and precipitation (Maher et al., 2018). Alongside the water samples, a monitoring of flow rates and directions of the water masses is needed. In a last step, acquired particle concentration values need to be related to the flow rates and directions during the monitored period to calculate the direction and extent of the fluxes.

5.1.5. Dissolved Carbon Fluxes

The stock of dissolved carbon in mangrove systems is not stationary and highly dynamic, as it is subject to exchange between the mangrove forests and the coastal water body driven by the tidal pump (Cabral et al., 2024). Therefore, a quantification of the dissolved carbon stock in mangrove forests is nearly impossible. Monitoring of dissolved organic carbon (DOC) and dissolved inorganic carbon (DIC) fluxes is primarily achieved through a combination of hydrological measurements and laboratory analyses of water samples. As for estimations of POC and PIC fluxes, accurate measurements of DOC and DIC fluxes require careful consideration of temporal and spatial variability in mangrove systems. The measurement of DOC fluxes often requires in-situ fluorescence sensors that detect fluorescent dissolved organic matter (FDOM) as a proxy for DOC concentrations. These sensors provide high-frequency, real-time data, which is particularly useful in dynamic environments like mangrove ecosystems. However, the reliability of FDOM as a proxy for DOC can be affected by factors such as variability in DOM composition and water matrix properties like ionic strength and pH. Therefore, calibration with laboratory-based measurements and the use of ancillary water-quality indicators are recommended to improve the accuracy of DOC estimates (Coble et al., 2014). Measuring DIC fluxes involves collecting water samples for laboratory analysis or using in-situ sensors that measure parameters like pH and total alkalinity. These measurements allow for the calculation of DIC concentrations using carbonate system equilibrium equations. However, in systems with low alkalinity or high concentrations of non-carbonate alkalinity, such as those with high DOC levels, these calculations can be less accurate. Therefore, direct measurements of DIC using methods like infrared detection or mass spectrometry are often employed to improve accuracy (Kaltin et al., 2005).

5.1.6. Greenhouse Gas Fluxes

Greenhouse gas (GHG) flux measurements are the last puzzle piece for calculating a comprehensive carbon budget. Two primary methods employed for this purpose are the static chamber technique and the eddy covariance approach. The static chamber method involves placing a sealed chamber over the sediment or water surface to capture GHG emissions over time. This technique is particularly useful for measuring localized fluxes of carbon dioxide (CO_2) and methane (CH_4). Connecting the chambers in situ to a portable gas analyzer allows for real-time detection of gas concentrations over time (Castellón et al., 2022). Static flux chambers coupled with a cavity ringdown spectroscopy analyzer allow the measurement of sediment GHG fluxes during wet and dry seasons (Martin and Moseman-Valtierra, 2015). While these studies highlight the effectiveness of the static chamber method in capturing temporal variations in GHG emissions, the static environment of closed chambers can interfere with natural flow rates and, thus, may not adequately reflect the dynamics of GHG exchange (Riederer et al., 2014). The eddy covariance technique provides continuous, high-frequency measurements of GHG fluxes over larger spatial scales. This method involves installing micrometeorological towers equipped with sensors above the mangrove canopy to measure vertical turbulent fluxes of gases like CO_2 and CH_4. For example, in the Sundarbans mangroves of India, eddy covariance measurements revealed significant diurnal variations in CH_4 fluxes potentially linked to tidal activity (Jha et al., 2014). Another study in a subtropical mangrove forest utilized a decade-long eddy covariance dataset to analyze temporal variations in carbon and water fluxes, providing insights into ecosystem-scale responses to environmental changes (Gou et al., 2023). In summary, both static chamber and eddy covariance methods offer valuable insights into GHG fluxes in mangrove ecosystems. While static chambers are effective for localized, short-term measurements, eddy covariance systems provide comprehensive, long-term data over broader spatial scales.

