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An ecosystem of carbon dioxide removal reviews – part 2: CO_2 removal *via* blue carbon ecosystems

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Carbon dioxide removal (CDR) is essential for achieving global climate goals, such as those outlined in the Paris Agreement. This systematic review investigates the current state of CDR through management pathways for blue carbon ecosystems (BCEs) and assesses their role in the global CDR portfolio. This article analyzed 2622 peer-reviewed and gray literature articles out of 13 859 identified using a machine learning-assisted review process published through November 2025. The review focuses primarily on mangroves, seagrass meadows, salt marshes, and macroalgae. These ecosystems collectively sequester approximately 270 MtC year^{-1} (106–516) to various long-term sinks, with macroalgal export to deep ocean environments and dissolved inorganic carbon (DIC) pools contributing the most. BCE conservation and restoration offer potential CO_2 emission reductions between 60–96 MtC year^{-1} and removals between 21–448 MtC year^{-1} by 2050, though these estimates remain uncertain and vary widely in cost and feasibility. The study highlights significant gaps in integrating BCEs into climate policy, particularly regarding their representation in national inventories, carbon accounting frameworks, and cost assessments. It emphasizes the need for multi-benefit strategies that balance carbon sequestration with local livelihoods and ecosystem co-benefits. Translating these insights into practice will require that climate policy frameworks explicitly incorporate blue carbon pathways through combining carbon sequestration with financial reward systems and additional incentives to realize co-benefits and interaction effects that would also increase social acceptance.

Broader context

As the global community moves toward net-zero targets under the Paris Agreement, it is increasingly clear that large-scale deployment of carbon dioxide removal (CDR) methods will be required. Among the various CDR methods, blue carbon ecosystems (BCEs) – mangroves, seagrass meadows, salt marshes and macroalgae – have emerged as a promising natural climate solution that deliver both carbon sequestration and important co-benefits such as enhancement of marine biodiversity. BCEs can store carbon over long timescales within their marine systems through plant biomass production and long-term burial of organic carbon in sediments. Scientific research on blue carbon has expanded rapidly. Still, it partially remains fragmented across disciplines, regions, and methodological approaches. Therefore, there is a need to comprehensively and systematically structure the existing synthesis on BCEs, integrating different research strands and findings. Questions persist around robust carbon sequestration and removal potential estimates, synthesizing often unclear and uncertain techno-economic feasibility estimations and assess the effective integration into national carbon accounting frameworks, carbon markets or climate policy instruments. This review provides a systematic synthesis of blue carbon literature, one that not only evaluates the state of scientific knowledge but also connects it to policy relevance and implementation challenges.

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1. Introduction

Ambitious climate policy in line with the Paris Agreement will require carbon dioxide removal (CDR) on a gigaton (Gt) scale in addition to deep reductions in greenhouse gas (GHG) emissions by the middle of the century and beyond.[1] Gt-scale CDR is currently only being achieved through afforestation.[2] Accordingly, various further CDR methods need to be utilized to achieve net-zero or even net-negative carbon emissions. Different CDR methods offer different potential, but also different co-benefits and side-effects, depending on how and where they are applied.

A CDR method that has gained particular scientific and policy traction is blue carbon.[3] This term was first introduced in 2009 and initially used in reference to biological carbon sequestration in all marine ecosystems.[4] In the gray literature in particular, this term is sometimes defined so broadly that it includes numerous ocean-based CDR methods. However, a more narrow and precise definition has since been introduced focusing on blue carbon ecosystems (BCEs) that have appreciable rates of carbon sequestration, can store carbon over long timescales relevant to climate mitigation (*i.e.*, >100 years) and can be managed for additional benefits.[5,6] Lovelock and Duarte (2025) examined 15 coastal ecosystems and applied five criteria to assess the extent to which they can be considered BCEs. The 15 potential ecosystems evaluated as BCEs include mangroves, tidal marshes, seagrass meadows, high intertidal salt flats and sabkhas, supratidal forests, macroalgae, phytoplankton, coral reefs, marine fauna, shellfish reefs, low intertidal mud flats, coralline algae, polar zoobenthos, marine sediment, and rocky reefs.[7]

According to their classification, only mangroves, salt marshes, and seagrass meadows meet all criteria for BCEs,[5,7] with several other ecosystems considered as emerging BCEs. The ''traditional'' BCEs sequester carbon primarily through plant biomass production and the long-term burial of organic carbon in anoxic sediments, where decomposition rates are slow. However, in the IPCC greenhouse gas and national inventories, mangroves and other coastal ecosystems, which also include above-ground carbon storage, are accounted for in the category forest. They are not displayed separately as BCE contributions. On the voluntary carbon market (VCM), the blue carbon category is dominated by mangrove projects because they can draw on various established accounting and carbon crediting methods from reforestation projects.

As interest in ocean-based CDR has increased, the definition of blue carbon has expanded to include other emerging BCEs such as macroalgae.[8] Although historically excluded from consideration due to their lack of roots and limited *in situ* carbon burial, macroalgal systems, composed primarily of brown and red algae, have gained attention for their high rates of primary productivity,[9] near global distributions,[10] and substantial role in capturing and exporting organic carbon to long-term sinks.[11,12] This has prompted greater interest in both wild macroalgae and large-scale cultivation as potential pathways for marine CDR.[10,13,14] However, because macroalgae cultivation relies on deliberate human interventions to achieve sequestration, such as harvesting, processing, or sinking biomass, their mitigation potential is often considered part of a broader category of ''blue carbon'' rather than a more narrowly defined ''BCEs''.[11,15–17]

Additional marine, *i.e.*, ''blue'' biomass-based methods have recently emerged, such as microalgae cultivation in photobioreactor systems.[18,19] However, since these are most commonly associated with carbon capture and utilization – for instance, in the production of synthetic fuels – they usually do not appear when searching for CDR in combination with blue carbon. Accordingly, the classification of blue carbon (ecosystems) varies depending on the perspective and discipline from which the topic is approached.

Literature on CDR methods has expanded significantly in recent years, with research on blue carbon, *i.e.* BCEs, growing at a high rate.[20] Despite increasing interest, there is still a need to comprehensively and systematically structure the synthesis on BCEs, integrating the diverse strand of BCE research and place these findings in relation to other CDR methods. This study aims to address this by providing a current state of BCEs as part of climate change mitigation policy and its application as a CDR method. This review is part of a wider CDR literature reviewing effort, aiming to deliver an ecosystem of systematic reviews on CDR, undertaken by scientists from across the globe. The first part covers direct air CO_2 capture and storage,[21] and this part, part two, covers BCEs.

Accordingly, as part of a coordinated effort based on a shared evidence map and coding protocol,[20] this review is designed to enhance comparability with other CDR options. To address these objectives, we conducted a systematic literature review (SLR) guided by harmonized inclusion criteria. Our search identified over 12 000 records which were narrowed down to 2622 relevant studies using machine learning-assisted screening and expert review, including both peer-reviewed and gray literature. Our search results for blue carbon in combination with CDR are dominated by literature on traditional BCEs, mangroves, seagrass meadows, and salt marshes, with an increasing share of macroalgae. To ensure inclusion of the latest scientific developments (*e.g.*, carbon stocks, sequestration rates) and rely on expertise knowledge, we also include recent, relevant studies identified non-systematically, thus ensuring our findings reflect the most current knowledge in the field.

2. Methods

This SLR was conducted in accordance with the PRISMA (Preferred Reporting Items for Systematic Reviews and Meta-Analyses) guidelines. The PRISMA framework provides a structured approach to improve the quality of reporting in systematic reviews and meta-analyses through enhanced transparency and completeness.[22] Our work followed this framework to ensure our methodology is well documented, reproducible, and aligned with best practices.

Compared to previous blue carbon bibliometric analysis studies, our systematic review is based on trained machine-learning software to exclude non-matching data. Our keyword search is based on Boolean operators, as most studies are, but we include CDR specifically. Further, most studies used VOS-viewer (co-occurrence or co-authorship analysis), CiteSpace (keywords and references), and the bibliometrix package in R Studio to evaluate and further process the data. Our selection and evaluation process are based on machine-learning-supported classifiers and a direct, internal option to evaluate these. Additionally, we use the bibliometrix package of R Studio. Below, we define the eligibility criteria, describe the search strategy, and detail the data collection process.

2.1. Eligibility criteria

To be considered eligible for inclusion, records were required to meet the scope of the review and discuss blue carbon (as a CDR technology). Eligibility was assessed in accordance with our codebook (rules for inclusion of records and rules for assigning each record) and the specific thematic categories targeted by this review. Records must include an abstract, have the full text available, and be written in English. Based on the abstracts and keywords each record was assigned to at least one category (*e.g.* carbon sequestration, techno-economic analysis) to make the review process efficient.

The abstract and full text are needed for the classification and data analysis. Although it may be useful to include literature in other languages, we are limited to English-language articles due to the databases available and in use. We do not exclude for type of publication or study type, because this review includes peer-reviewed and gray literature records. The literature search was conducted from April 2024 to November 2025, which leads to our time frame until 2025.

2.2. Data sources and search strategy

Three bibliographic databases were selected to cover existing peer-reviewed and gray literature: Web of Science (WoS), Scopus, and Open Alex. WoS and Scopus are two of the largest bibliographic databases and act as our main sources for peer-reviewed literature. Open Alex, one of the largest open research catalogues, was selected to cover additional gray literature sources.[23]

The search strategy is based on a harmonized search string that includes relevant terminology to identify a broad and significant body of literature. The electronic search was guided by the peer review of electronic search strategies (PRESS) recommendations[24] and was not restricted by time frame or meta-analytic conditions. Boolean operators were used to better capture blue carbon-specific literature. The string was tested and adjusted multiple times to optimize the search precision and reflect the research objectives. A validation with a sample set of expert-identified studies was conducted to test the search string. The translation of the string itself was done manually and automatically (review management tool), following the guiding principles of the respective databases (Table 1).

2.3. Data selection process

Three reviewers carried out the screening process. The reviewers conducted the screening (title, abstract, keywords) independently. To ensure consistency and reduce ambiguity within inclusion criteria, the reviewers performed several calibration rounds using random record samples (n = 50 each round). During these rounds discrepancies and disagreements were resolved and the rules for inclusion (codebook) iteratively refined. The screening process was supported by a machine learning (ML)-based review management tool (apsis.mcc-berlin-net; NACSOS) that was trained using ClimateBERT.

Table 1 Illustration of the database-specific search strings used, and the corresponding number of records retrieved from each database. Each search string was adjusted to match the search criteria of the respective database

Database	Platform	Search string	No. of identified documents (n)
WoS	Online	TS = (((“blue carbon” OR seagrass OR mangrove* OR coastal OR saltmarsh* OR marsh OR kelp OR macroalgae OR seaweed OR “coastal wetlands”) NEAR/3 (carbon OR CO_2) AND (sequest* OR stor* OR remov* OR capture OR utiliz* OR fuel* OR product OR harvest OR “deep ocean” OR “deep sea” OR NDC OR “Paris Agreement” OR trade OR governance OR local OR “open access” OR dump* OR chemical OR energy OR CCS OR “sequestration potential” OR sink)) NOT “Prussian”)	4567
Scopus	Online	TITLE-ABS-KEY((((“blue carbon” OR seagrass OR mangrove* OR coastal OR saltmarsh* OR marsh OR kelp OR macroalgae OR seaweed OR “coastal wetlands”) W/3 (carbon OR CO_2) AND (sequest* OR stor* OR remov* OR capture OR utiliz* OR fuel* OR product OR harvest OR “deep ocean” OR “deep sea” OR NDC OR “Paris Agreement” OR trade OR governance OR local OR “open access” OR dump* OR chemical OR energy OR CCS OR “sequestration potential” OR sink)) AND NOT “Prussian”))	5738
OpenAlex	Online	TS = ((((seagrass OR mangrove OR mangroves OR coastal OR saltmarsh OR saltmarsh OR marsh OR kelp OR macroalgae OR seaweed) NEAR/3 (carbon OR CO_2)) OR ((“coastal wetland” OR “blue carbon”) AND (carbon OR CO_2))) AND ((carbon OR CO_2) NEAR/3 (sequestration OR sequester OR sequestering OR storage OR store OR storing OR remove OR removing OR removal OR removals OR capture OR utilization OR utilize) OR fuel OR fuels OR product OR harvest OR “deep ocean” OR “deep sea” OR NDC OR “Paris Agreement” OR trade OR governance OR local OR “open access” OR dump OR dumps OR chemical OR energy OR CCS OR “carbon capture and storage” OR “sequestration potential” OR sink)) NOT “Prussian”)	3554
Total			13 859

ClimateBERT is a transformer-based pre-trained language model fine-tuned to more accurately capture domain-specific language relevant to the climate change context, including terminology commonly used in scientific abstracts.[20] Automatic classifiers helped to code studies for relevance and to remove records that were not relevant. The subsequent manual screening process took place on the NLP assisted classification, synthesis and online screening (NACSOS) platform.[25] The manual screening process served as a second monitoring step, screening for relevance and sorting into pre-defined categories (see eligibility criteria). After the screening process, the list of all relevant records (studies) was exported and merged with a second set of records identified by leading experts. Lastly, duplicates and missing data were resolved.

Through this process, we identified a total of 13 859 records across all three databases. After performing the first steps of the screening process and removing duplicates, 4377 records remained for the systematic literature review. The last steps excluded 1645 records, because of missing abstracts, non-English language or other quality problems, and 110 records could not be retrieved as full text. This leaves a final dataset of 2622 records (Fig. 1) and several datasets containing topic-specific studies. These datasets formed the basis for the review team to conduct the assessment. In addition, a search engine was developed containing all studies coded as relevant (final dataset). The internal search engine has two built-in functions: (i) it allows for keyword searches based on keyword matching to indicate whether certain keywords occur side by side, and (ii) it allows for custom search strings to identify a specific part of the literature. All studies are entered as plain text, so tables and figures are missing. However, the search engine provides direct links to the studies, to allow further specification of individual records. This allowed for faster and more detailed analyses for internal search purposes.

The final dataset together with the more refined topic-specific datasets were then distributed to the review teams. Due to the extensive body of literature, we could not include all eligible literature in our review. We prioritized literature that presented primary, robust and well-documented data or findings, as determined by the expertise of our experts. Moreover, our focus was on more comprehensive syntheses that also provide broader insights. In particular, since a substantial part of the literature consists of case studies on regional carbon sequestration, we assessed and selectively included those that offered robust and clear outcomes.

