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Seascape carbon management beyond wetlands as eligible blue carbon activities

Scientific review

Ana M Queirós, Mingxi Yang, Saskia Rühl, Cornelia Rindt, Melissa Ward and Steve Crooks

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ACRONYMS / ABBREVIATIONS

C	Carbon
CH_4	Methane
$CHBr_3$	Bromoform
cm	Centimetre
$CACO_3$	Calcium Carbonate
CCM	Carbon Concentrating Mechanism
CO_2	Carbon Dioxide
CO_2e	Carbon Dioxide Equivalent
DIC	Dissolved Inorganic Carbon
DOC	Dissolved Organic Carbon
Gt	Gigaton
GHG(s)	Greenhouse Gas(es)
ha	Hectare
IPCC	Intergovernmental Panel on Climate Change
ISA	International Seabed Authority
IUCN	International Union for Conservation of Nature
m	Metre
Mt	Megaton
N_2O	Nitrous Oxide
NPP	Net Primary Production
OIF	Ocean Iron Fertilization
Pg	Petagrams
pmol	picomol
POC	Particulate Organic Carbon
REDD	Reduced Emissions from Deforestation and Degradation
SCE	Seascape Carbon Ecosystems
t	Metric tonne

Tg	Teragram
VCS	Verified Carbon Standard
VM	VCS Methodology
yr	year

Definitions

- Alkalinity
  - Total alkalinity is a biogeochemical term usually employed to describe the balance of (ionic) charges in seawater, in the specific context of the ocean carbonate system, of which CO_2 is a part. The carbonate system is a set of equations that jointly describes the dissolution of CO_2 in seawater. The 'ocean alkalinity pool' is a term often employed to refer to total alkalinity, and specifically to refer to the carbonate and bicarbonate ion concentrations (two stable forms of CO_2 when dissolved, or dissolved inorganic carbon), in addition to a few other minor compounds.
- Blue Carbon
  - Organic carbon stored in above- and below-ground biomass and soil pools in tidal and marine ecosystems (e.g., mangrove forests, seagrass meadows, and tidal marshes).
- Carbon Pools
  - A reservoir of carbon that has the potential to accumulate (or lose) carbon over time, which for AFOLU projects or programs encompasses aboveground biomass, belowground biomass, litter, dead wood, soil, and wood products.
- Methodology
  - A specific set of criteria and procedures, which apply to specific project activities, for identifying the project boundary, determining the baseline scenario, demonstrating additionality, quantifying net GHG emission reductions and/or removals, and specifying the monitoring procedures.
- Methodology Development
  - The process by which new or revised methodologies, modules, and tools are developed and reviewed under the VCS Program.
- Methodology Developer
  - An entity that develops a methodology, tool, or module.
- Open Ocean
  - Area located beyond coastal waters, typically beyond the influence of coastal processes.
- Project Activity
  - The specific set of technologies, measures, and/or outcomes, specified in a methodology applied to the project, that alter the conditions identified in the baseline scenario and which result in GHG emission reductions or removals.
- Program
  - A formal or organized program, system, or arrangement for the recognition of activities leading to GHG emission reductions or removals, and/or the crediting or issuance of instruments representing, or acknowledging, GHG emission reductions or removals.

- Recalcitrant
  - Referring to organic carbon that is not easily re-mineralized (i.e., converted to inorganic forms), typically by microorganisms.
- Seabed/seafloor
  - The benthos, the ocean floor, may be comprised of soft-sediment material, such as sand and mud; rock; or biogenic material, such as a coral reef or a maerl bed. Where soft sediment occurs, carbon sequestration may occur if a number of additional, measurable characteristics are also observed.
- Seascapes

  Ecosystem components, which may be adjacent or distal, connected by natural flows of carbon. Includes all autotrophic systems (phytoplankton, bacteria, seaweed, and wetland vegetation), as well as the seabed, from the near shore and deep water adjacent to the continental shelf.
- VCS Program
  - The Verified Carbon Standard (VCS), formerly the Voluntary Carbon Standard is a standard for certifying carbon emissions reductions. VCS is administered by Verra, a not-for-profit organization.
- Verified Carbon Unit (VCU)
  - A unit issued by and held in the Verra Registry representing the right of an account holder in whose account the unit is recorded to claim the achievement of a GHG emission reduction or removal in an amount of one (1) metric tonne of CO_2 equivalent that has *been* verified by a validation/verification body in accordance with the VCS Program rules. Recordation of a VCU in the account of the holder at the Verra Registry is *prima* facie evidence of that holder's entitlement to that VCU.