5.1.7. CO_2 emissions of mangrove (re-)establishment

When estimating the net CO_2 removal achieved through mangrove reforestation and afforestation, greenhouse gas emissions associated with its implementation are often omitted. These emissions arise from activities such as propagule collection, nursery operations, seedling cultivation, transportation, and site preparation. Although they may be relatively small for local restoration projects, their contribution is likely to become increasingly important as restoration efforts are implemented at larger spatial scales. The scale of this challenge is illustrated by ambitious national restoration programmes in countries such as China and Indonesia, which have invested heavily in mangrove restoration over recent decades with established targets ranging from approximately 48,000 to 600,000 ha (W. Hu et al., 2020; Sasmito et al., 2023). Achieving reforestation at this scale requires extensive nursery production and transportation networks, all of which are expected to contribute additional greenhouse gas emissions. However, these emissions remain largely unquantified because comprehensive life-cycle assessments of mangrove reforestation projects are still scarce. Although the importance of integrating life-cycle greenhouse gas emissions and economic assessments into restoration planning is increasingly recognized, this remains a significant knowledge gap (Bodycomb et al., 2026; Su et al., 2021). Addressing this gap is essential for producing transparent and robust estimates of the net carbon removal achieved through mangrove restoration.

5.2. Monitoring beyond CO_2 sequestration potentials

Assessing the success of mangrove forest (re-)establishment requires integrated, transdisciplinary assessments that encompass not only environmental processes but also socio-economic outcomes and compliance with regulatory boundaries. Thus, the responsibility of a comprehensive MRV process is to track a core set of indicators that reflect project success on net carbon removal and, equally important, environmental and ecosystem repercussions and socioeconomic impacts (Su et al., 2021; Swoboda and Oschlies, 2025).

5.2.1. Environmental repercussions

Mangrove reforestation or afforestation are widely promoted nature-based CDR methods, because they generally carry lower risks of disrupting adjacent coastal systems than more intrusive geoengineering approaches (Ellison et al., 2020). Nevertheless, robust environmental monitoring must be integrated throughout the entire project design to detect potential alterations in biophysical and socio-ecological processes across the coastal landscape (e.g., Zimmer et al., 2022). Monitoring designs should begin by defining the ecosystem processes that underpin key regional services, such as fisheries support, nutrient cycling, shoreline stabilization, biodiversity support and cultural benefits, and selecting process-level indicators that directly reflect those services (Su et al., 2021). This service-based approach enables separation of desirable from undesirable changes and facilitates communication with regulators, local stakeholders and private actors. Empirical syntheses show that (re-)established mangrove forests provide measurable ecological and socio-economic benefits, although their species richness, carbon stocks and some other parameters need decades to reach values comparable to natural, old growth stands, making baselines and timelines essential when interpreting trends (Su et al., 2021).

5.2.2. Societal impacts

Ensuring the long-term sustainability of mangrove reforestation and afforestation projects requires systematic monitoring of their socioeconomic impacts alongside their ecological outcomes. This should include assessing the income, employment stability, and working conditions of individuals engaged in restoration activities, such as nursery management and tree planting, as well as evaluating broader effects on household economic resilience. Restoration initiatives that provide fair wages and stable employment opportunities have been shown to enhance local support and contribute to project success (Walton et al., 2006; Datta et al., 2012). Monitoring should also evaluate how landscape changes associated with mangrove restoration, particularly the conversion of degraded areas or other coastal ecosystems such as mudflats, influence local access to natural resources, food security, and traditional resource-use practices (Ebeler et al., 2025). Although restored mangrove forests can enhance fisheries productivity, coastal protection, and other ecosystem services, they may also reduce the availability of land for agriculture, alter patterns of resource access, and modify existing land tenure arrangements (Lyons & Westoby, 2014). To minimize potential conflicts and promote equitable outcomes, restoration projects should incorporate participatory decision-making processes, secure and clearly defined land and resource tenure, and transparent, equitable benefit-sharing mechanisms (Datta et al., 2012). Integrating these social dimensions into monitoring frameworks can improve the social legitimacy, equity, and long-term sustainability of mangrove restoration initiatives.