3. Results

3.1. Current status of literature

The growing number of bibliometric studies in recent years reflects the development of 'blue carbon' as a research topic (Table 2).[26,27] Over the period from 1980 to 2025, we identified 2622 relevant articles on blue carbon. The growth rate of the literature is generally similar to other CDR options, but with a slightly higher increase (increased share) in recent times (Fig. 2).[20] In particular, the three "traditional" BCEs (mangroves, seagrasses, salt marshes) form the core of research[17,23] but we can also observe an increase in macroalgae literature (Fig. 2). Over the period chosen for this systematic review, most studies concentrate on only one BCE, though recent years show a shift toward multi-ecosystem studies.

3.2. Natural distribution of blue carbon ecosystems

To the best of our knowledge, the most recent and robust estimates suggest about 15 million hectares (ha) for mangrove forests (range: 8–16 million ha),[32–40] about 5 million ha for salt

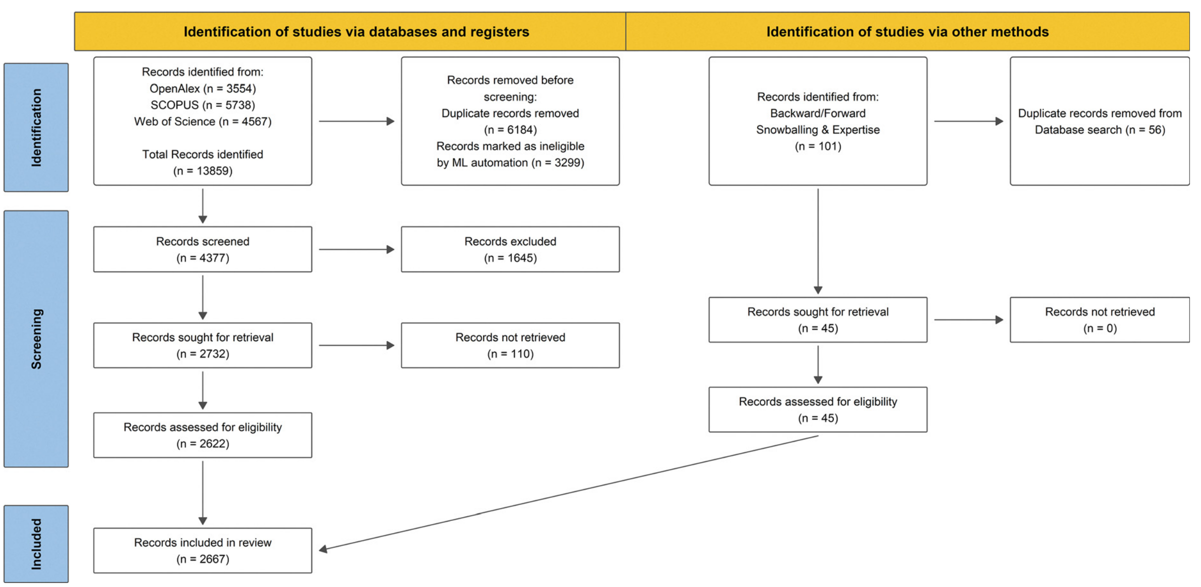

Fig. 1 Overview of the data preparation and cleaning process used to generate the final dataset of records for the SLR. We transparently document and report all records across the specific SLR stages, thereby aligning our process with the PRISMA guidelines. ML = machine learning. Figure follows the PRISMA scheme,[22] rebuild in R.

marshes (3–6 million ha,[41] recently up to 9 ha[37]) and about 21 million ha (14–27 million ha) for seagrass meadows[37,42,43] (with earlier estimates between 30–60 million ha).[38] These estimates, providing a combined estimate of 25–52 million ha, are at the lower end of earlier estimates, ranging up to 185 million ha (with high uncertainties for seagrasses and salt marshes distribution).[44] These uncertainties are reflected in the wide spread of previous estimates, ranging from 16 to 165 million ha for seagrass[35,44–46] and from 2 to 40 million ha for salt marshes.[46–48] Large mangrove forest areas are concentrated in Southeast Asia, which account for approx. 40% of the global total, followed by substantial extents in Western Africa and America.[30,49,50] The highest compiled seagrass areas have been documented in Oceania, Asia, the Caribbean[51] and North America.[42,50] Salt marshes occur predominantly on the North Atlantic region – about 45% of the global distribution – with particularly extensive coverage in the United States.[41]

Compared to the "traditional" BCEs, the extent of macroalgal systems is much larger,[8,11,19] with recent estimates suggesting a total area between 606–722 million ha (including all seaweeds) and with brown algae (including kelp) alone estimated to be between 150 and 250 million ha.[52] However, these estimates are likely inflated due to the lack of high-resolution bathymetry and substrate maps to constrain the models. By contrast, observed (mapped) macroalgal extent is far smaller, primarily capturing surface canopy kelp species like giant kelp (*Macrocystis pyrifera*) that are detectable *via* remote sensing.[53] Subsurface macroalgae, which make up most wild habitats, remain poorly mapped and underrepresented in global maps. Fig. 3 shows the spatial distribution of mangroves (Panel A), salt marshes (Panel B), seagrasses (Panel C), and macroalgal systems (Panel D). The ecosystems show distinct spatial patterns and varying degrees of concentration.

The spatial distribution is usually backed by estimates about the carbon storage in these reservoirs, distinguishing between storage in biomass and storage in soils. Mangroves have the highest carbon stock at approximately 481 tC ha^{-1} (268–722 tC ha^{-1})[56] (Table 3) compared to that of salt marshes of 268 tC ha^{-1} which is though poorly constrained,[57] lacking global biomass estimates, and to that of seagrasses 122 tC ha^{-1} (0–1722 tC ha^{-1}) (Table 3).[58] The calculated global mean stock values of 7.2, 1.3, and 2.5 GtC for mangroves, salt marshes, and seagrasses, respectively, align only to a limited extent with the C stock estimates reported in the literature. For mangrove BCEs, both lower figures—such as in this overview, which provides a derived, robust estimate of 5.8 GtC (Table 3)—and significantly higher figures, such as 11.7 GtC,[59] are discussed. Similarly, higher estimates for the total carbon stock of salt marshes and seagrass meadows BCEs are reported in the literature, 5.6 GtC and 5 GtC, respectively.[60] These discrepancies can be explained, on the one hand, by skewed distributions across the range of estimates, but also—particularly in the case of the higher estimates—by outdated area estimates. More detailed estimates are reported at the country or even regional level. Tracking (blue carbon) changes in stocks across time and space is a precondition to determine their carbon sequestration (Section 3.3) and, in turn, their climate change mitigation contribution (Section 3.6).

3.3. Carbon sequestration and loss rates

3.3.1. General overview of organic carbon stocks and sequestration in blue carbon ecosystems. Coastal vegetated ecosystems sequester carbon into their biomass *via* photosynthesis, a portion of which can be buried in underlying soils where anaerobic conditions slow decomposition rates, leading to the accumulation of soil carbon stocks. BCEs occupy the coastal intertidal to photic subtidal zones and typically occur in depositional environments (except kelps and macroalgae), where some of the most carbon-dense soils accrete over decadal- to millennial timescales. In addition, BCEs often import and store organic matter from adjacent marine and terrestrial habitats, increasing their potential as carbon sinks.[73] Indeed, co-located BCEs often have higher carbon stocks than neighboring isolated systems,[74] though such allochthonous carbon sources are often excluded from carbon accounting in order to minimize the risk of double counting.[75] Conversely, a portion of sequestered carbon can be exported and contribute to carbon sinks in hydrologically connected habitats, including shelf sediments and the deep sea.[76] Further, $CaCO_3$ production and accumulation in underlying soils and how (Ta the relative rates of the organic carbon and the $CaCO_3$ cycles affect chemical changes in the water column, determine the direction

Table 2 Overview of a representative set of bibliometric studies on blue carbon to enable comparison of data, analytical approaches and scope with this review. ML = machine-learning, WoS = web of Science, CKNI = China knowledge network internet, GS = google scholar

Source	Database(s)	Data	Methods/data analysis	Search
28	WoS, CKNI	483 articles, 2001–2021	CiteSpace, textual analysis	Blue carbon or marine carbon sink
27	WoS	1257 articles, 1990–2023	CiteSpace	String incl. blue carbon
23	WoS, Scopus, ProQuest, GS	1179 articles, 2009–2021	ArcGIS, Excel, VOSviewer	Blue carbon, keyword and string
17	WoS	2613 articles, 2003–2021	R bibliometrix, VOSviewer	String incl. blue carbon
29	WoS	1348 articles	BibExcel, VOSviewer	String incl. blue carbon
26	WoS	1729 articles	R bibliometrix, VOSviewer	Coastal wetlands, sequestration and storage
30	WoS	908 articles, 1985–2021	CiteSpace, pathfinder pruning method	Blue carbon
This study	WoS, Scopus, OpenAlex	1812 articles, until 2024	ML-supported, semi-automated analysis	Blue carbon and CDR

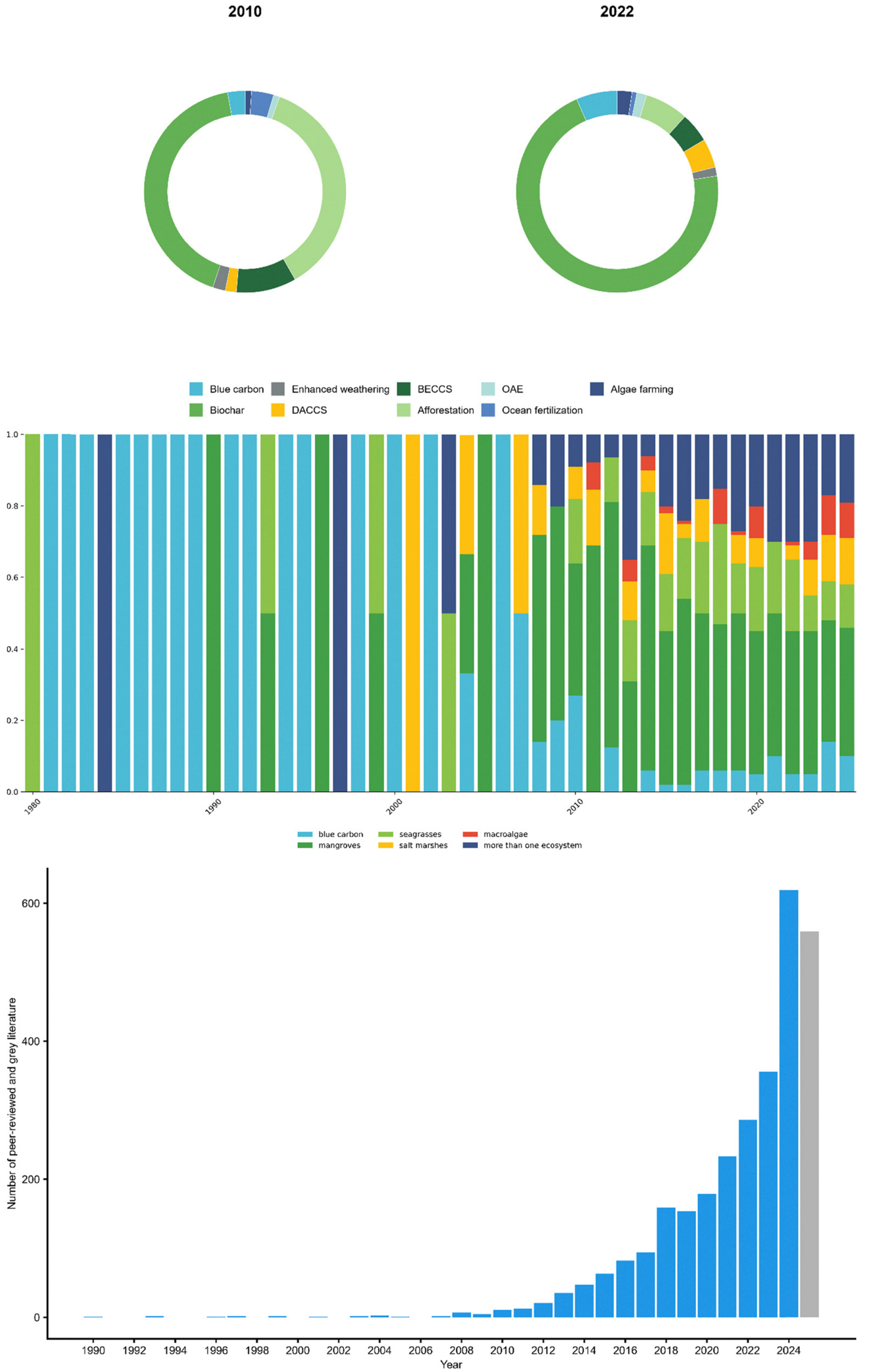

Fig. 2 Development of the share of the literature on BCE within the CDR literature based on two different time points, 2010 and 2022 (Panel A), shares of blue carbon ecosystems in the literature by year (Panel B) and relevant BCE literature from our dataset over time (Panel C). Data collection took place in late 2025. Comparative data for Panel A was taken from the State of CDR Data Portal.[31] For Panel B: If it is not directly clear from the abstract which BCEs are intended or if just the term blue carbon is used, they have been categorized as 'blue carbon' (see Panel B).