Executive Summary

This Issues Paper reviews peer-reviewed scientific evidence on the potential to expand carbon finance methodologies under the Verified Carbon Standard (VCS) to encompass Near Shore Seascape Carbon, beyond current ecosystems considered under such mechanisms, i.e., vegetated wetlands. For this assessment, no evidence was therefore reviewed on tidal wetlands, seagrasses, and mangrove ecosystems, for which carbon offset methodologies already exist.

Management of blue carbon ecosystems has become an area of extreme interest in the context of providing nature-based solutions, or nature-inclusive designs for environmental management, that may help to deliver climate change mitigation. Carbon market methodologies for such defined management activities outline procedures that projects must follow to deliver GHG emission reductions or removals that are real, measurable, additional, permanent (>100 years), independently verified, and conservatively estimated. These methodologies must be rooted in scientific understanding of the global carbon cycle so that projects can develop high quality and credible carbon offsets for the carbon markets. Such carbon projects are coming online for tidal wetlands, particularly mangroves. A lack of scientific consensus and in many cases data gaps, amongst other challenges, have prevented the inclusion of other marine ecosystems under carbon market mechanisms thus far. This has limited the ability to harness private finance to further support the growth of ocean carbon conservation. In recent years, science has advanced at pace.

This report reviews current scientific evidence on oceanic carbon cycling and storage to assess the potential of managing other ecosystem types toward climate mitigation. The following summarize our key findings:

1. **Flow of carbon from the ocean surface to seafloor and the deep ocean**

Surface ocean waters absorb CO_2 from the atmosphere, including almost a quarter of anthropogenic CO_2 emissions in the last decade. The direction and rate of air-sea CO_2 exchange at the ocean surface is driven largely by a chemical equilibrium in CO_2 concentration between the atmosphere and the ocean. Photosynthesis at the ocean surface (or organic carbon remineralization to inorganic carbon) enhances (or hinders) the uptake of atmospheric CO_2, there by locally decreasing (or increasing) CO_2 concentration in surface water. The 'biological pump' acts to remove this CO_2 from ocean surface waters, though the physical transfer of organic matter resulting from photosynthesis (by micro and macroalgae) downward through the water column toward the deep ocean and the seabed, where long term storage may take place. Blue carbon projects beyond wetlands aim to enhance surface ocean CO_2 uptake and the transfer of surface ocean productivity to the deep ocean interior and the seabed, through the management of seascape-wide carbon flows.

2. **Long-term carbon sequestration in the ocean**

It is known, with high confidence, that recalcitrant particulate organic carbon within the seafloor is a long-term store of carbon where certain conditions are met (high organic carbon loading, low abundance of oxidants and low physical disturbance of the seafloor). Whether carbon that reaches the seafloor becomes sequestered depends on the properties of that material, natural characteristics of the seafloor and the site's history of natural and human disturbance. Seafloor fauna may enhance carbon burial but also re-mineralization and the balance of both processes is site-specific. The scientific consensus, with high confidence, is that areas below 1000 m deep in the ocean are long-term stores of particulate and dissolved carbon. Despite some limitations in data availability, it is well established that sedimentary processes can increase carbon sequestration, particularly in coastal and shallower areas of shelf seas. However, our understanding of the exact locations and carbon sequestration rates across the ocean seafloor is still limited. Direct measurement of carbon sequestration in the deep ocean is highly challenging. It is thought, with moderate confidence, that the following are also long-term stores of ocean carbon: dissolved inorganic carbon in the deep ocean and within the ocean alkalinity pool; and recalcitrant dissolved organic carbon.

3. Ocean carbon sources and sinks

Beyond wetland systems, other ocean organisms such as micro and macroalgae fix inorganic carbon into their living biomass through photosynthesis. This carbon may be quickly converted back into inorganic carbon if it enters the food web. Biomass that reaches the ocean floor and the deep ocean is subject to slower rates of decomposition. Carbon sources and sinks are thus not necessarily co-located. Loss of source may disrupt the flow of carbon but not necessarily lead to a loss of the sink or storage. Managing such ecosystem components through concerted ocean management and carbon project activities remains challenging, due to limited scientific understanding and governance mechanisms.