5.3. Verification in CDR Projects: Insights from Wetland to Mangrove Reforestation and Afforestation

Verification ensures that reported CDR rates are accurate, additional and permanent. In nature-based solutions, such as wetland and mangrove forest (re-)establishment, robust verification processes are essential to validate climate change-mitigation efficiency and maintain the integrity of carbon markets (Griscom et al., 2017). Without effective verification, there is a risk of overestimating carbon sequestration, undermining the credibility of CDR projects (Bellassen et al., 2015). Historically, wetland (re-)establishment projects have been incorporated into voluntary carbon markets, often referred to as the "grey carbon market" (Sapkota and White, 2020). Yet, challenges have arisen in verifying actual carbon sequestration and storage rates due to variable environmental conditions and long-term carbon stability concerns (Ouyang and Lee, 2014). While restored wetlands can sequester substantial amounts of carbon, factors such as methane emissions from anaerobic decay must also be accounted for to ensure net climate change-mitigation benefits (Poffenbarger et al., 2011). Mangrove reforestation or afforestation projects share similarities with other wetland initiatives but present unique verification challenges due to the complex coastal environments (Alongi, 2020). Tidal fluctuations, latitudinal gradients, or climatic dynamics complicate the accurate measurement of carbon sequestration and storage in these ecosystems (Sanders et al., 2016). Additionally, the potential for carbon loss through storm events or land-use changes poses risks to the permanence of carbon stocks (Rogers et al., 2019). Ensuring that carbon credits from mangrove forest (re-)establishment projects are both additional and permanent requires meticulous verification methodologies. The Verified Carbon Standard (VCS) and the Intergovernmental Panel on Climate Change (IPCC) provide guidelines for evaluating the environmental integrity of carbon offset projects, ensuring that they meet rigorous standards (Bellassen et al., 2015). To address these challenges, a range of complementary solutions has been proposed. The development of standardized methodologies specifically designed for mangrove ecosystems can improve the accuracy and comparability of carbon sequestration and storage estimates, ensuring that carbon credits are issued on the basis of reliable data (Donato et al., 2011). Advances in monitoring technologies, including the use of remote sensing, satellite imagery and unmanned aerial vehicles, further enhance precision by enabling quasi-continuous observation and facilitating the early detection of ecological changes that may affect sequestration or storage rates (Simard et al., 2019). Finally, the active involvement of local communities in monitoring and verification processes can reinforce oversight while aligning project outcomes with local conservation and development priorities, thereby increasing both sustainability and social acceptance (Boissière et al., 2017).

6. Conclusions

While mangrove reforestation and afforestation presently constitute a relatively minor CDR pathway compared with their terrestrial counterparts, they may nonetheless provide a meaningful contribution to portfolios addressing residual and hard-to-abate emissions. Achieving long-term carbon sequestration outcomes is shaped by informed decision-making made during the initial stages of project development.

Successful mangrove (re-)establishment efforts start with the selection of a fitting project site. It is crucial for an integral selection to incorporate the needs, values and perspectives of all relevant actors, governance frameworks and also the environmental settings related to a potential site that determine not only the suitability of potential mangrove species, but also influence the overall CDR potential generated and its long-term storage potential. Disregarding either of these aspects is likely to result in reduced CDR potential or even the complete failure of the effort. Downstream efforts of the outlined process chain, including maintenance activities, such as survival monitoring, thinning, pest control, or potential secondary planting events, are as important to ensure a successful long-term CDR as initial evaluations. Successful mangrove reforestation and afforestation efforts are much more likely to have such downstream measures integrated than unsuccessful ones. All these subsequent and interlocked steps in the outlined process chain build towards the monitoring of the CDR potential of mangrove (re-)establishment efforts.