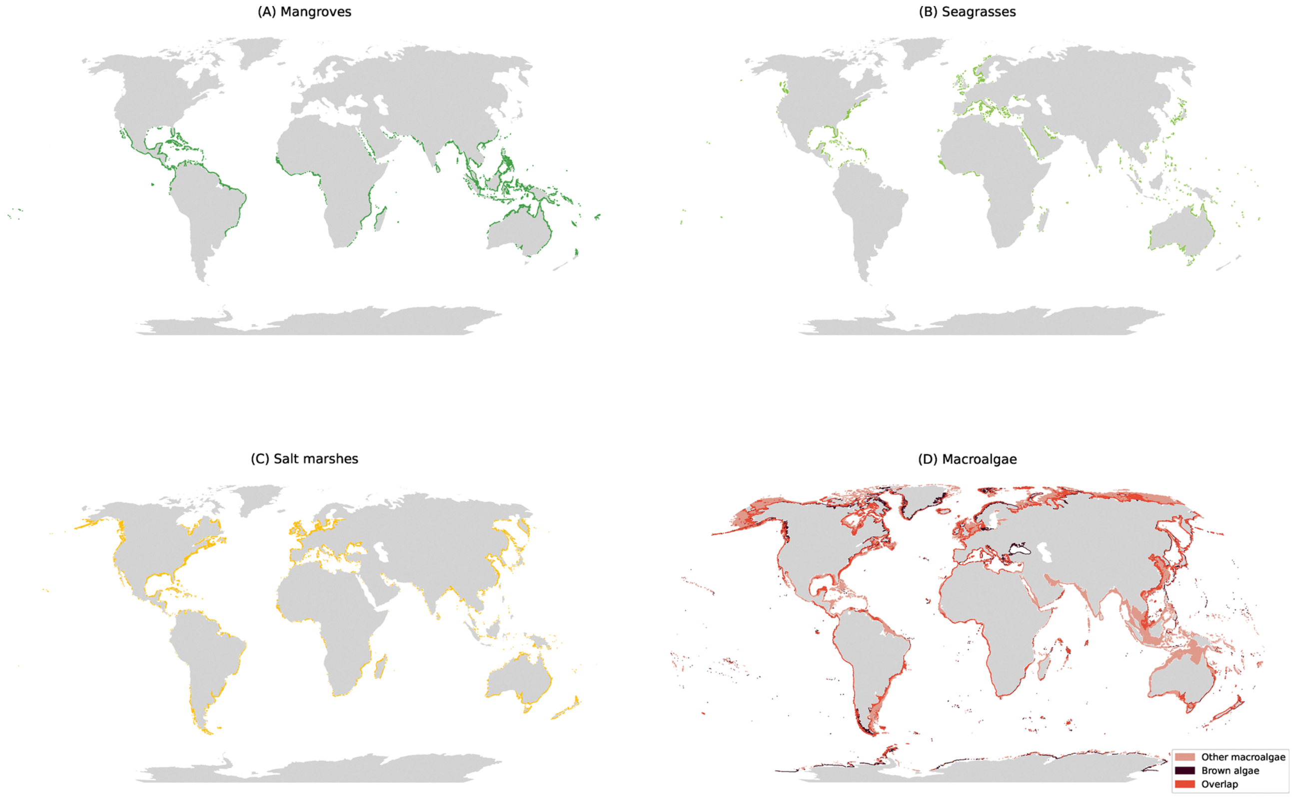

Fig. 3 Global distribution of blue carbon ecosystems. (A) mangroves, (B) seagrass meadows, (C) salt marshes, and (D) macroalgae. Data sources: Mangroves,[33] seagrasses,[54] salt marshes,[41] and macroalgae (other macroalgae and benthic brown algae).[10,55]

of CO_2 exchange across the air-sea boundary and sequestration potential.[73,76,77]

3.3.2. State of knowledge of BC in mangrove ecosystems. The scientific community has a clear understanding of the mangrove carbon cycle and has produced highly constrained estimates for major pools and fluxes.[56] Global carbon stock estimates vary depending on methodology and areal extent, with most recent studies suggesting mangroves store 1.21 GtC in their biomass,[62] and 2.26 GtC[78] to 6.4 GtC[79] in their soils (compare these estimates to estimates mentioned above). Carbon densities in biomass and soil pools vary spatially, driven by or correlated with factors such as geomorphic setting, suspended sediment concentration, and some climatic variables. Key fluxes of carbon into the mangrove system include above-ground biomass production, litterfall production, root production, and soil organic carbon accumulation. Above-ground biomass production is on average 8.3 t ha^{-1} $year^{-1}$, with substantial regional variation,[80] and leaf litter production is expected to add a further 9.7 t ha^{-1} $year^{-1}$.[56] Below the ground, root production adds on average 7.7 t ha^{-1} $year^{-1}$.[81] Soil organic carbon accumulation driven by both autochthonous and allochthonous sources can add 2.3 tC ha^{-1} $year^{-1}$,[82] with relative-sea level rise and availability of accommodation space being important drivers of this process.[83] However, these carbon inputs must be balanced against vertical CO_2, CH_4 and N_2O emissions from stems and soils, driven by processes such as respiration. Emissions in some mangrove systems can exceed the amount of carbon sequestered, meaning that they may be net sources of carbon and contributors to the atmospheric carbon balance.[84]

3.3.3. State of knowledge of BC in seagrass ecosystems. The range in the current best estimate of global seagrass extent (14 to 27 million ha, see above)[42] highlights the uncertainties in quantifying this BCE. Difficulties in mapping seagrass using remote sensing arise where waters are deeper, less clear, and where seagrasses are sparse and co-occur with spectrally similar benthic features. However, novel mapping approaches still uncover large previously unmapped seagrass meadows, such as the 6.6 million ha of seagrass mapped by instrument-equipped sharks in the Bahamas.[85] Similarly, early estimates of global seagrass carbon stocks (140 tC ha^{-1} in the top 1 m of soil[86]) proved to be an overestimate, and a recent synthesis including data from a more diverse set of seagrass habitats put the global median at 78 tC ha^{-1} (at 1 m depth).[58] Carbon stocks in seagrass biomass are much lower than those in underlying soils, with a global mean of 1 tC ha^{-1}, where about twice as much carbon is stored in belowground roots and rhizomes compared to above-ground leaf material.[87] Considerable variability in seagrass carbon stocks was found among seagrass functional groups and geomorphic settings, with larger soil C stocks associated with large, persistent seagrass species and in sheltered coastal environments that potentially receive terrestrial C fluxes, such as lagoons and small deltas.[58,73]

Table 3 Estimated carbon stocks and sequestration rates by ecosystem and carbon pool. Per-area carbon stock and sequestration rate values are given in tons of carbon per hectare and tons of carbon per hectare per year, respectively. Global carbon stock and sequestration rate values are given in megatons of carbon and megatons of carbon per year, respectively

	Per-area								Global			
	ABG [tC ha^{-1}]		BGB [tC ha^{-1}]		SOC[a] [tC ha^{-1}]		C Seq. rate [tC ha^{-1} year^{-1}]		Total C Stock [MtC]		Total C Seq. rate [MtC year^{-1}]	
Ecosystem type	Mean	Min–Max	Mean	Min–Max	Mean	Min–Max	Mean	Min–Max	Mean	Min–Max	Mean	Mi–Max
Mangroves	115[56]	33–261	16.1[61]	2.9–41.1	350[62,63]	232–470	1.68[64]	1.2–1.9	5810[62,63]	—	13.7[64]	10.0–32.0
Salt marshes	ND	—	ND	—	185.3[57]	—	2.42[64]	2.2–2.7	1440[57]	870–1620	10.1[64]	9.0–14.7
Seagrasses	0.3[65]	0.0–5.0	0.8[65]	0.0–18.8	120.8[58]	0.2–1698.0	1.34[66]	0.01–51.5[66]	5234[58]	6–101 904	17.0[66]	14.7–24.9[66]
Kelp												
Cultivated	ND	—	ND	—	ND	—	ND	—	1.5[67]	—	0.3[67]	0.0–3.8
Wild	2.88[8,68–70]	0.8–4.3	ND	—	ND	—	0.9[69,70]	0.4–1.5	ND	—	56[55]	10–170
Macroalgae												
Cultivated	ND	—	ND	—	ND	—	3.6[71]	3.2–4.0	ND	—	ND	—
Wild	ND	—	ND	—	ND	—	0.9[72]	—	ND	—	173[11]	61–268

[a] Carbon stocks represent the total soil organic carbon and are given for the top 1 m of sediments for mangroves, marshes, and seagrasses. AGB = above-ground biomass, BGB = below-ground biomass, SOC = soil organic carbon (top meter).

The perhaps least well-constrained components of seagrass blue carbon are net primary production rates (0.00–51.5 tC ha^{-1} year^{-1}, mean of 6.7 tC ha^{-1} year^{-1}) and carbon burial rates (0.24–0.83 tC m^{-2} year^{-1}),[43,65,88] both of which show large natural variability, reflecting differences in environmental setting and seagrass traits. The carbon burial estimates correspond to 15–20% of net community production and represent the transfer of short-term (weeks to months) carbon storage in biomass to long-term (years to centuries) storage in sediments.[43] Uncertainty in burial rate estimates arises because not all seagrass meadows are located in depositional environments, with episodes of accretion and erosion potentially occurring at the same sites at different times. Available methods for measuring burial rates in seagrasses differ in the time horizon they capture (0.1–10 years for surface elevation tables; 10–150 years for Pb-210 dating; up to 1000 years for carbon dating), where burial rate measurements over short periods are biased toward accretion, while measurements over longer periods are generally lower ('Sadler-effect'),[89] introducing further uncertainty into seagrass carbon burial rate estimates. Not included in these stock and rate estimates are GHG fluxes other than CO_2, although both CH_4 and N_2O fluxes are likely small in seagrasses compared to other BCEs.[77]

3.3.4. State of knowledge of BC in salt marsh ecosystems. Studies prior to the 1980s have noted the high C production of tidal salt marshes. Similar to mangroves, the largest C stocks within this ecosystem type are in soil, yet above-ground biomass can be less significant owing to the annual turnover of leaf tissue of dominant plant species. However, tidal marshes with perennial or suffrutescent and woody biomass can have larger above-ground biomass stocks that appreciably increase total ecosystem C stock. Recent global estimates reported for tidal marsh soil C stocks are 1.44 GtC in the top 1 m of soil, with 83.1 and 185.3 tC ha^{-1} reported for 0–30 and 30–100 cm depth, respectively.[57] Variation among nations was pronounced, while among biogeographic regions, SOC per unit area was within the same range, with the exception of the Arctic. The necessary omission of tidal marsh areas of the Arctic and Tropics due to high prediction errors suggests that the global value is underestimated. Maxwell *et al.* (2024) reported higher variable importance of drivers of environmental variation in tidal marsh SOC stocks as soil depth, elevation, temperature, total suspended solids, and plant greenness (NDVI).[57] As compared with mangroves, key fluxes of C in tidal marshes are derived from belowground production and soil organic carbon accumulation, with above-ground production and litterfall contributing negligible fluxes of C, depending on plant species type. As with mangrove ecosystems, relative-sea level rise and availability of accommodation space (both vertical and lateral) are important drivers of carbon accumulation.[83,90] Less well understood are the roles of periphyton and submerged aquatic vegetation in tidal marsh C cycling, except for in specific study areas. Loss rates of POC and DOC from both restored and degraded tidal wetlands are also areas in need of further research to better constrain tidal salt marsh net ecosystem C balance. Freshwater inflows, sea-level rise and saltwater intrusion, and tidal marsh hydrologic management and restoration can have large influences on belowground C, gaseous (CO_2, CH_4, and N_2O), and particulate fluxes from soils and plant tissues to the atmosphere, soil porewater, and water column, influencing their net ecosystem C balance.

3.3.5. State of knowledge of BC in macroalgal systems. A growing body of evidence suggests that amount of carbon sequestered by macroalgal systems, particularly kelps and other large brown algae, is globally significant[11,12] and points to the potential for large-scale cultivation to further enhance carbon removal.[13,14,52] However, macroalgal systems differ fundamentally from other blue carbon ecosystems in how carbon is sequestered and stored, creating challenges for measuring, monitoring, and incorporating them into current blue carbon and CDR accounting frameworks. Unlike mangroves, seagrasses, and salt marshes, macroalgae store relatively little carbon locally and instead largely contribute to sequestration processes *ex situ* through the export of organic carbon to depositional environments where long-term sequestration and storage can occur.[8,91] It is this spatial decoupling between carbon fixation, transport, burial, and retention processes that underpins the persistent data gaps and uncertainties surrounding the climate mitigation potential of macroalgal systems.[55,92] While there has been significant progress in estimating macroalgal carbon sequestration across different spatial scales,[55,67] these estimates remain highly sensitive to the large error bars in estimated extents, productivity, and export rates, and the assumptions made about downstream carbon fates and long-term retention rates.

3.3.6. Wild macroalgae. In contrast to traditional BCEs, carbon sequestration by wild macroalgae depends on series of biological, oceanographic and geochemical processes affecting rates of detachment, transport, deposition, burial and retention in shelf or deep-sea environments.[92] The efficiency of these processes is strongly influenced by species traits, hydrodynamic conditions, coastal geomorphology, as well as connectivity between coastal areas and depositional environments.[9,55,92] Uncertainty increases along this sequence with early stages of the carbon cycle (production and export) generally being better constrained than downstream processes. As a result, the carbon sequestration potential of wild macroalgae is expected to vary substantially across space and among regions.

Carbon storage in macroalgal biomass (including kelp) is the most resolved, although it is considered temporary storage because of the short life span of most species, ranging from 0.8 to 4.3 tC ha^{-1} (mean = 2.9, SD = 1.49; Table 3). An estimated 45–61% of carbon fixed by macroalgae enters marine food webs as particulate organic carbon (POC) or cycles through microbial communities as dissolved organic carbon (DOC). However, only portion of this fixed carbon is expected to contribute to long term carbon sequestration through three main pathways: (i) the release and creation of refractory DOC, (ii) the burial of POC in coastal sediments on the continental shelf – including those of other blue carbon habitats; and (iii) the export of POC off the continental shelf to deep ocean, where long residence times allow carbon to remain sequestered even if remineralised.[67]

First-order global estimates suggest that wild macroalgae could sequester between 61 and 268 MtC $year^{-1}$, more than all other blue carbon habitats combined.[11] From these, approximately 117 MtC $year^{-1}$ is estimated to enter DOC pools below the mixed layer, 14 MtC $year^{-1}$ is buried in coastal sediments, and 35 MtC $year^{-1}$ reaches the deep ocean.[11] More recent studies suggest that deep-ocean export may exceed 56 MtC $year^{-1}$ (range 10–170 MtC $year^{-1}$),[55] with country-level contributions ranging from 0.01 to 3.8 MtC $year^{-1}$ (Table 3). However, much of the existing literature emphasizes the supply side of sequestration, namely measuring net primary productivity and export, with relatively little data available on retention and accumulation rates in marine sediments (on the shelf or at depth). Recent synthesis of available data suggests sedimentary accumulation rates range between 19.9 and 91 gC m^{-2} $year^{-1}$ (Table 3), although these estimates are among the least constrained components of the macroalgal carbon budget.