4. Seaweed

Of the potential nature-based carbon project activities beyond wetlands reviewed here, the most mature evidence base was found for seaweed habitat conservation (which is usually separated into restoration and afforestation projects) and seaweed farming. It is known, with high confidence, that (non-farmed) seaweed habitats provide a range of biodiversity and other benefits to local people and ecosystems, which should be accounted for alongside local and distal effects on carbon flows. Seaweed is known to have very large CO_2 uptake rates with high confidence, and it is also known that a large fraction of their very high annual production is released into surrounding habitats as particulate organic detritus and, potentially, as dissolved organic carbon. It is thought, with moderate confidence, that some of this particulate carbon ends up in long term storage in the ocean and seabed. It is not known where the hotspots of this accumulation are but first studies tracing these flows are beginning to emerge. We have low confidence about the final destination of seaweed-derived dissolved carbon and its longevity.

Restoration of seaweed habitats has not been done successfully at scales larger than 1 ha outside of scientific projects, but the evidence base is growing. Proximity to natural populations seems to be an important attribute of successful projects. Japan and China present a rich history of seaweed habitat use and management, which should guide the development of the research in this area. Climate-resilient project design is needed to help ensure such projects are *via*ble in the mid-term due to the sensitivity of seaweed to long-term and extreme warming of the ocean. Variability in site and species effects at present suggests that the locations of farms and selection of species are important in developing carbon projects and must be carefully considered to ensure positive effects on net CO_2 rates.

There is uncertainty about how to account for carbon exported from natural and farmed seaweed in a carbon project, and the science underpinning this is not mature. There is no peer-reviewed evidence to suggest that purposeful off-shore sinking of seaweed bales will lead to long term sequestration of this carbon and there is substantial concern about how to do this, while minimizing impacts on ecosystems. For example, a site's proximity to deep water (if sinking seaweed) or distance to ports (if seaweed is intended for non-sinking purposes) both may greatly impact the cost and quantity of carbon offset. The production of other climate active gases by seaweed, including methane and halogens, requires further investigation.

5. Seabed management

Marine sediments are thought to contain 87Gt of organic carbon in their top 5 cm, 0.12 – 0.35 Gt of which are sequestered annually. It has been argued that limiting physical impacts on seabed soft sediments could lead to enhanced carbon sequestration or at least the avoidance of emissions that could result from the disturbance of carbon sequestered there. Sedimentary carbon storage potential is dependent on physical and chemical sediment characteristics as well as biological, hydrographic and anthropogenic influences. Areas with fine-grained, cohesive sediments with low resuspension potential, low turbulence and little biological and anthropogenic interaction are ideal for long-term carbon storage. However, the exact location of such sites is frequently unknown due to poor seafloor mapping, though statistical approaches are being deployed to fill evidence gaps in the meantime. Particularly impactful anthropogenic activities include the use of demersal fishing gear, sediment dredging, aggregate extraction and mining activities, and the establishment of large solid structures on the seabed. And while methane fluxes are largest in near shore environments, continental shelf methane hydrates are estimated to be storing many hundreds of Gt of C. With current knowledge, ideal targets for long term sedimentary carbon storage are thus soft sediment deep-sea environments that are not yet affected by anthropogenic disturbances, but *in situ* verification remains a challenge.

6. Shellfish farming, conservation and restoration

Shell material production in shellfish (a process known as calcification) is a source of CO_2 emissions. The use of shellfish farming to sequester carbon appears, on the surface, to have limited application. Shellfish aquaculture is a globally important and fast-growing industry, and while farmed shellfish carbon stocks are more easily assessable than natural stocks, the net impact on carbon flux and storage remains dependent on the shelf waste material disposal methods beyond harvest. Organic carbon uptake and emissions by shellfish may roughly amount to a neutral net flux rate, though in some cases shellfish can enhance the carbon content of sediments by accelerating sediment accumulation or reducing surface erosion, though in others it may accelerate organic content loss (species dependent).