One aspect of adequate MRV at the downstream end, we want to emphasize, is the determination of comprehensive baseline data. Only with such data can CDR potentials be calculated accurately. To our knowledge, until this date, baseline determinations of carbon fluxes and initial carbon stocks are rarely, if at all, done before the start of mangrove (re-)planting efforts. This results in a large uncertainty on what preexisting carbon fluxes were at the site prior to the actual (re-)establishment efforts. Mangrove forests (re-)establishment, as other nature-based CDR strategies, requires time to develop. In combination with the highly heterogeneous environments, our global estimate of the CDR potential reaching up to 41 Mg CO_2 per year per hectare of (re-)planted mangrove forest can be fitting for some areas, but vastly differs from reality in others. The currently available evidence highlights a critical need for the generation of long-term time series datasets from a broader range of mangrove restoration and afforestation sites across tropical regions. Such data collection should commence at the onset of restoration activities and be coupled with sustained monitoring over ecologically relevant timescales. We argue that these efforts require close collaboration between researchers and practitioners, fostering a transdisciplinary framework that can both strengthen the empirical basis for global-scale assessments and advance process-level understanding of carbon dioxide removal (CDR) dynamics. Improved observational records would also enhance the robustness of monitoring, reporting, and verification approaches necessary for the quantification and validation of carbon sequestration outcomes.

Ultimately, mangrove reforestation and afforestation are unlikely to offset hard-to-abate CO_2 emissions at the scale required as a standalone CDR strategy. However, as part of a diversified portfolio of carbon dioxide removal approaches, they can contribute to the management of residual emissions following substantial reductions in fossil fuel and land-use emissions. Beyond their carbon sequestration potential, restored mangrove ecosystems provide numerous co-benefits, including coastal protection, biodiversity conservation, and the provision of ecosystem services that support local livelihoods. These multiple societal and ecological benefits, combined with the operational maturity of restoration approaches, make mangrove reforestation and afforestation a promising nature-based CDR option for near-term implementation.