3.3.6.1. Cultivated macroalgae. Cultivated macroalgae represents a managed extension of these same carbon sequestration pathways. Although, seaweed farming currently occupies a much smaller footprint than wild macroalgae – an estimated 2000 km^2 globally[52] – it holds significant potential for expansion both in coastal and offshore environments.[13,91] Each year, approximately 1.5 MtC is fixed and assimilated into harvested macroalgal biomass.[52,92] Some of this carbon is lost prior to harvest and may settle into underlying sediments, where, under favorable conditions, it can accumulate at rates between 25 to 71 gC m^{-2} $year^{-1}$ and be stored for longer timescales.[92–94] However, the magnitude of realized sequestration is expected to vary widely with the species composition, cultivation technologies, harvest regimes, and local environmental conditions that influence rates of biomass production, loss, transport, and retention.[14]

As with other biomass-based systems, however, net climate mitigation from cultivation depends on sustained regrowth following harvest and on the ultimate fate of harvested or lost biomass. Key sources of uncertainty include post-harvest losses, the balance between long-term storage and rapid remineralization, and life-cycle emissions associated with cultivation, processing, and transport.[14] While first-order global estimates suggest that current seaweed farms contribute between 0.01 and 3.8 MtC $year^{-1}$ of carbon sequestration, their net carbon removal benefit may be more limited when emissions from energy usage and material inputs are considered.[92]

Table 3 summarizes the results regarding carbon stocks and sequestration rates, both per area and globally. It should be noted that this selection does not reflect the full range of estimates found in the reviewed literature but rather a selection of plausible estimates. Note, the estimates in Table 3 are not necessarily consistent when extrapolated from per-unit-area to global values. Based on Table 3, the total natural sequestration accumulates therefore to 270 MtC $year^{-1}$, ranging between 106.1 and 515.9 MtC $year^{-1}$.

3.3.6.2. Carbon loss rates. Blue carbon management approaches aim not only to achieve atmospheric carbon removal in the narrower sense, but the literature also emphasizes that destroying ecosystems would lead to foregone carbon sequestration and increased emissions. BCEs face substantial threats from human activities and external pressures. These threats can lead to the erosion of sediment carbon stocks,[95–97] which store the majority of carbon in these ecosystems,[98] as well as declining productivity and flux for seaweeds/macroalgae. BCE loss rates have historically been estimated to be between 0.4 and 8.0% per year, with mangroves estimated to have higher rates than seagrasses and salt marshes.[47,99,100] The accuracy of these estimates remains constrained by inconsistencies in ecosystem mapping and the lack of consistent and systematic remote sensing methods. Loss rates also have a time element. Although ecosystem loss rates may no longer be as high as in the 1970s and 1980s, BCEs still continue to be lost within that range.[101,102] More recently, the most robust global remote sensing analyses show annual loss rates in the 21st Century of 0.13–0.62% for mangroves, 1–2% for seagrasses, and 0.28% for salt marshes (Table 4).[51,97,103,104] For mangroves, these estimates are lower than earlier assessments,[39,105] which reported annual losses up 1.9%[106] or even 3.0%.[47] A similar pattern is observed for seagrasses and salt marshes: seagrass estimates vary substantially across study sites, with reported high rates between 1990 and 2006,[107,108] and general historical loss rates around 0.7% per year before 1940 and as high as 7.0% annually towards the end of the 20th century,[47,109] although recent global mapping of shallow-water seagrasses suggest more moderate loss rates of 1.1% from 2019–2024.[51] For salt marshes loss rates in the range of 1–2% annually were reported.[47,102] Salt marsh loss rates were driven by high losses in Russia and the USA between 2015 and 2019.[109] More recent analyses suggest substantially reduced loss rates at around 0.28% for the period of 1996 to 2016.[109] Since the early 21st century, data shows reduced mangrove loss rates[110] and higher recovery rates,[111] especially shown in Mexico,[112] a stabilization of lost seagrass ecosystems, especially in Europe[113–115] and a declining trend in loss rates for salt marshes.[109] Macroalgae loss rates are estimated (limited data) to be around 1.8%[72] (Table 4) with mixed trends[116] and results indicating that 38% of global kelp forests have been declining over the past decades.[116]

The loss rates of blue carbon ecosystems arise primarily due to a combination of non-climatic and climatic drivers. Among the non-climatic factors, erosion, land-use and land-cover changes,[79,88,114,121] such as forestry activities, agriculture, aquaculture (*e.g.*, fish farming and shrimp pond production),[122,123] dredging,[109,122] water quality deterioration, and shoreline modification play significant roles. Increasingly these local pressures are also interacting with climate drivers.

Table 4 Global estimates of blue carbon ecosystem annual loss rates by ecosystem. Annual loss rates are given as percentages with reference to measured time period

Ecosystem	Annual loss rate [%]	Time period	Ref.
Mangroves	0.62	1985–2020	117
	0.14	1996–2020	32
	0.16	2000–2012	118
	0.13	2000–2016	119
	0.17	2000–2020	34
Seagrasses	<2.00	1880–2016	120
	1.10	2019–2024	51
Salt marshes	0.28	2000–2019	109
Macroalgae	1.80	1975–2014	72 and 116

In addition to human pressures and modifications, climatic change drivers, including sea-level rise, extreme weather events,[123,124] gradual warming, and episodic marine heatwaves,[109,124,125] are increasingly altering the distribution, stability, and persistence of BCEs[126] and their carbon stores.[127,128] Mangroves and salt marshes are particularly vulnerable to sea-level rise, storm-driven erosion, and coastal squeeze processes that constrain the potential for landward migration under climate-driven redistribution.[129,130] Whereas seagrasses and kelp forests are highly sensitive to ocean warming, marine heatwaves, and shifting biological processes (*e.g.*, disease and herbivory processes) that can drive widespread redistribution, mortality and regime shifts.[131,132] In addition, ocean warming, episode heatwaves, and shifting disturbance regimes are likely to undermine the permanence of blue carbon stores by accelerating carbon remineralization rates and reducing long-term burial efficiency.[133,134] Explicitly accounting for climate-driven redistribution and disturbance dynamics in future assessments will be critical for constraining uncertainty around the long-term climate mitigation and carbon removal potential of these ecosystems.

3.3.6.3. Mitigation and carbon removal potential. Information about current and historical loss rates provides the basis for estimates for carbon sequestration *via* restoration. For example, Macreadie *et al.* (2021) report potential habitat areas for restoration in the magnitude of 9–13 million ha, 8.3–25 million ha, and 0.2–3.2 million ha for mangroves, seagrasses, and salt marshes, respectively.[44] Using such area estimates, the literature derives estimates for the mitigation potentials. Table 5 provides an overview of estimates in the literature, mainly obtained from review studies. It should be noted though that these estimates in the literature are not necessarily consistent with the estimates on carbon stocks and sequestration rates (Table 3) since the studies have different assumptions regarding what they consider as net sequestration rates, in how far they account for changes in carbon fluxes outside the enhancement area (in case of restoration projects), and whether the removal potential is net of other non-CO_2-emissions and if yes, how, under which global warming metric these emissions have been netted against the carbon removal.

The mitigation potential from reducing the CO_2 emissions from mangroves, seagrasses, salt marshes, and macroalgal systems degradation ranges between 35.49 to 218.34 MtC year^{-1} by 2030 and between 60.05 to 95.53 MtC year^{-1} by 2050. The low upper rate in 2050 is likely to result from the assumption in the studies that a large proportion of habitats will already have been lost by then.

Restoring the traditional BCE habitats, *i.e.* mangroves, seagrass meadows and salt marshes, it projects to provide a

Table 5 Mitigation potential estimates [MtC year^{-1}]. Conservation (or protection) means legal protection of BCEs and reduced degradation rates, restoration means returning to levels of preexisting conditions[19,72]

	2030	2050	Ref.
Conservation			
Mangroves	5.46–10.92	13.65–35.48	19 and 135
Seagrasses	8.19–177.40	35.48–43.67	19, 58 and 135
Salt marshes	10.92–19.10	10.92–16.38	19 and 135
Macroalgae[a]	10.92		135 and 136
Total	35.49–218.34	60.05–95.53	
Restoration			
Mangroves	13.65–21.83	2.73–163.76	19 and 135
Seagrasses	2.73–5.46	8.19–57.31	19 and 135
Salt marshes	1.09–2.73	2.73–10.92	19 and 135
Total	17.47–30.02	13.65–231.99	
Increased macroalgae production *via* aquaculture[b]	2.73–5.46	7.37–215.61	19, 135, 137 and 138

[a] Kelp forest conservation (high uncertainty). [b] For macroalgae, different storage pathways exist, and significantly more storage is possible in combination with BECCS.

mitigation potential between 17.47 and 30.02 by 2030 and between 13.65 and 231.99 MtC year^{-1} by 2050. The upper limit increases to 608.83 MtC year^{-1} if macroalgae-based estimates are included (Table 5). Hence, the combined mitigation and removal potential in the year 2050 ranges between 73.70 and 327.52 MtC year^{-1} for the traditional BCEs (including also kelp protection) and 81.07 and 543.13 MtC year^{-1} including also macroalgae-based methods.

As mentioned above these ranges in Table 5 should be interpreted with caution because the studies have different assumptions about how the measures are implemented. The lower end of the estimates is often based on bottom-up estimates that consider implementation limitations, while the upper end is often based on Earth system model estimates. Accordingly, in the table, we have given preference to estimates from review articles and supplemented them with select individual estimates. Nevertheless, it should also be noted that some individual studies find significantly higher estimates. This is particularly true for macroalgae-based studies. In these studies, the CDR potential is considerably higher depending on how the biomass is used. For instance, Wu *et al.* (2023) combine macroalgae cultivation with sinking the biomass into the deep ocean, resulting in an annual potential of approximately 3400 MtC.[139] If combined with upwelling of nutrient-rich water, the annual potential increases to 5600 MtC (see also next section on combination with other CDR options).

While the global CDR potential is relevant for global assessments and integrated assessments, national climate policies require information on the local potential. For example, for the USA, the National Academies of Sciences, Engineering, and Medicine (NASEM) (2019) estimate that restoration efforts in national coastal wetlands could lead to an additional cumulative carbon removal of 1500 MtC by 2100.[140]

Blue carbon sequestration through macroalgae cultivation and the use of marine biomass could offer a significantly higher potential, with feasible annual carbon removal estimated between 800 and 1100 MtC year^{-1}.[140] Again, such estimates should not be confused with the realistic CDR potential, as these must take into account the costs at which the measure can be implemented and the regional variation in demand for CDR.

3.4. Side effects and interactions with other CDR options

3.4.1. Positive and negative side effects. The most common co-benefit of BCE restoration analyzed and identified across studies is the enhancement of marine biodiversity.[141,142] BCEs serve as vital habitats for fish, birds, and invertebrates,[143–147] provide coastal protection,[148] and act as natural barriers against erosion, storms, and flooding, while stabilizing sediments.[149–152] They also offer important climate adaptation benefits, such as protecting coastal infrastructure from the impacts of rising sea levels (Fig. 4).[153] For instance, coastal wetlands alone are estimated to provide a median annual global value of 447 billion USD in storm protection services, saving thousands of lives each year.[154]

BCEs, particularly mangroves, seagrasses, salt marshes and macroalgae (including kelp) ecosystems, serve as vital nursery habitats for many commercially important species. These habitats provide shelter and abundant food resources, which are crucial for the early life stages of fish, leading to increased survival rates and larger fish populations.[155–159] For example, Heimhuber *et al.* (2023) estimate that restoration of salt marshes and mangroves in the Lake Wooloweyah segment of the Clarence River estuary in eastern Australia can result in an additional 230 tons of eastern school prawn harvested annually, valued at approximately AUD 3.1 million, representing a 50% increase over current harvest levels.[158] Vondolia *et al.* (2020) estimate that improved management of kelp forests along the Norwegian coast has increased cod catches by 8%.[160] A global meta-analysis confirmed a strong positive correlation between mangrove area and local fishery catches, with an overall effect size of r = 0.72 (95% CI: 0.61–0.81), highlighting the importance of mangroves in supporting fishery resources.[161] Furthermore, these habitats attract coastal tourism and offer provisional services including recreational, educational, and medicinal uses alongside cultural

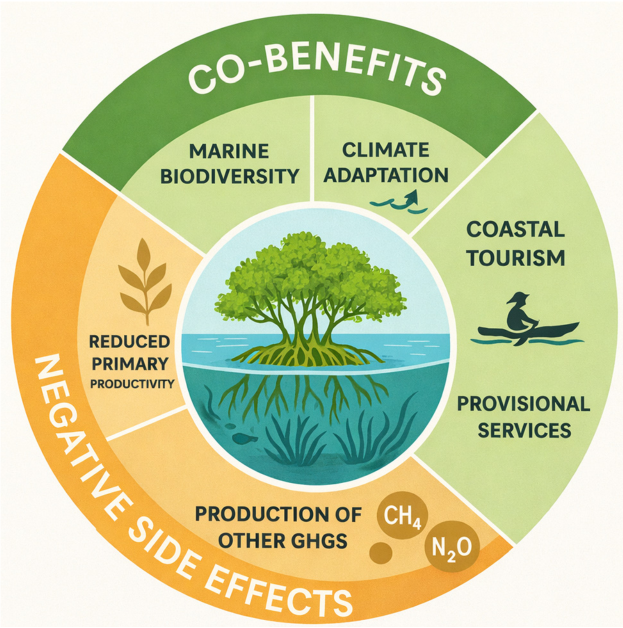

Fig. 4 Visualization of the positive co-benefits and negative side effects for blue carbon ecosystem restoration and protection. The co-benefits outweigh the side effects.

values.[162–164] Restoration projects also generate employment opportunities, contributing to equitable economic growth, especially in developing regions.[153] Lopez-Rivas and Camilo Cardenas (2024) analyze 67 studies and estimate that the economic value of provisioning services for marine and coastal ecosystems (among them BCEs) is between USD 99 and 1535 per hectare annually, cultural services are between USD 45 and 2170 per hectare annually, and recreation and tourism services range from USD 185 to 895 per individual per year. The data on seagrass for provisioning services show that it appears to be at the upper limit of the range, whereas mangroves are in the middle. However, for cultural services, the values for mangroves are above average.[165]

However, negative side effects can arise from the production of other GHGs, like methane (CH_4) and nitrous oxide (N_2O). These gases are produced and released during the decomposition of organic matter, which partly offsets the climate benefits of carbon sequestration, as they have significantly higher warming potentials than CO_2.[166,167] BCE restoration should closely monitor these GHGs as the release is highly site-specific and different estimates prevail.[166–168] A study by Roth *et al.* (2023) reveals that methane emissions from macroalgae, mixed vegetation, and surrounding sediments can offset up to 35% of the CO_2 these habitats absorb annually.[169] Further negative ecological side effects can arise, especially if restoration sites are not carefully selected. Restoration or large-scale expansion can lead to nutrient competition,[170] reducing the productivity of other primary producers like phytoplankton by altering marine food webs.[139] According to model simulations, the expansion of open-ocean macroalgae farms and the subsequent sinking of biomass can create new Oxygen Minimum zones on the seafloor by oxygen consumption from remineralization of sunken biomass.[139]

3.4.2. Interactions with other CDR and geoengineering methods. Few studies have examined how BCEs interact with other marine carbon dioxide removal (mCDR) strategies such as ocean alkalinity enhancement (OAE), ocean fertilization (OF), or artificial upwelling (AU).