Shellfish harvesting may also lead to a long-term export of inorganic carbon from the marine system to land. The ultimate fate of the waste shell material could be a key determinant factor in climate mitigation value. Shell material has various uses; the one that best promotes long-term carbon storage is the utilization of shells as building materials, as a byproduct of the fishing processing industry (i.e., no new generation of animals to this end). Aquaculture may also cause a net efflux of nitrous oxide and methane, other important climate active gases, which would require consideration under a carbon project methodology. Mitigation of shellfish reef habitat loss from climate change and other anthropogenically induced pressures through conservation and restoration may, however, avoid additional carbon efflux from these previously stable inorganic carbon stores. This may require additional research on how to limit shell dissolution due to ocean acidification, limitation of demersal fishing which destroys shellfish reef habitats, a limitation of the impacts of shellfish farming on the seafloor, conservation, and restoration to filter-feeding species. Future refinement of waste shell material disposal and recycling would be needed to this end. More research is needed to fully assess the net carbon flux caused by shellfish farming, conservation, and restoration.

7. **Seaweed based alkalinity production**

There is poor scientific understanding of both the scale and rates of processes whereby seaweed contribute to the oceanic alkalinity pool and dissolved organic carbon pool.

8. **Ocean fertilization**

Of the ocean fertilization methods considered to sequester carbon (addition of iron (OIF), addition of macro nutrients, artificial upwelling of deep water), OIF shows the most potential, both from a logistical and theoretical point of view. Large scale adoption of OIF seems feasible from the perspective of the current global iron production, but the environmental impacts of doing so remain largely unclear and would require further research to demonstrate the safety of such approaches.

1 Introduction

1.1 Project Setting

The term blue carbon emerged 15 years ago. It is a concept recognizing the importance of coastal and marine ecosystems as components in the global carbon cycle and that improved management of these ecosystems through conservation and restoration can lead to emissions reductions and reversals (Nellemann and Corcoran 2009, Windham-Myers, Crooks and Troxler 2018), Bindoff, Cheung et al. (2019). The concept of blue carbon evolved to particularly highlight the importance of soil carbon storage as a key carbon pool. During the early actions to include coastal and marine ecosystems within climate mitigation policies, there was a focused push to recognize the significant carbon storage capacity of tidal wetlands, tidal forests (particularly mangroves) and seagrass meadows (Pendleton, Donato et al. 2012). This was because these ecosystems extract CO_2 from the atmosphere and waters and store it on site in biomass and soils. However, it is well established that many other ocean components contribute to the global regulation of the carbon cycle and the climate system. Indeed, in its recent report on Oceans and Cryosphere, the IPCC recognized a programmatic role for blue carbon addressing both climate mitigation and adaptation benefits and adding the 'seascape' dimension to the land and vegetation focus of today's climate change policies (Bindoff, Cheung et al. 2019, Bashmakov, Nilsson et al. 2022).

In the carbon market, the Verified Carbon Standard recognizes Wetland Restoration and Conservation (WRC) as eligible project activities. Two global methodologies provide procedures to meet the requirements of the Standard for blue carbon project eligibility: 1) restoration of mangroves, tidal wetlands, and seagrass meadows (VM0033) and 2) conservation of the same ecosystems (soil modules under VM0007). No global standard yet recognizes other marine ecosystems or habitats.

A number of reviews in the scientific literature have begun to explore the potential to develop broader ocean carbon management as a climate mitigation action beyond traditional blue carbon ecosystems. However, a gap exists between the peer-reviewed scientific literature and how that information can support carbon project development. This gap motivated the need for an integrated research project, where both expert scientists and project developers collaborated to provide a robust review of that literature through the lens of carbon project developer needs.