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Supplemental material

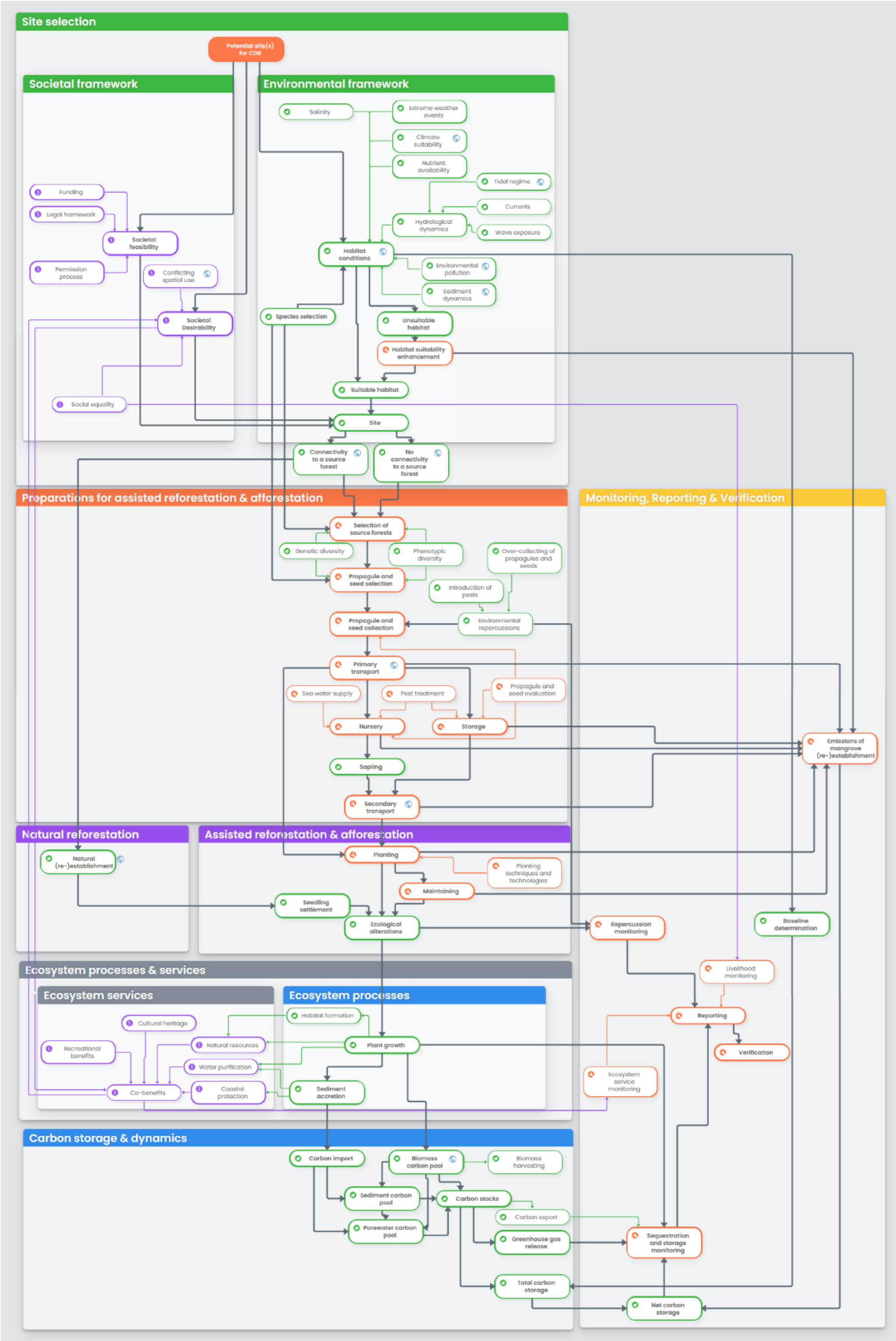

Figure S1 | Detailed steps for mangrove (re-)establishment that were defined within the process chain. Overarching themes have been defined for the main steps of mangrove (re-)establishment. These themes cover "site selection based on environmental and societal considerations" (green), "project preparations" (orange), "execution and maintenance" (purple), "ecosystem co-benefits" (grey), "carbon uptake and storage" (blue) and "monitoring, reporting and verification" (pink). Thicker dark grey lines indicate the core path of the process chain and thinner, coloured lines indicate explanatory inputs for considerations within the core path.

Table S1 | Global carbon and greenhouse gas fluxes across scientific studies (in teragrams of carbon dioxide equivalent per hectare of mangrove forest, per year).

GPP	**Litter production**	**Wood production**	**Root production**	**Dead wood production**	**Sediment respiration**			**Tree respiration**			**Reference**
					CO_2	**CH_4**	**N_2O**	**CO_2***	**CH_4**	**N_2O**	
66.1	17.1	14.6	11	11.5	14.3	1.4	-	31.5	-	-	Adame et al., 2024
-	-	-	-	-	-	10.3	0.4	-	-	-	Allen et al., 2007
-	14.2	12.4	17.4	-	22.4	0.54	-	29.3	-	-	Alongi, 2022
80.3	-	-	-	-	-	-	-	-	-	-	Alongi, 2025
-	-	-	-	-	-	-	-	-	-	-	Boto & Bunt, 1981
-	15.6	15.4	18.8	-	-	-	-	-	-	-	Bouillon et al., 2008
-	-	-	-	-	-	-	-	-	-	-	Breithaupt & Steinmüller, 2022
-	-	-	-	-	-	-	-	-	-	-	Cabral et al., 2025
-	-	-	-	-	-	-	-	-	-	-	Call et al., 2019
-	-	-	-	-	-	-	-	-	-	-	Chen et al., 2026
-	-	-	-	-	-	-	-	-	-	-	Fu et al., 2025
-	-	-	-	-	-	-	-	-	3.3	-	Gao et al., 2021
72.7	-	-	-	-	-	-	-	32.6	-	-	Gou et al., 2023
-	-	-	-	-	-	-	1.8	-	-	-	Hershey et al., 2021
-	-	-	-	-	-	-	-	-	-	-	Jennerjahn, 2021
-	-	-	-	-	-	1.2	-	-	0.01	0.03	Liao et al., 2024
-	-	-	-	-	-	-	-	-	-	-	Lugo & Snedaker, 1981
-	-	-	-	-	-	-	-	-	-	-	Maher et al., 2016
-	-	-	-	-	-	-	-	-	-	-	McLeod et al., 2011
-	-	-	-	-	-	-	-	-	-	-	Murray et al., 2018
-	-	-	-	-	-	-	-	-	-	-	Reithmaier et al., 2023
-	-	-	-	-	-	-	-	-	-	-	Saderne et al., 2019
-	-	-	-	-	-	-	-	-	-	-	Saderne et al., 2021
-	-	-	-	-	-	-	-	-	-	-	Sippo et al., 2016
-	-	-	-	-	7.6	16.2	-	-	-	-	Wang et al., 2016
-	-	-	-	-	-	-	-	-	1.6	-	Zhang et al., 2022
71.9	-	-	-	-	-	-	-	-	-	-	Zheng & Takeuchi, 2022