OAE is a promising mCDR method that captures atmospheric CO_2 through an increase in seawater alkalinity. Through electrochemical reactions, the addition of alkalinizing solutions (*e.g.* NaOH, $Mg(OH)_2$), or enhanced mineral weathering (EMW) *via* the *in situ* dissolution of ground carbonate and/or silicate alkaline minerals (*e.g.* limestone, basalt, steel slag, olivine), CO_2 in seawater is stored in as bicarbonate and carbonate ions resulting in durable carbon storage on the order of 10 000 to 100 000 years.[137,171–174]

Geochemical conditions within blue carbon sediments may further promote mineral dissolution and influence the outcomes of EMW. In BCEs, photosynthesis drives oxygen into the sediment, where biological activity (redox reactions, respiration) and chemical conditions (high CO_2, low pH) could enhance mineral dissolution in vegetated sediments.[171] Estimates suggest the mangrove and seagrass restoration potential can lead to additional permanent carbon removal by increasing ocean alkalinity through increased rates of anaerobic respiration and increased dissolution of calcium carbonate in sediments. Combined, the restoration-induced OAE yields removal of $\sim$9–221 $MtCO_2$ $year^{-1}$.[142]

The mineral olivine has received significant attention for use in EMW due to its global abundance and fast dissolution rate.[175] A lower pH increases free protons (H^+) in solution, enhancing olivine dissolution with the exchange of H^+ with ions of Iron (Fe^{2+}), Magnesium (Mg^{2+}), Nickel (Ni^{2+}), and Chromium (Cr^{3+}).[171,176] The release of these cations not only influences seawater chemistry, but may have broader ecological impacts within BCEs by providing essential nutrients and introducing heavy metals.

Research is in the preliminary stages to determine the impact of potentially toxic metals (*e.g.* Ni^{2+} and Cr^{3+}) that are released during olivine dissolution.[177,178] Pairing olivine dissolution with BCE management may help mitigate these risks, as plants could uptake and sequester harmful byproducts such as Ni^{2+}, thereby improving the safety of this approach.[179]

Continued research is needed to determine whether dissolved nutrients such as Fe^{2+} and Mg^{2+} can enhance plant biomass and productivity. For example, iron is essential for the growth of marine phytoplankton and other primary producers, yet its availability is often limited in environments dominated by calcium carbonate sediments.[176,180] The release of Fe^{2+} during olivine dissolution may stimulate productivity in iron-limited systems, leading to increased biomass, greater photosynthetic CO_2 uptake, and enhanced carbon sequestration.[176]

In principle, a further possible interaction would be to combine macroalgal blue carbon with artificial upwelling, a technique that pumps nutrient-rich deep ocean water to the surface—which could stimulate macroalgae growth, particularly in offshore, nutrient-poor (oligotrophic) regions. Using current technology, the global CDR potential from macroalgae growth driven by AU is estimated to be about 0.1 $GtCO_2$ $year^{-1}$.[181] A modelling study of AU in selected Chinese kelp farms showed increased kelp growth and higher plant density[182] by increasing nitrate and phosphate concentrations. A study by Wu *et al.* (2023) suggests that macroalgae open-ocean mariculture and sinking (MOS) has significant CDR potential (see previous section).[139] However, this technique causes significant ecological side effects, such as local and remote nutrient depletion in surface waters, reduced phytoplankton productivity *via* competition for nutrients and light (shading of deeper waters), and changes in oxygen minimum zones. To date, interactions between OF and coastal or pelagic BCEs have not been explicitly studied.

While most CDR strategies focus primarily on CO_2 sequestration, the restoration and management of BCEs provide a range of additional benefits beyond carbon storage. Although their additional CO_2 removal potential may be lower than large-scale marine CDR approaches like OAE, BCEs are generally perceived more favourably by the public as a ''natural'' or ''nature-based'' solutions (Table 6).

While this literature almost exclusively focuses on interaction with other CDR methods, mitigating and adaption to climate change might require including also symptomatic approaches by reflecting incoming sunlight. As first question arises therefore in how far different BCEs and their restoration affect the (ocean) albedo. Bach *et al.* (2021) find that macroalgae cultivation reduces the ocean albedo and, in their scenario, (extra floating sargassum in Great Atlantic Sargassum Belt) even exceeds the reduction in radiative forcing from carbon sequestration.[186] However, the study was contested with respect to the site selection and used data,[187,188] showing that the results on carbon sequestration are insufficient supported, implying also that the insights from Bach *et al.* (2021) on ocean albedo change can be considered to be inconclusive. In addition to changes in the albedo in seawater, changes in BCE coverage and in particular macroalgae cultivation could increase dimethyl-sulphide (DMS) emissions, promoting marine cloud formation and associated cooling effects.[189] However, our systematic search did not identify any articles that quantify this effect for BCE conservation and restoration or macroalgae cultivation. A second question addresses how SRM intervention impact the biological carbon sequestration underlying the various BCE methods. While terrestrial ecosystems may benefit from reduced heat stress and increased diffuse light, enhancing productivity and net carbon uptake by up to $\sim$20%,[190–192] reduced precipitation could negatively affect tropical regions.[193] The effects on BCEs are poorly understood and may arise from changes in temperature, circulation, wind stress, and light availability.[190] While modelling indicates that solar shading redistributes phytoplankton productivity without altering total production,[194] circulation changes may permit continued deep-ocean and polar warming, sustaining cryosphere melt and sea level rise at reduced rates.[195] In a more recent review, Roberts *et al.* (2026) provide an overview about earth system model (ESM) and marine ecosystem models assessments on the impact of stratospheric aerosol injection (SAI) and marine cloud brightening (MCB).[196] Applying these

Table 6 Comparison of side effects and co-benefits across CDR methods. Categories were grouped, *e.g.* provisional services include recreational, education, and medicinal uses alongside cultural values or climate adaptation benefits include counteracting SLR effects and buffering of storm surges. Categories in bold indicate positive co-benefits, categories in italics negative side effects. The comparison is based on recent literature[1,183–185]

CDR method	Side effects and co-benefits
AF	**Enhancement of biodiversity**
	Improved soil carbon, nutrient water and cycling impacts
	Fight desertification
	Employment opportunities
	Possible biodiversity losses for high-carbon monocultures
	Less agricultural exports
	Higher food prices
	Albedo change
Biochar	**Increase soil quality and crop yields**
	Reduced methane and nitrous oxide emissions from soils
	Nutrient and water cycling impacts
	Impact on food systems
	Competition for biomass resources
	Down-regulation of plant defense genes
BECCS	**Economic diversification, market opportunities**
	Energy independence, technology development
	GHG emissions substitution
	Impact on food systems, health impacts
	Biodiversity losses, deforestation and forest degradation
	Albedo change
	Social acceptance risk
EW	**Increase soil quality and crop yields**
	Improved plant nutrition
	Counteract ocean acidification
	(human) health impacts
	Ecological impacts (mining and transport)
	Direct and indirect land use change
OAE	**Counteract ocean acidification**
	Ecological impacts (mining and transport)
	MRV, difficulties to verify carbon sequestration
	Partly unknown impacts on marine biology and food web structure
OF	**Potential increase in fish catches**
	Partly unknown impacts on marine biology and food web structure
	Changes to nutrient balance
	MRV difficulties to verify carbon sequestration
DACCS	**Low biodiversity impact**
	Employment and business opportunities
	Subject to a predictable CO_2 price
	Social acceptance risk
	High energy demand (fossil fuel risks)
	High front-up capital costs
	Unknown waste implications
BCE	**Enhancement of marine biodiversity**
	Climate adaptation benefits
	Coastal tourism
	Provisional services, employment opportunities
	Production of other GHGs
	Nutrient competition
	Reducing productivity of other primary producers
	MRV, difficulties to verify carbon sequestration

The abbreviations indicate AF: afforestation, BECCS: bioenergy with carbon capture and storage, EW: enhanced weathering, OAE: ocean alkalinity enhancement, OF: ocean fertilization, and DACCS: direct air carbon capture and storage.

methods at scale potentially reduces net primary production *via* reduced atmosphere and ocean temperature, while simultaneously reducing non-CO_2 emissions from these ecosystems and increasing CO_2 solubility in seawater. In turn the net impact is scenario dependent, *i.e.* in a high emissions scenario, reducing heat stress can result in increased net climate benefit of BCEs and the other way around. If we consider not only traditional BCEs but also coral reefs, there is of course an additional adaptation benefit due to SRM and MCB in particular. While studies demonstrate that coral reefs and the Great Barrier Reef in particular would benefit from reduced heat stress, they also show that ecosystem restoration is complex and MCB interventions should be combined with additional measures like the control of coral predators and introducing thermally tolerant coral species.[197–199]

3.5. Technological readiness and costs

Technological readiness levels (TRLs) assigned to BCE CDR methods give an estimate of the present availability of technological components and the time of development before implementation.[200] However, BCE CDR is not strictly technology-based, and for those deployment steps involving technology, separate components of technologies differ in their development stage.[201] While various studies report about implemented case studies (*i.e.* TRL basically achieved), other studies suggest new approaches for restoring, involving for example autonomous seeding robots, implying that for such deployment strategies TRL is rather low.[202] If scalability is included in these estimations, the margin widens even further. We identified only a small number of studies related to blue carbon explicitly including TRL assessments. Most studies do not mention TRL in their title or abstract. Some publications estimate BCE as a whole CDR technology, while others only assess certain sub-technologies (*e.g.*, kelp reforestation). Compared to other CDR methods, BCE approaches occupy an intermediate (to high) level in both technological readiness (Table 7).[1,200,203–209] Certain methods, such as mangrove restoration and kelp farming, can be seen as highly functional[200] varying by ecosystem.[8,200] Afforestation, along with several direct air capture with carbon storage (DACCS) and bioenergy with carbon capture and storage (BECCS) technologies, generally exhibit higher TRLs. In contrast, methods such as ocean alkalinity enhancement (OAE) and ocean fertilization (OF) remain at earlier stages of development and deployment.

Presenting the costs of various BCE measures as well as various CDR measures in general is challenging and should be done in the context of how this information is used. Initially, the screening process identified a larger number of cost estimate publications (35% of all TEA-related publications). However, the cost components taken into account in individual studies vary considerably, making them difficult to compare, and aggregation results in very wide ranges. Table 8 shows therefore the results of three large multi-CDR assessments and thus results that should be at least consistent within themselves. This means that the studies within the column can be compared row by row, but there are already limitations when comparing across columns.

The information in Table 8 allows the various CDR methods to be roughly compared in terms of their different costs and global cost functions for integrated assessment models to be calibrated. Nevertheless, the possibilities for making statements about the regional cost advantages of different CDR methods are limited at this level of aggregation, as we are not dealing with individual methods here, but rather groups of methods. To be fair, it must be said that studies such as those by Cobo *et al.* (2023)[184] in particular present different technical variants within the method groups. For example, we have summarized nine different BECCS techniques from the study by Cobo *et al.* (2023). However, in these multi-CDR assessments, BCE is still very often presented in a highly aggregated form, with exceptions limited to macroalgae applications. These are at least decoupled in the studies by Mertens *et al.* (2024)[183] and Cobo *et al.* (2023) and presented separately. Still,

Table 7 Overview of recent TRL assessment across CDR methods to enhance comparability. Technological readiness for blue carbon most recently lies within the medium to high range, depending on the ecosystem

CDR method	TRL assessment
AF	8–9
Biochar	6–7
BECCS	3–8 [3–5[b], 6–7[c]]
EW	1–4[a]
OAE	1–2
OF	1–3
DACCS	6–7 [2–7][a]
BCE	5–6 [2–6]

[a] Indicates differences due to technology usage. [b] Power. [c] Industry.[183,204–206,210] The abbreviations indicate AF: afforestation, BECCS: bioenergy with carbon capture and storage, EW: enhanced weathering, OAE: ocean alkalinity enhancement, OF: ocean fertilization, and DACCS: direct air carbon capture and storage.

Table 8 Cost ranges from selected CDR multi-assessment studies for various CDR groups

CDR method	Babiker *et al.* (2022)[1]	Mertens *et al.* (2024)[183]	Cobo *et al.* (2023)[184]
	USD per tCO_2		
AF	0–240	5–50	5–53
SCS	0–100	0–100	0–105
Biochar	10–345	30–120	32–127
BECCS	15–400	100–200	21–1365
EW	24–578	50–200	63–211
OAE	40–260	40–260	82–181
OF	50–500	50–500	24–519
DACCS	84–386	100–600	67–600
BCE	100–10 000	170–220	200
Seaweed cultivation		25–125	
Macroalgae farming and sinking			150–200

The abbreviations indicate AF: afforestation, SCS: soil carbon sequestration, BECCS: bioenergy with carbon capture and storage, EW: enhanced weathering, OAE: ocean alkalinity enhancement, OF: ocean fertilization, and DACCS: direct air carbon capture and storage.

this aggregates cost estimates for measures that vary too greatly, such as the protection of mangrove forests and the restoration of seagrass beds.

An exception is the study of Claes *et al.* (2022) which is restricted to BCEs but presents different point estimates for restoration of mangroves, salt marshes, and seagrass meadows at USD 15, 200, and 300 per tCO_2, respectively.[135] According to Claes *et al.* (2022), the restoration of macroalgae (including kelp) would cost USD 6000 per tCO_2, however, if combined with farming (seaweed), carbon removal would be feasible at USD 280 per tCO_2. Claes *et al.* (2022) also provide cost estimates for BCE protection, *i.e.* reducing emissions, which for all BCEs are below USD 10 per tCO_2 and a median estimate of USD 6 per tCO_2.[135] Yet, other studies suggest that not all costs of the restoration activities are properly accounted for. While the presented studies usually cover the cost of seeding, planting, and farming, Bayraktarov *et al.* (2016) report that the real costs of restoration are likely to be two to four times higher when land-based operating, monitoring and capital costs are included, and it remains often unclear which life-cycle emissions and other GHG emissions are included in the calculation (see Section 3.4).[211] Also, when accounting for associated non-CO_2 emissions, in particular CH_4, BCE restoration targeted at carbon sequestration only is estimated to cost between USD 491 per tCO_2 and USD 560 per tCO_2 for coastal wetlands and mangrove restoration, respectively.[212] Furthermore, it remains unclear whether the point estimates presented in Claes *et al.* (2022) apply to a specific region since both, the natural preconditions but also labor cost vary regionally.