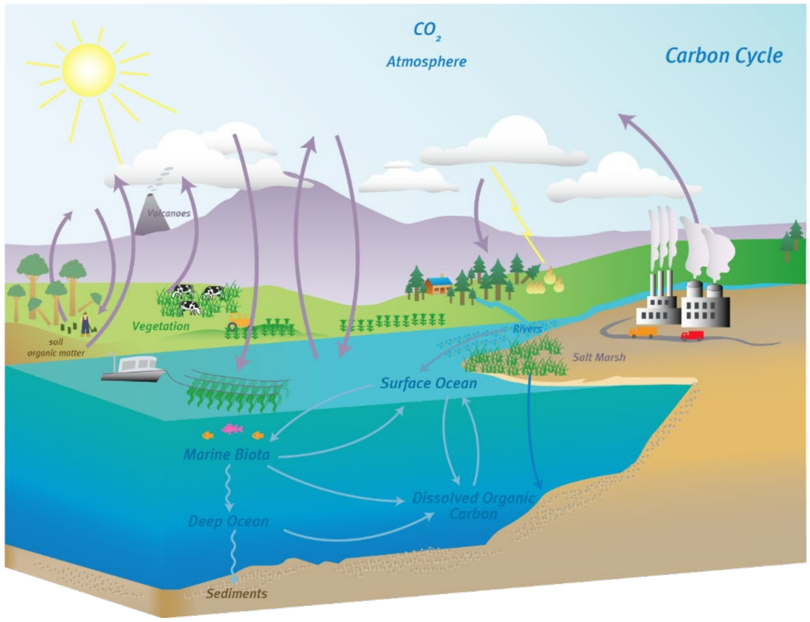

Figure 1: Depiction of carbon cycling between atmosphere, land and open water marine systems. Bindoff, Cheung et al. (2019).

1.2 Objectives

The objective of this report is to provide a summary of the peer-reviewed scientific literature that has heretofore assessed types of seascape-based management of carbon fluxes in coastal and marine waters beyond wetlands, that could become blue carbon project activities in the future. We focus specifically on studies seeking to understand whether such projects could enhance biological processes involved in oceanic carbon dioxide uptake, and their potential effects in broader greenhouse gas inventories. Matters pertaining to avoided emissions resulting from said management are only briefly explored here, where relevant, as these are seen to be better suited to economic studies beyond the focus of this report.

The list of potential new project activity types for which literature is reviewed (Table 1) emerged from extensive voluntary carbon market stakeholder engagement taking place as part of the ongoing Seascape Carbon Initiative ([Projects | Silvestrum Climate Associates | United States]). For each listed project activity type and the ecosystems underpinning them, we endeavor to determine the degree of maturity of the scientific understanding with regard to: i) the underlying carbon (and other greenhouse gases) fluxes; ii) whether any issues have been identified regarding the need to discount any carbon removals due to natural ecosystem processes, which cannot be managed or avoided; iii) future research needs. For each activity, the report is structured to provide an answer to those three topics. In i) we present a clear qualitative judgment of the uncertainty in the science supporting each principle identified. In cases with conflicting evidence, our ambition was also to provide an expert assessment about where the weight of that evidence lies; that is, about what is most likely in each case (where possible). In some cases, this reflected what most studies have found despite variation between studies. In other cases, where the evidence base is narrow, it required us to look more widely into the literature, to seek potential explanations about why differences may have been found between different studies. In this process, we critically assessed the scientific robustness of studies reviewed.

We then used this information (i-iii) to inform about optimal conditions under which each type of potential Project Activity may be successful, from a carbon project activity perspective (Table 1). The authors of this report do not advocate either for or against possible expansion of blue carbon project activity types, and we recognize that blue carbon activities complement the role of larger scale climate mitigation approaches focused on decarbonization.

1.3 Seascape carbon flows beyond wetlands

We now introduce key types of carbon flows (and of other relevant greenhouse gases) across seascapes. Fluxes associated with seaweed are reviewed in greater detail in sections 2.1-2. 2.

1.3.1 Photosynthesis

Biological drawdown of CO_2 (photosynthesis) by marine biota takes up CO_2 from the water, while respiration of organic material returns CO_2 to the water. During the growing season, photosynthesis outpaces respiration, resulting in positive net community production, net consumption of CO_2, and production of O_2. This biological drawdown (along with temperature) is partly responsible for the seasonal variability in CO concentration in the surface waters of the temperate ocean.