Table S1 continued

Water respiration			Soil accumulation		TA	Litter	FPOC	DOC	DIC	Reference
CO2	CH4	N2O	OC	IC						
-	-	-	-	-	-	3.6	2.2	-	-	Adame et al., 2024
-	-	-	-	-	-	-	-	-	-	Allen et al., 2007
12.1	0.43	-	5.9	-	41.8	-	7	14	91	Alongi, 2022
-	-	-	-	-	-	-	-	-	-	Alongi, 2025
-	-	-	-	-	-	13.2	-	-	-	Boto & Bunt, 1981
-	-	-	-	-	-	-	5	5.5	-	Bouillon et al., 2008
-	-	-	5.1	-	-	-	-	-	-	Breithaupt & Steinmüller, 2022
4.9	0.03	-0.07	10.6	-	-	-	-	-	13.2	Cabral et al., 2025
28.7	0.05	-	-	-	-	-	-	-	-	Call et al., 2019
10	-	-	-	-	-	-	-	-	-	Chen et al., 2026
-	1.2	7.1	-	-	-	-	-	-	-	Fu et al., 2025
-	-	-	-	-	-	-	-	-	-	Gao et al., 2021
-	-	-	-	-	-	-	-	-	-	Gou et al., 2023
-	-	-	-	-	-	-	-	-	-	Hershey et al., 2021
-	-	-	8.6	-	-	-	-	-	-	Jennerjahn, 2021
-	-	-	-	-	-	-	-	-	-	Liao et al., 2024
-	-	-	-	-	-	11.6	-	-	-	Lugo & Snedaker, 1981
-	-	-0.2	-	-	-	-	-	-	-	Maher et al., 2016
-	-	-	8.3	-	-	-	-	-	-	McLeod et al., 2011
-	-	0.03	-	-	-	-	-	-	-	Murray et al., 2018
-	-	-	-	-	10.6	-	6.6	5.1	11	Reithmaier et al., 2023
-	-	-	-	3.3	-	-	-	-	-	Saderne et al., 2019
-	-	-	-	-	52.8	-	-	-	-	Saderne et al., 2021
5.3	-	-	-	-	5.3	-	-	-	9.3	Sippo et al., 2016
-	-	-	-	-	-	-	-	-	-	Wang et al., 2016
-	-	-	-	-	-	-	-	5.3	-	Zhang et al., 2022
-	-	-	-	-	-	-	-	-	-	Zheng & Takeuchi, 2022

Notes: CO2*, the respiration value listed here is the combined respiration of mangrove flora and fauna, as no meaningful values for independent flora respiration could be retrieved from scientific literature; CH4 was converted into CO2e based on the IPCC AR6 – non-fossil Methane conversion factor of 27; N2O was converted into CO2e based on the IPCC AR6 – non-fossil

Methane conversion factor of 273; GPP value from Zheng & Takeuchi, 2022 was averaged based on determined and reviewed values from this study, as all research was conducted in the same area. Abbreviations: OC, organic carbon; IC, inorganic carbon; DIC, dissolved inorganic carbon; DOC, dissolved organic carbon; POC, particulate organic carbon; FPOC, fine particulate organic carbon; TA, total alkalinity; CO2, carbon dioxide; CH4, methane; N2O, nitrous oxide; GPP, gross primary productivity.