However, particularly in more recent publications, there are an increasing number of studies that attempt to remedy these shortcomings and compile this information consistently, at least for a specific BCE group. This applies primarily to mangrove BCEs, of course, as this is the most advanced BCE method from a CDR perspective. Most notably is the recent study of Goto *et al.* (2025),[213] who use data from 249 mangrove restoration projects in 25 countries to estimate the spatial cost of mangrove restoration. They derived a marginal implementation cost curve and found that 75 percent of the total restoration potential of 1.1 million hectares (mha) could be realized at an implementation cost below USD 10 per tCO_2, and 85 percent could be realized at an implementation cost below USD 20 per tCO_2. Note that these shares are slightly smaller if also the cost for acquiring land are included. These low cost, compared in particular to the cost ranges presented in Table 8, result from realizing mangrove restoration in West Africa, East Africa, Madagascar, and Southeast Asia countries. At the same time, the study of Goto *et al.* (2025) also shows that realizing the full CDR potential *via* mangrove restoration requires carbon prices of USD 500 per tCO_2, resulting from certain high cost areas like for example in New Zealand. This shows that although cost ranges are informative, the distribution within this range in the form of a marginal cost curve ideally provides a much more comprehensive assessment of the CDR potential.

For BC methods that involve more processing steps, like macroalgae cultivation and harvesting, the range in cost increases even further. For example, DeAngelo *et al.* (2023) show that macroalgae production costs vary between USD 190 and 7000 per ton of dry weight biomass, representing the most cost-effective areas in the equatorial Pacific, Gulf of Alaska, and southeastern South America and less favorable regions, respectively.[214] These disparities are driven by factors such as nutrient availability, capital investment, labor, and operational expenses, underscoring the importance of focusing implementation efforts in naturally and economically advantageous regions. For instance, Coleman *et al.* (2022) estimate that, initially, the cost of using kelp aquaculture for CDR could range from USD 1257 to 17 048 per ton of CO_2.[215] Conversely, Kite-Powell *et al.* (2022), using a techno-economic model, suggest that large-scale seaweed farms (over 1000 hectares) located up to 200 km offshore can achieve production costs between USD 200 and 300 per dry ton (2021 USD).[216] With nearshore support infrastructure and optimal conditions, these costs could drop to USD 100 or even lower per dry ton. While dry biomass cannot be directly equated with long-term carbon storage,[71,217,218] some CDR approaches—such as specific biochar production methods—may offer very low or even negative net removal costs.[219]

Combining marine biomass production with subsequent processing steps, such as biochar production, provides a clearer picture of costs in terms of carbon removal. However, investment costs are still incurred at the beginning of these projects, while carbon removal occurs later. Furthermore, estimating removal rates over time is challenging in restoration projects, as is translating them into cost estimates per ton of CO_2 removed. Different assumptions about the time horizon and discount rates result in significantly different costs per ton of CO_2 removed.

Therefore, for BCE restoration projects, presenting the restoration cost per hectare is also worth considering, if not advantageous. This is all the truer because BCE has numerous other benefits besides CO_2 removal (see Section 3.4), and accordingly, allocating the restoration costs per hectare to "only" CO_2 removal leads to very high costs, while the other co-benefits are not taken into account in this metric. For projects carried out for other purposes, making them viable for carbon markets requires then only "additional" carbon-removal monitoring costs, resulting in extra CDR costs that range between USD 0.75 and 4 per tCO_2 for tidal wetlands and seagrass meadows, respectively.[140]

According to our literature review, the study by Bayraktarov *et al.* (2016)[211] remains the authoritative overview for cost estimates per hectare for the restoration of the various BCEs. We present their estimates below (converted from 2010 USD into 2020 USD, using the consumer price index (CPI) in 2020/CPI2010), and discuss recent advances in the literature. Bayraktarov *et al.* (2016) find a median (mean) total restoration costs, including capital and restoration costs, for mangrove BCEs of USD 2985 (17 780) per hectare, for developed countries the cost increase though to USD 61 887 (53 920) per ha. Goto *et al.* (2025)[213] find a median (mean) restoring costs for mangrove BCEs of USD 2099 (9739) per hectare. This includes areas like in Myanmar with a median cost of USD 62 USD per hectare and in the U.S. with a media cost of USD 48 701 per hectare. Goto *et al.* (2025) present for 111 countries estimates for the total restorable area and a country-specific cost estimate distribution.

Bayraktarov *et al.* (2016)[211] find a median (mean) total restoration costs for salt marsh BCEs of USD 179 844 (1 240 118) per ha with no individual information for developing countries because of a lack of projects and therefore studies. Wang *et al.* (2022)[220] provide a systematic review of salt marsh restoration projects, providing information on the restoration cost per hectare, differentiating between middle- and high-income countries, restoration within and outside of protected areas, and differentiating between different restoration methods. However, they present cost ranges only, no information about the distribution of the cost, finding a cost range of USD 495 067–876 564 and 454 285–694 724 per ha for middle-income and high-income countries for restoration projects outside protected areas. Within protected areas, the two cost ranges change to USD 956 609–1 418 770 and 472 101–633 032 per ha. However, it remains somewhat unclear which of the various restoration methods are included in the total cost estimate. If we limit ourselves to the restoration of vegetation, which is particularly relevant for carbon sequestration, the cost range is only USD 29 945–42 756 per ha and USD 90 765–139 724 per ha for middle-income and high-income countries, respectively. Within protected areas, vegetation restoration is apparently more complex and the two cost ranges rise to USD 89 319–133 875 per ha and USD 123 338–179 867 per ha. However, if additional, far more complex measures are implemented that are particularly important for other ecosystem services, such as the control of invasive species, the costs for these measures alone in protected areas in middle-income countries can amount to USD 7 953 495–11 781 099 per ha.

For other BCEs, no comprehensive cost assessment going beyond case studies has been published since Bayraktarov *et al.* (2016).[211] Accordingly, we report their estimates for coral reef, seagrass, and oyster reef BCEs for completeness. They find a median (mean) total restoration costs for coral reef BCEs of USD 193 321 (390 959) per ha, for oyster reef BECs of USD 225 701 (1 022 305) per ha, and for seagrass BCEs of USD 456 570 (832 435) per ha.

Turning from cost estimates to observed, market transactions, the low cost for mangrove conservation and restoration appear to be consistent with the observed transactions on the voluntary carbon market. Even though only 0.1% of the total transaction volume on the voluntary carbon market covered by Ecosystem Marketplace data are based in on macroalgae restoration in the year 2022, buyers pay an average price of USD 26 per tCO_2 for these credits.[221] Given the observed market prices, low costs are required because these projects would not be realized otherwise. In addition to the traditional voluntary carbon market where in particular individuals seek to offset emissions, there emerged recently a "company" voluntary carbon market which has stronger focus on developing the CDR methods which can scale and about 80 percent of the documented market transactions are forward sales, *i.e.* the removal has not yet been realized.[222] Given the rather limited potential of traditional BCEs restoration, these methods do not appear on this market, however, macroalgae-based approaches. Macroalgae-based approaches are summarized on this market on the somewhat broader category 'marine biomass capture and sequestration'. In this category removal transaction at a price of USD 250 per tCO_2 are recorded but also that already two leading companies in this field have already gone bankrupt. This is not a sign that macroalgae-based methods do not work, but rather a typical development in new technologies and markets. However, these developments in the market have been little documented in the literature to date, which is all the more true when the focus is limited to blue carbon.[223]

Basically, it can be said that the absence of other BCEs projects *via* salt marsh and seagrass meadow, restoration on these markets is also consistent with high-cost estimates in the literature, implying that these projects are not financially viable based on carbon credit revenues alone. Still, in contrast to restoration projects, not observing conservation projects of these BCEs is surprising and not consistent with discussed cost estimates above, indicating that unresolved accounting issues to provide appropriate incentives hinder the protection of these ecosystems.

3.6. Role in climate policy

BCE projects, like other CDR methods, can offset CO_2 and other GHG emissions for climate change mitigation purposes. Offsetting can take place at different times if future net-negative emission periods compensate for current and past emissions or in the form of a circular CO_2 economy in which biogenic raw materials replace fossil fuels.[14,52,224] Offsetting can be organized at different levels of accounting, at the national or corporate level, under mandatory or voluntary targets. In turn, various options exist for the integration of the different BCE projects into climate policies with different monitoring requirements and economic incentive schemes. Even though almost all articles emphasize the role of BCE projects for climate change mitigation, effective integration into climate policy is hindered by certain factors, such as lack of enforcement, financial constraints or misguiding state governance.[225–228] Further, it's unclear how macroalgae as a BCE fit into existing climate policy frameworks.

The estimates of carbon storage rates in Section 3.3 are 'reported' and can be used, for example, in integrated assessment modelling. However, including BCE habitats and projects in climate policy and counting them towards targets requires more sophisticated and standardized monitoring, reporting, and verification (MRV) processes. The IPCC defines MRV for emissions or removal tracking as a structured process, involving systematic collection, analysis, and dissemination of data related to the CDR project and conservation efforts. The IPCC emphasizes the importance of standardized frameworks and methodologies to enhance the reliability and comparability of data across different projects and regions. The monitoring process involves tracking various parameters, like pH, turbidity, suspended solids, and CO_2 levels using specific sensors[229] and involves then the preparation of regular reports that detail the status, trend, and outcome of the project[230] which can then be verified by an independent third party to validate the reported data and to ensure its accuracy and reliability.[231] The MRV requirements vary across different BC projects, depending in particular on whether carbon is stored in biomass and soils, or storage is achieved in subsequent processing steps, like for example in the case of macroalgae cultivation and harvesting to use the biomass for biochar or BECCS.

Furthermore, the derived carbon accounting implications depend on the underlying liability framework. The inclusion of BC habitats into national inventories implies a strong liability framework since the inventories are updated by states, in principle ensuring that issues of non-permanent carbon storage are addressed. Regarding the inclusion in national inventories, the IPCC (2014) wetland guide was decisive since it defines rules for including BC into climate policies.[122]

Under the Paris Agreement 74 countries mention coastal wetlands in their Nationally Determined Contributions (NDCs),[19] while the wider and related term of coastal and marine nature-based solutions is mentioned by 97 of 148 countries.[232] Yet, quantitative targets regarding the BCE contribution are not (or only indirectly) specified in the NDCs. Pledges from 55 countries include coastal carbon sequestration[19,233] and are moving towards accounting for BCEs in their GHG inventories.[234] These pledges include (i) protection and restoration of coastal ecosystems, (ii) established protected areas and coastal zone management, and (iii) climate-ready fisheries communities.[19,235] In particular small island states have large potentials to include BCEs in their NDCs to offset their low fossil fuel emissions or land use emissions. Friess (2023) provides an overview of mangrove and seagrass BCE integration into small island states' climate change mitigation targets. Around one fourth have implemented quantitative BCE targets. Large potential remains to further specify and include BCEs, especially seagrass, into the NDCs and national climate change policies of small island states if scientific and policy capacity or funding is provided.[236]

Our systematic review did not identify comprehensive reviews of BCE or BC in general in countries' inventories and thus is limited to individual countries national inventories with respect to representation of coastal ecosystems.

3.6.1. Australia. Australia includes carbon removal and emissions resulting from changes in coastal ecosystems in its national inventories. Under the category "wetlands converted to forest land" Australia reports 4.7 $MtCO_2$ of carbon sequestration for the period 2021–2022, which resulted from mangrove forests growing on tidal marshes. Net emissions from lost mangroves in the "land converted to settlements" category amount to 2.6 $MtCO_2$ in the 2021–2022 period. Although Australia also accounts for seagrass loss due to capital dredging, no emissions occurred in this category for the period 2021–2022. Beyond accounting in the national inventories, the Australian Bureau of Statistics reported the spatial extent of salt marshes and intertidal seagrass as part of the national ocean accounts. Here, an additional annual carbon sequestration of 10.1 $MtCO_{2eq}$ by salt marshes is reported for 2021, differentiating between sequestration in vegetation and soil, and also with attribution to national states.

The Australian government tried to report the spatial extent of kelp habitats, but data availability was insufficient.[237] The FullCAM accounting tool[238] applied in the National Inventories has been extended to measure the carbon removal and climate benefits of coastal ecosystem restoration (BlueCAM);[239] however, the method is still contested.[75,240,241]

3.6.2. United states. In the category "coastal wetlands remaining coastal wetlands", the U.S. accounts for carbon stock changes and CH_4 emissions of all privately- and publicly-owned coastal wetlands (*i.e.*, mangroves and tidal marsh) along the oceanic shores of the conterminous U.S. (excluding Hawaii and Alaska) and nitrous oxide emissions from aquaculture, resulting in a net removal of 4.3 $MtCO_{2eq}$ in 2021.[242] However, there were periods in which the loss of vegetated coastal wetlands to open waters due to hurricanes exceeded removals, resulting in a net source of about 5.7 $MtCO_{2eq}$ in this category between 2006 and 2011.[242] In the most recent inventory, seagrass is not included because of insufficient data. However, initiatives at the state level are underway to improve the accounting for BCEs, including seagrass meadows and their carbon sequestration, on the state and subsequently on the national level, led by Maryland and North Carolina.[243,244] For the U.S. inventory, emissions and removals are calculated using the stock change method for soil carbon (C) and the gain-loss method for biomass and dead organic matter.[242] Further potential and actual inclusion of coastal wetlands in the national inventories of the U.S. are analyzed, for example, in Holmquist *et al.*,[245–247] supported by further studies at the state level.[233,248–250]

3.6.3. Japan. Japan accounts for approximately 0.35 $MtCO_{2eq}$ of removals by mangroves, seagrass meadows, and macroalgal beds in its national inventory in the period 2021–2022.[251] The inclusion of salt and tidal marshes is planned for the future. While the inventory is very transparent about the data and methods applied, it does not report the removals by different blue carbon methods.