1.3.2 Sedimentation

Materials (‘floc’) sinking through the water column towards the seafloor can be made up of inorganic and organic components from various sources. Irrespective of hydrological forces, sinking velocities are dependent on material density, shape, size and composition (Iversen and Ploug 2010, Trudnowska, Lacour et al. 2021). As between 17 and 39 % of sinking particles’ mass is made up of particulate organic carbon alone (Alldredge 1998), this sinking flux constitutes an important carbon export pathway. However, variability in floc abundance on a spatial, as well as daily and seasonal/temporal, basis can be considerable (Lampitt, Wishner et al. 1993). Once matter (particulate and/or dissolved) has reached the seafloor, it may be buried and stored there. In non-turbidic, oxic environments, sedimentation rates are positively correlated with pelagic organic carbon content (Stein 1990). In contrast to coastal regions, in which local sedimentation rates vary seasonally and inter-annually (Rühl, Thompson et al. 2020, Rühl, Thompson et al. 2020), deep open water sedimentation is more consistent (Emerson, Quay et al. 1997).

Deep water convection can lead to the formation and maintenance of nepheloid layers, in which sinking material is prevented from reaching the seafloor for unknown periods of time. Aggregated particulate matter in suspension within these nepheloid layers contains more inorganic material and hosts fewer microbial interactions than aggregates found in the water above (Ransom, Shea et al. 1998).

1.3.3 Sequestration

The process of carbon sequestration, the capturing and long-term storage of carbon (>100 years), is often described in relation to the seafloor. It is regulated locally by the surrounding physical and biological environmental conditions. Blue Carbon habitats, such as tidal marshes, seagrass and mangrove forests, promote sequestration (Mcleod, Chmura et al. 2011), but the actual long-term immobilization of carbon is tied primarily to subsequent sedimentary storage below these habitats. Sequestration is also accepted to occur for carbon, in various forms, that enter regions of the ocean below water depth of 1000 meters. Convective deep-water formation regions at high latitudes, such as in the North Atlantic and the Southern Ocean, have the highest mean sequestration times with more than 400 years, though sequestration efficiency is impacted significantly by up- and downwelling systems and can vary abruptly between adjacent areas (DeVries, Holzer and Primeau 2017).

1.3.4 Lateral carbon flows between reservoirs

Fluxes of carbon in its many forms between the reservoirs are thought to be continuously ongoing, and varied in magnitude, often dynamically and simultaneously operating at various spatial and temporal scales. At the sediment-water boundary for example, there are exchanges of dissolved and particulate matter, driven by physical resuspension and deposition, biological mixing, advective flushing, bio-irrigation and diffusive flux (Rühl, Thompson et al. 2020). The uptake of loose particulates by pelagic fauna can increase sedimentation (pelagic) and burial rates. Recently, the role of dissolved organic and inorganic carbon fluxes in the ocean has also been highlighted, and much less is known about the size and dynamics of those fluxes (Santos, Burdige et al. 2021).

1.3.5 Methane (CH_4) production and consumption

Methane (CH_4) is a greenhouse gas more than 20 times more potent than CO_2, but it is shorter lived in the atmosphere (half-life of ~12 years). CH_4 production occurs predominantly in anerobic sediments (that is, in the absence of oxygen) at salinities less than half that of seawater. However, methane production has also been observed to occur in aerobic environments (i.e., in the presence of oxygen, Hilt, Grossart et al. (2022)). Once released to the water column, CH_4 consumption by marine microbes is rather rapid (lifetime of days), such that significant CH_4 emission to the atmosphere from marine environments is generally limited to very shallow waters.

1.3.6 Nitrous (N_2O) oxide production and consumption

Nitrous oxide (N_2O) is a greenhouse gas that is about 300 times more potent than CO_2 and has an atmospheric lifetime of ~120 years. N_2O is produced mainly through nitrification (oxidation of ammonium to nitrate) and denitrification (anaerobic transformation of nitrate to N_2 in low oxygen waters). The former is thought to be the main production pathway in the oxygenated open ocean (Freing, Wallace and Bange 2012). N_2O is lost from the surface oceanic waters *via* microbial consumption (reduction to N_2) and ventilation to the atmosphere. The oxygen minimum zone (OMZ) of the ocean, where sharp gradients in O_2 concentration occur, are regions of the most active N_2O production and consumption.

1.3.7 Fauna

Marine biomass (including both plants and animals) is estimated to contain roughly 6 Gt C, which is about two orders of magnitude less than the terrestrial equivalent (Bar-On, Phillips and Milo 2018). Consumption, and ulterior incorporation of carbon into faunal bodies is balanced roughly evenly by excretion of carbon products through respiration and defecation (Howard et al., 2017). Thus, faunal biomass is not a significant means of long-term carbon storage. The excretion of fecal pellets after ingestion of suspended particulate matter can lead to faster sinking speeds and thereby facilitating increased sedimentation, though fecal matter is often at least partially consumed and respired or excreted again by pelagic microbial communities before it can reach the seafloor (Kiørboe 2001, Turner 2002). Seabed fauna are thought to have large and additional effects on carbon transport to and within the seafloor through particle mixing and bio-irrigation (seawater flushing of sediments), which fall under the umbrella term “bioturbation” (Kristensen, Penha-Lopes et al. 2012). Seabed fauna may transport carbon in organic particles to and within the sediment matrix, increasing burial to sediment layers below their burrows through physical mixing, though this can also lead to carbon loss due to increased oxidation and thus decomposition (Song, Santos et al. 2022). Burrowing fauna promote fluid and particle exchanges between the sediment-water interface and sedimentary depth of their burrows, which increase the ratio of oxic : anoxic sediments, stimulating microbial organic matter re-mineralization, as well as the binding of organic carbon to deposited Fe (III). The latter, however, promotes the long-term stability of sequestered organic matter within sediments (Longman, Faust et al. 2022). The effects of sediment fauna on carbon fluxes are thus complex, contributing both to carbon sequestration to, and loss from, sediments. Net effects should thus be measured (e.g. Queirós, Stephens et al. 2019), not inferred.

1.4 Anthropogenic pressures on marine carbon stores and fluxes

Managing carbon flows (and those of other greenhouse gases across seascapes also requires consideration of key pressures on the ecological processes underpinning them. These pressures are summarized below:

1.4.1 Climate change

Through increasing greenhouse gas emissions from anthropogenic sources since the industrial revolution, humans have caused incontrovertible changes to the global climate. Particularly impactful consequences of the changing climate on the marine environment are changes in water temperature, pH, and dissolved oxygen concentration, while changes in ocean productivity and circulation also affect carbon cycling (Masson-Delmotte, Zhai et al. 2021). This report highlights the aspects that are known to impact seascape fluxes relevant to the reviewed potential new carbon project activity types.

1.4.2 Seabed disturbance

Seabed contact gears (e.g., as used for trawling and aggregate extraction) generally increase sediment resuspension and thereby inhibit long-term sedimentation and sequestration (Masson-Delmotte, Zhai et al. 2021). Meanwhile, undisturbed deep areas bordering disturbed areas, such as underwater canyons, may act as sinks for the material thus displaced (Martín, Puig et al. 2008). At present, most seabed disturbance (including trawling) is limited to continental shelf areas. As a result, much material kept in suspension is carried off-shelf to deeper areas, where it may be deposited. Oberle, Storlazzi and Hanebuth (2016) calculated that the amount of material moved from shelves to deeper areas through these processes roughly equals the flu*via*l input of sediment onto the shelves. However, carbon previously sequestered in sediments, in low oxygen environments, may in this way become exposed to the water column where oxygen is available for aerobic processes, leading to organic matter degradation and remineralization during transport (Aller 1982). Dredging of ports and shipping channels is similarly disruptive to the seabed and can lead to changes in local mean sedimentary grain size and organic carbon content (Wildish and Thomas 1985, Nayar, Miller et al. 2007). Sediment disruptions caused by aggregate extraction and mining operations can reduce sedimentary carbon stocks for decades after the events (Van Dalfsen, Essink et al. 2000, Stratmann, Lins et al. 2018). Deep sea mining is seen as potentially the most impactful, causing disturbance in deep ocean areas, typically seen as the long-term stores of oceanic carbon.

1.4.3 Nutrient enrichment of coastal waters

Many marine habitats are susceptible to increased nutrient concentrations, particularly nitrogen and phosphorus compounds. Coastal regions are particularly vulnerable due to their proximity to terrestrial and flu*via*l sources of anthropogenic nutrient inputs. Nutrient loads from agricultural run-off have increased drastically since the introduction of synthetic fertilizers, while legislative restrictions, such as the Water Framework Directive (European Commission, 2000) implemented in European waters, have ameliorated the situation.