3.6.4. Other countries. In the EU, only Malta reports blue carbon removal in the category "wetlands remain wetlands", though in 2022, it accounted for only 14 tCO_{2eq} of removal.[249,250] Other countries with notable blue carbon habitats, like Indonesia, which have considerable emissions and carbon removals, account for positive and negative fluxes from primary and secondary mangrove and swamp forests in the category forest land but do not report the specific BCE contribution.[251] Yet, in these regions, BCE habitats in general, and mangrove habitats in particular, require accurate accounting regarding their magnitude and, in turn, their contribution to reaching emissions targets.[252,253] Further details on the inclusion of BCEs in national inventories can be found in Green *et al.* (2025).[254]

3.6.5. Result-based incentive schemes. The greater transparency concerning the different categories and thus the disclosure of the contribution of BCEs in the national inventories of Japan and Australia might be explained by the fact that, in addition to project-based incentives, result-based incentives exist *via* credit trading. Japan combines various carbon pricing instruments, including both complementary and voluntary schemes, at the regional and national level, which allow for J-credits. These credits certify the amount of GHG emissions reduced or removed by sinks in Japan.[255] For example, in 2023, Japan launched its emissions trading system for companies, GX-ETS, initially a voluntary scheme but supposed to become a compliance system from 2026 onwards[256,257] The GX-ETS allows for the inclusion of carbon removal credits, which can be generated by DACCS, BECCS, and blue carbon projects (further CDR methods will be integrated later). Blue carbon projects are certified as J-Blue carbon credits, supervised by the Japan Blue Economy Association, with the aim of realizing the various co-benefits resulting from such projects.[255,258] While in principle the inclusion of international credits is possible, the focus lies on domestic CDR activities.

Similarly, the Australian Carbon Credit Unit (ACCU) scheme allows participants to earn credits for carbon emissions reduction or removal projects, which then can be sold to the government or companies. Eligible carbon removal projects also include restoring blue carbon ecosystems for which the "tidal restoration of BCE method" is applied.[259] Application of this method to earn blue ACCUs is expected to unlock Australia's blue carbon potential for mitigation purposes, and reintroducing tides to all suitable areas of Australia's coast is expected to provide about 58 $MtCO_{2eq}$ removals over 25 years.[260] The EU does not currently include a result-based crediting scheme for incentivizing BCE projects, even though the EU climate policy design, with the land-use, land-use change, and afforestation (LULUCF) pillar and the possibility to transfer removals exceeding the country-specific targets in this sector to other sectors, could be augmented to include BCE projects under the given strong liability framework.[261]

3.6.6. Voluntary carbon market. In addition to these predominantly domestic schemes, the international VCM, on which companies and private persons either directly or *via* carbon traders buy carbon credits for offsetting purposes, also includes blue carbon projects. Trading blue carbon credits on the VCM is an option for financing the conservation and restoration of BCEs.[262] Article 6 of the Paris Agreement enables international carbon market cooperation to transfer carbon credits, transferring Internationally Transferred Mitigation Outcomes (ITMOs) through bilateral or multilateral agreements (art. 6.2) and a crediting mechanism (PACM) – succeeding the Clean Development Mechanism (CDM) – to trade carbon credits under UNFCCC (art. 6.4).[263] Countries have already announced plans to use the mechanism provided by the Paris Agreement. Certain countries, such as Switzerland, have already established agreements to trade ITMOs, and BC credits are likely to neatly fall within that scope.[263]

Perera *et al.* (2025) reviewed 70 blue carbon projects, which have an aggregated volume of 154 $MtCO_2$.[264] However, the majority of projects are still under development, and only a small fraction already provides credits. In the period 2020–2023, only BC credits representing about 11 $MtCO_2$ have been traded in the voluntary market, representing less than one percent of the transactions in the market,[265] although their offset market potential is projected to grow strongly by 2030.[266] According to Perera *et al.* (2024), 57 projects are registered under Verra, 12 projects under Plan Vivo, and one under the Australian Carbon Credit Unit scheme. Of the 70 projects, only one project addresses seagrasses; the other projects focused either solely on mangroves or mixed ecosystems in which other terrestrial ecosystems are combined with mangroves.[264] Not all projects are removal projects; 12 projects aim at avoiding emissions (reducing emissions from deforestation and forest degradation, REDD), and 16 projects combine emissions avoidance and removal.[264] Given the strong mangrove focus, it is not surprising that almost all blue carbon projects are in tropical countries,[265] and of the 70 projects reviewed by Perera *et al.* 2024, 13% are located in Mexico, and 10% in Myanmar and India each. According to Perera *et al.* 2024, only 24 projects provided information on non-permanence risk assessments and these projects applied either Verra's Agriculture Forestry and Other Land Use (AFOLU) Non-Permanence Risk tool version 3.0/4.0 or Plan Vivo approved 'Assessing risk and setting the risk buffer' method to determine how many credits needed to be set aside in a buffer account to insure against non-permanence events.[264]

The dominance of mangrove projects among the BC projects in the VCM shows that this market and the corresponding actors have a strong focus on forest-related projects and, in turn, on corresponding methods. A rather new market segment has recently emerged, going beyond the more traditional VCM project developers by large-scale market commitments of companies to buy carbon removal. This initiative is led by Microsoft[267] together with other companies under the Frontier initiative,[268] alongside the X-price carbon removal initiative.[269] Unlike the previous project developers on the VCM, these initiatives rely heavily on forward trading, *i.e.*, concluding contracts with delivery of the removals in the future, and the focus is less on current prices and more on the development potential for larger quantities of carbon removal. Obviously, in this segment, small-scale, traditional BCE projects are not represented since their prospects of scaling up are too limited. However, novel blue carbon projects are represented where carbon storage is achieved in subsequent processing steps. Among those, four ocean-based CDR companies that made it to the X-prize final, one company is utilizing a macroalgae-based approach to provide marine biomass for products and storage. Accordingly, given the project demand for CDR, novel blue carbon methods are likely to become more important.[270]

Nevertheless, traditional BC methods will remain a small but important pillar in future (climate) policies because of their various co-benefits.[45] Yet, to realize these co-benefits (see Section 3.5), payment schemes based on a project level or contract level rather than result-based payments are potentially more appropriate.[271,272]

3.7. Legal, political, and social feasibility

Within the international policy arena, blue carbon governance is embedded in a fragmented legal and regulatory landscape.[273] The starting point for blue carbon governance occurred at the 16th UNFCCC Conference of the Parties (COP 16) in 2010. A small group of scientists (namely the Blue Climate Coalition) introduced the mitigation potential of blue carbon to the international climate policy community. Subsequently, scientific research on blue carbon expanded – also driven by the UNEP report in 2009 introducing the term 'blue carbon'.

After the adoption of the Paris Agreement under the UNFCCC discussions started around the inclusion of blue carbon in NDCs.[274,275] Nevertheless, to date, only a limited number of countries have incorporated blue carbon into their NDCs or established quantitative targets for blue carbon mitigation (see Section 3.6).[276,277] Further political momentum emerged at COP 21 with the establishment of the state led 'Because the Ocean' initiative, which emphasized the role of coastal systems in climate action. 23 countries launched this initiative calling for more recognition of the ocean (and also BCEs) in NDCs.[275]

The Paris Agreement framework further supports the project-based protection of BCEs by enabling the trade of carbon credits in accordance with Article 6.[276,278] Several market-based mechanisms were already in place during the early stages of efforts making blue carbon governable, notably the Voluntary Carbon Standard (VCS). Verified projects under the UNFCCC Clean Development Mechanism (CDM) could be included. The REDD+ framework provided a relevant policy instrument for mangrove forests. With the inclusion of Article 6 in the Paris Agreement, new pathways for carbon market cooperation were established. However, the detailed modalities for the implementation were only finalized nearly a decade later.[275]

To strengthen the linkage between climate protection and biodiversity restoration, the Convention on Biological Diversity (CBD) integrates mangrove conservation into sectoral programs and provides incentives for their protection.[279,280] This overall target is further operationalized through the Kunming–Montreal Global Biodiversity Framework (KMGBF), which establishes a target to restore 30% of degraded ecosystems by 2030, thereby encouraging states to initiate BCE projects.[7,37] Completing these efforts, the Ramsar Convention on Wetlands sets targets for the initiation, completion, updating and dissemination of national wetland inventories.[37]

In addition to the international framework, national strategies and policies implement these objectives further (*e.g.*, in the NDCs).[281] Several states have launched blue carbon protection strategies. Indonesia, home to a large share of the world's mangrove BCEs, has developed a National Strategy for Mangrove Ecosystem Management (SNPEM) to stop the rate of damaged mangrove ecosystems (currently 52%) and achieve their goal of 3.49 million ha in good condition in 2045.[282–284] China integrated blue carbon into its strategy to become carbon neutral by 2060, and into several other government documents. In 2014, China launched its Blue Carbon Plan, which focuses on increasing the carbon stock rather than protecting BCEs.[285] Australia has protection legislation in all sub-national states, which is implemented by relevant agencies. In addition, for example, seagrass meadows are also indirectly protected through measures for fish habitat protection.[286]

To provide an effective polycentric framework in which different interactive subsystems interact flexibly and adaptively,[287] local communities and stakeholders should be involved,[7,272] power imbalances reduced,[288] and benefit-sharing rules established[7,272] to implement projects successfully. Yet, blue carbon protection and accounting for their carbon sequestration can create tensions with the local acceptability and feasibility when they interfere with the current uses of coastal and marine areas.[272,289] Issues can range from nuisance for beach users and the local population caused by beach wrack at the shores,[290–292] seaweed overgrowth[293] to impacts on livelihoods such as tourism, access to food, firewood, or the coast.[294–298] Restoring and enhancing coastal ecosystems can necessitate changes in the local populations' behaviors that might have led to the degradation of BCEs in the first place.[289,299] There can also be positive effects on fish stocks and, in turn, on livelihoods. These livelihood benefits might also result from new jobs, though these might be low-skilled or temporary.[300] This is also reflected in an analysis of 8 local coastal management plans in the Philippines, where carbon sequestration was less frequently covered than general management activities or other ecosystem services.[296]

Still, the policy and legal framework contains barriers for successfully promoting BCE management and restoration processes.[273,301,302] The legal status of coastal ecosystem services is often unclear, which results in their direct and indirect benefits and uses rarely being considered in decision-making,[303,304] suggesting that property rights assignment or enforcement is mostly lacking. Effective and just governance would have to ensure that local communities, often not the owners of land rights, participate and are acknowledged when their use rights are affected by the restoration or enhancement of BCEs.[273,305,306] It is essential to understand and account for both the pressures on BCEs and the co-benefits they provide. In contrast to national or international actors, local actors have, for example, been found to be highly aware of the provisioning, supporting, or regulating services but a lot less of carbon storage.[307–311]

Our SLR found 45 empirical studies on public or stakeholder perceptions of blue carbon management approaches or projects. This is a small share of the overall publications. Like Thomas (2014),[312] we find no publications that empirically analyze social aspects of blue carbon management before 2015. Only in 2021, we see a substantial rise in publications. This development lags behind the rise in attention for blue carbon in the marine CDR literature that started in 2014.[313] At the time of our systematic literature search in 2024, we found 28 studies that looked at mangroves, 14 at seagrasses, 6 at (salt-) marshes, and none were about kelp or macroalgae in terms of public perception. This means some studies also analyzed perceptions of more than one ecosystem, and 26 took a broader perspective on blue carbon. 27 publications looked at BCE perceptions in local contexts, while others looked at broader socio-political perceptions of blue carbon management. The geographical distribution of studies reflects the natural distribution of the ecosystems. Many studies looked at mangroves in South-east Asian countries, especially the Philippines (n = 13; some studies look at several countries) and Indonesia (n = 11) are dominant in the dataset. Furthermore, studies in Africa focus on Kenya and Madagascar (4 each). Among industrialized countries, Australia (n = 4) and the US (n = 3) are most prevalent. We found 28 studies looking at stakeholder perceptions and 21 at public (lay person) perceptions. Only 4 also assessed expert views. Most studies used surveys (n = 22) or interviews (n = 18). Quite often, more than one type of participant was addressed, and a method mix of interviews, focus groups, group discussions, surveys, or content analysis was used.

The specific focus on blue carbon limits the scope of the literature we found, as there are also studies that look into the public and stakeholder perceptions of coastal ecosystem (restoration) without a focus on carbon sequestration and storage.[314,315] At least until 2017, most of the few studies on marine and coastal habitats focused on perceptions of coral reefs and beaches. Very few looked at mangroves, wetlands, intertidal areas, or seagrasses.[316] This has changed at least for mangroves, especially in South-East Asia.[317] This literature can also provide insights into the perceptions of BCEs but lies largely beyond the focus of this review.

Analyses of media coverage of seagrass or interviews with experts on seagrass or macroalgae report low interest in these species.[292,318] This stands in contrast to the awareness of seagrass in local communities in the Philippines where study participants were mostly aware of its ecosystem services (except carbon storage),[311] pointing towards a potential difference between perceptions in rural and urban areas.[319,320] Studies in industrialized countries found that, while on the one hand a sense of responsibility for restoring the oceans[309,321,322] and a perception of oceans as fragile led study participants to more positive views on coastal restoration, on the other hand stronger beliefs in the oceans' capacity to adapt led to lower support for coastal restoration.[321]

Turning to the appeal of voluntary carbon offsets from BCEs, consumers seem to be willing to pay a premium for mangrove restoration and conservation. According to Forest Trends' Ecosystem Marketplace (2024),[221] consumers were willing to pay about 4 times more for mangrove restoration and conservation carbon credits in the year 2023 compared to the average price (USD 26 per tCO_2 compared to the average price of USD 7 per tCO_2).[221] However, mangrove restoration constituted only 0.1% of the total transaction volume on the voluntary carbon market covered by Ecosystem Marketplace data in the year 2022 and declined even further in 2023. For other BCEs, the number of projects is too small, and in turn, there is too little price and transaction data.

On a general socio-political level that abstracts from local contexts, the conservation and restoration of coastal ecosystems is preferred over other marine CDR methods that are perceived as more technical[317,321,322] but as the marine ecosystems are less well known compared to land-based counterparts, these activities evoke less positive reactions compared to afforestation.[309,323] This points toward a mismatch between removal potential and public support and thus political feasibility.[324] While ecosystem restoration has many benefits, among them emissions avoidance, the additional removal potential is limited.[325]

4. Discussion

In general, 15 types of Blue Carbon Ecosystems (BCEs) and further blue carbon methods could be considered. However, our search yields mainly results related to the traditional BCEs: mangroves, seagrass meadows, and salt marshes with an increasing share of macroalgae among the publications which are related to blue carbon and carbon dioxide removal (CDR).

According to recent estimates, mangroves, seagrass meadows and salt marshes cover a combined estimate of 25–52 million ha which is at the lower end of earlier estimates, that ranged up to 185 million ha.[44] Improved methods for estimating BCE coverage have produced a more realistic picture of their distribution, but the discrepancy between earlier estimates can also partly be explained by the recent annual losses of up to 2% in these ecosystems. The extent of macroalgae is much larger, with recent estimates suggesting a total area between 606 and 722 million ha (including all seaweeds). Brown algae alone cover an estimated area between 150 and 250 million ha.[10]

The four main BCEs are estimated to provide an aggregated mean natural carbon sequestration of 270 MtC year^{-1}, (106—516 MtC year^{-1}) with the largest contributions from wild macroalgae (173 MtC year^{-1}) and wild kelp (56 MtC year^{-1}). This value is relatively small compared to the total marine and terrestrial annual net carbon uptake, which is estimated to be 2880 and 2302 MtC year^{-1}, respectively.[326] These sequestration estimates bear uncertainties, not least because studies vary in how far they translate net primary production into carbon sequestration achieved *via* long-term carbon burial in sediments. Furthermore, across studies area-based and global aggregated estimates do not align, even for otherwise robust studies, indicating skewed distributions across the estimation bandwidth but also using different assumptions about spatial coverage of BCEs.

In addition, seeking to determine the climate change mitigation, the production of other GHGs (*e.g.* methane) during organic matter decomposition can partly offset the net CO_2 sequestration. Furthermore, it is difficult to determine anthropogenic interference with BCEs, as this is not limited to the direct degradation of these ecosystems for use, but also includes numerous indirect impacts, particularly through over-fertilization induced eutrophication,[47] but also the effects of bottom trawling, for example.[327] Accordingly, in the context of BCEs, but also in general, properly estimating all components of the natural carbon sink is necessary to improve estimations of remaining carbon budgets and to avoid confusing anthropogenic activities with natural carbon sequestration.[76,77,328]

In the literature we surveyed, we found estimations of the aggregated mitigation potential of BCEs through conservation ranging from 60 to 96 MtC year^{-1} by 2050. Focusing instead on the year 2030, the upper range increases to 218 MtC year^{-1}, indicating the uncertainties and inconsistencies in estimating emissions changes once more. However, it also indicates that emissions from degradation will decrease as the remaining BCE area continues to shrink.

Restoring the traditional BCEs, mangroves, seagrasses and salt marshes could remove between 14 and 232 MtC year^{-1} by the year 2050. This estimate does not include macroalgae-based approaches, since the net primary production potential if farmed is about one magnitude larger than with traditional BCEs, but the actual carbon removal than usually requires additional carbon storage measures. Accordingly, we present estimates on the macroalgae-based removal potential up to 216 MtC year^{-1} in Table 5, but the upper estimate is poorly constraint. Combining macroalgae-based approaches with bioenergy utilization and carbon capture and storage (BECCS) or biomass sinking into the deep ocean provide theoretically much larger CDR potentials (*e.g.* up to 1430 MtC year^{-1}). In summary, established solutions – primarily traditional BCEs – are estimated to compensate about 1–3% of total annual CO_2 emissions.[135,329,330] This climate change mitigation potential could increase if emerging solutions are fully deployed and integrated into climate mitigation pathways.[135]

At the same time, theoretical potentials should not be confused with economically viable sequestration potentials since the upper ranges presented in the literature were usually obtained with earth-system model simulation studies, neglecting the considerable supply chain and in turn practical and therefore cost challenges involved in realizing these potentials. However, it is difficult to derive reliable cost estimates for CO_2 removal, *i.e.*, denominated in tCO_2 removed, because the studies take different components of the total costs into account, *i.e.*, the numerator is determined differently, and the studies also varies in terms of how they determine the net CO_2 removal, *i.e.*, the denominator is also determined differently. In terms of costs, it is often unclear whether capital costs are taken into account, for example, and in terms of net CO_2 removal, it is unclear to what extent CO_2 emissions from operational implementation or other greenhouse gases are taken into account.

The derivation of robust cost estimates is practically limited to the restoration of mangrove forests, as there are current studies that not only specify a cost range, but also how these costs are distributed within the range, and we can compare these with transactions on the voluntary CO_2 offset market. Hence, while the upper range of the cost for carbon removal *via* the restoration of mangroves is in the ballpark of 500 USD per tCO_2, Goto *et al.* (2025)[213] show that the cost distribution is considerably skewed to the right and that a large fraction of the mangrove-based CDR potential can be realized at cost below USD 20 per tCO_2. This is broadly confirmed by actual observed prices on the voluntary offset market where a very small amount of mangrove-based carbon offsets are traded, but at a market price of USD 26 per tCO_2 in the year 2022.[221] Accordingly, for the other BCE-based methods, simply taking the mean of the upper and lower cost ranges might provide little insights on the actual economic viability of the methods in different regions.

Turning to BC approaches which have the potential to scale, the focus narrows to macroalgae-based approaches. As discussed above, engineering this BC approach could provide CDR one magnitude larger than that of the other, traditional BCEs. However, current cost estimates are inconclusive in this respect, since costs vary considerably across studies and by location. This indicates different assumptions regarding farming and harvesting setups and potential additional remuneration, for example, if marine biomass is used for energy provision (in combination with point-source carbon capture and storage to achieve CDR) or as part of a circular economy. Within the voluntary offset market, focusing on that segment where large companies and CDR buyer clubs are active, macroalgae-based approaches are among those marine CDR methods where market participants expect large developments, both in terms of scaling-up and cost reductions. However, these developments are not yet well covered in the scientific literature, indicative for the systematic review of the (blue carbon) CDR literature as such: scientific questions related to carbon burial, net climate impact, co-benefits and side-effects are discussed in the scientific literature. Highlighting CDR related questions, more comprehensive overviews and analyses are provided in the grey literature by an increasing share of publications presented by consultancies. This does not necessarily have to be a negative development, as the latter can gather, aggregate, and interpret information at a completely different speed. At the same time, however, it is of course important that these communities remain in lively exchange with each other, including which question to answer.

For example, this includes improving cost estimates for BCE removal and how these estimates are interpreted within IAMs. While global IAMs emphasize the need for large-scale CDR, they do not yet explicitly include BCE solutions like mangrove, seagrass meadow, and/or salt marsh protection and restoration. It is unclear to what extent BC methods are implicitly represented as part of aggregated, generic CDR cost functions. However, global IAMs usually assume a global unique CO_2 price to assess the viability of individual CDR options. In reality, CO_2 (market) prices vary regionally, as does demand for CDR.[331] Therefore, future realization of BCE CDR projects will not only be determined by environmental conditions but to a large degree by the regional policy framework and the ambition of the emissions reduction target.

The national policy context is relevant for providing a CO_2 price signal for removals and defining the carbon accounting method that determines the financial rewards. Note that these rewards are not necessarily on a per-ton basis, but could also be project-based, the latter allowing to put more emphasis on the realization of co-benefits. The impacts of conserving, restoring, or managing BCEs are not limited to carbon sequestration. They also include various co-benefits, such as preserving marine biodiversity, providing nurseries for different species, and contributing to coastal protection. At the same time, negative side effects can arise for example from the production of other GHGs, which can offset the climate benefits of carbon sequestration in some contexts. While co-benefits are commonly discussed in the literature, few papers discuss side effects. From the literature, it is unclear whether carbon sequestration is the main ecosystem service provided by coastal management approaches involving BCEs.

This review shows that BCE should play an important role in national climate mitigation strategies – particularly in countries with degraded or threatened coastal and marine ecosystems. Still, to achieve large-scale implementation certain constraints need to be overcome. Effective implementation depends on the integration of local, cultural, ecological, and logistical perspectives (incl. technical expertise). Further constraints arise from rather weak policy and legal frameworks as well as limited institutional capacity to manage existing coastal ecosystems. In addition, potential high technical and implementation costs (large cost estimate ranges), substantial upfront capital requirements and the questions of funding sources pose challenges to sustained project implementation or deployment.[8,332–334] Clearly, different national conditions, both biological and regulatory, necessitate distinct strategies for safeguarding and expanding BCE. As discussed in the regional profiles, the land use, land use change and forestry pillar of the European Union could be opened to include BCE (mainly seagrass meadows). Since this pillar of EU climate policy is designed to compare carbon sequestration to a historical baseline, including BCE would actually imply a preservation mandate, increasing also the incentives for stricter agricultural regulation in terms of nutrient runoff.[264] In Japan, carbon removal *via* BCEs is included in the company emissions trading scheme, which is supposed to become a compliance system from 2027 onward. In other regions, however, a policy that is not solely focused on CO_2 sequestration may be more effective. This is particularly true of the U.S., given its departure from climate policy. There, local coastal residents and politicians can be convinced to promote BCEs due to their numerous co-benefits, especially with regard to coastal protection. At the same time, renumeration on the voluntary market shows that people are willing to pay a premium for BCE (in this case mangrove) based CO_2 offsets, indicating that they are well aware of the extra, beyond carbon sequestration services provided. This is confirmed by the reviewed literature on social acceptance which shows that BCE CDR methods are preferred to more technical CDR methods.

Having the EU, Japan, and the U.S. in the spotlight, it is important to recognize that BCEs management activities risk reproducing or deepening existing inequalities if policy and governance frameworks do not adequately address issues of equity, particularly in the Global South. Aligning ecological objectives with principles of social justice requires that local communities – whose livelihoods and food security are closely linked to coastal resources – be recognized as active participants in the design, implementation, and benefit-sharing of BC activities. This, in turn, requires the integration of social dimensions into regulatory frameworks. Clear and enforceable rules are therefore essential to integrate local communities with their knowledge and safeguard local rights to prevent adverse outcomes such as land grabbing and dispossession.[335,336]

5. Conclusion

With the start of the IPCC's 7th Assessment Cycle and the establishment of CDR-related policies and targets, a comprehensive understanding of the current state of evidence for CDR is required. While previous research suggests that a substantial and rapidly growing body of evidence on CDR exists, the few available overviews of the field have quickly become outdated.[337] Accordingly, Lück *et al.* (2025) applied a systematic mapping technology to comprehensively analyze the increasing literature of CDR. However, Lück *et al.* (2025) consider "all" CDR methods (Afforestation/Reforestation, Restoration of landscapes/peats, Agroforestry, Soil Carbon Sequestration, Blue Carbon Management, EW, OAE, Ocean Fertilization/Artificial Upwelling, BECCS, DACCS, Biochar), putting emphasis in their analysis on how the field is developing in general, including 5339 documents (100–600 per CDR method) in their review.[20] However, the large scope of the study prevents detailed insights regarding specific CDR methods. Furthermore, different CDR methods face different challenges in terms of climate policy. For example, methods involving forests or BCEs are complicated by the fact that deforestation contributes significantly to anthropogenic CO_2 emissions. Accordingly, the overview of Lück *et al.* (2025)[20] is followed by a series of CDR-specific systematic reviews, the purpose of which is to provide more detailed insights.

As part of this series, this comprehensive systematic review provides an updated foundation of knowledge across multiple dimensions critical for understanding of BCE approaches for CDR. Based on 2622 articles from peer-reviewed and gray literature, our analysis offers refined estimates of carbon sequestration rates (270 MtC $year^{-1}$ across four main BCEs), updated coverage assessments (25–52 million ha for traditional BCEs, 606–722 million ha for macroalgae), current cost ranges (USD 5–1700 per tCO_2), and an evaluation of co-benefits and potential interactions with other CDR methods. However, significant challenges remain in translating these scientific advances into effective climate policy frameworks. There is a central gap between natural science-based estimates of carbon sequestration or removal potentials and how they are accounted for or omitted in climate policy.

Comparative analyses detailing the different design options for the integration of BCE approaches into climate policy are still sparse and focus primarily on the voluntary carbon market. While some papers have analyzed how often terms related to

BCEs appear in countries' NDCs, a comprehensive review of the role of BCEs in national inventories has yet to be provided. Here, our review relied on individual national inventory data from each country. Accordingly, the next wave of studies should track how such regional incentives schemes like in Australia and Japan have affected BCE conservation and restoration. These studies should assess how such carbon sequestration incentive schemes compare to broader conservation and restoration efforts, like for example the Kunming-Montreal Global Biodiversity Framework as an element of the Convention on Biological Diversity or the Ramsar Convention.

Furthermore, developing BCE projects involves various local stakeholders and beneficiaries. Providing financial revenues only for one BCE ecosystem service, namely carbon sequestration, might not result in a restoration strategy that maximizes welfare. For example, a restoration strategy targeted at coastal protection might be designed differently.[272] However, a restoration strategy that benefits coastal inhabitants the most, either by increasing fish stocks or by providing coastal protection, could call into question whether the project is "additional" in a climate policy context. But this is a crucial prerequisite for receiving financial payments in exchange for carbon sequestration. At the same time, if the carbon sequestration is considered a co-benefit, the additional costs per tCO_2 would be very low (including extra monitoring, reporting and verification efforts to quantify the carbon sequestration $<$USD 10 per tCO_2).[140]

The next wave of studies would benefit from considering BCEs as part of a CDR portfolio strategy which also considers potential interactions. For example, BCE approaches might benefit from mCDR strategies involving artificial upwelling, ocean iron fertilization, or ocean alkalinity enhancement due to nutrient feedback or might even mitigate negative side-effects of such measures. Such a combined strategy would require properly addressing carbon sequestration through financial reward systems while providing additional incentives to realize the co-benefits and interaction effects. A strategy that combines financial incentives for multiple benefits and respects or enhances local livelihoods can also positively affect public perception on both local and broader societal levels.

Author contributions

CR: conceptualization, project administration, data preparation, methodology, writing – original draft, writing – review & editing; ISG: data preparation, writing – original draft, writing – review & editing; DAF: conceptualization, writing – original draft, writing – review & editing; PG: conceptualization, writing – original draft, writing – review & editing; DK: conceptualization, writing – original draft, writing – review & editing; JK: conceptualization, writing – original draft, writing – review & editing; SL: data generation, methodology; CMP: conceptualization, writing – original draft, writing – review & editing; JMH: conceptualization, writing – original draft, writing – review & editing; CM: conceptualization, validation, writing – original draft, writing – review & editing; TT: conceptualization, writing – original draft, writing – review & editing; RV: conceptualization, writing – original draft, writing – review & editing; WR: conceptualization, project administration, validation, writing – original draft, writing – review & editing.

Conflicts of interest

There are no conflicts to declare.

Data availability

The data supporting this article have been included as part of the supplementary information (SI). The SI file includes all primary articles identified during the data search stage. Further inquiries can be directed to the corresponding author. Supplementary information is available. See DOI: https://doi.org/10.1039/d5ee04922a.

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