Low levels of nutrient enrichment can be beneficial to primary productivity (fixing inorganic carbon in seawater into organic compounds and living biomass), and thus promote carbon uptake e.g., in seagrass (up to 300%, Armitage and Fourqurean 2016), macroalgae (Kraufvelin, Lindholm et al. 2010), saltmarshes and mangroves (Palacios, Trevathan-Tackett et al. 2021). However, associated soil carbon contents may be affected adversely, leaving open the question of the sustainability of this increased level of biotically stored carbon and its potential long-term sequestration (Armitage and Fourqurean 2016, Palacios, Trevathan-Tackett et al. 2021). Nutrient enrichment within the water column can lead to increased carbon sedimentation through increased levels of zooplanktonic activity (Olsen, Andersen et al. 2007). In extreme cases, enrichment can lead to a state of eutrophication in which excessive nutrient concentrations and abundances of organic matter reach levels that can have lasting impacts on the affected ecosystem (Nixon and Fulweiler 2009). Harmful algae blooms, in some cases fueled by increased nutrient availability from anthropogenic sources, can be lethal to fish and shellfish communities, affecting human health (Hallegraeff 2003, Gowen, Tett et al. 2012).

1.4.4 Wind sector interactions

Offshore wind farm structures host epiphytes, epifauna and associated communities that would not otherwise inhabit the area in question and contribute to increased suspended particulate matter concentrations in the water surrounding them (Baeye and Fettweis 2015). This process has been hypothesized to stimulate local transfer of suspended organic carbon from the water column to nearby sediments *via* feeding and subsequent fecal production by filter feeders attached to pilons (Degraer, Carey et al. 2020). All things considered, this alters sedimentation rates not only in the areas immediately surrounding wind farms but also in the surrounding areas through displacement of depositable material from tidal and other hydrodynamic interference. In addition, added turbulence and vorticity created by the wind farm structures increases resuspension potential and thereby decreases the likelihood of long-term sedimentation and/or sequestration (Grashorn and Stanev 2016).

2 Potential blue carbon project activities beyond wetlands

Here we summarize a scientific review of evidence for new potential blue carbon projects based on the seascape wide management of carbon fluxes, beyond wetlands – the core of this report. An overview of reviewed evidence can be found at the end of section 3, in Table 1.

2.1 Wild seaweed conservation

2.1.1 *Wild seaweeds and marine ecosystem benefits*

Seaweeds serve an important functional role as ecosystem engineers that provide habitat, increase habitat complexity, and thus promote species co-existence and biodiversity (Steneck, Graham et al. 2002). Their physical and chemical effects on ecosystems further include wave attenuation and reduced coastal erosion, amelioration of thermal stress for under-canopy species, and providing potential local buffering of ocean acidification, all of which provide benefits to local species and biodiversity (Bulleri, Eriksson et al. 2018, Wahl, Schneider Covachã et al. 2018). They further provide trophic subsidy to local and distal communities (Krumhansl and Scheibling 2012, Kopp, Lefebvre et al. 2015, Queirós, Stephens et al. 2019), as well as supporting commercial and non-commercial species (Bennett, Wernberg et al. 2015, Blamey and Bolton 2018, Bekkby, Torstensen et al. 2023). These effects are seen as key evidence supporting a recent push to conserve these ecosystems (Filbee-Dexter, Wernberg et al. 2022).

2.1.2 *The effect of wild seaweeds on oceanic carbon*

The contribution of seaweed to carbon fixation in the ocean is thought to be substantial, though the location of final sinks sites for seaweed carbon remains elusive (Figure 2). Seaweed-dominated habitats are thought to be some of the most productive vegetated marine habitats (Smith 1981, Pessarrodona, Assis et al. 2022), occurring in narrow fringes in rocky coastal waters. There is still considerable uncertainty about their true extents, but recent modeling estimates about 6.06–7.22 million km^2, dominated by red algae (Duarte, Gattuso et al. 2022). Their global net primary productivity (NPP) also remains uncertain, with estimates varying from 0.03-2.9 Pg C yr^{-1} (De Vooys 1979, Duarte 2017). The most recent global review suggests this value may be 1.32 Pg C yr^{-1} and dominated by brown algae such as kelp (Duarte, Gattuso et al. 2022). NPP rates vary over several orders of magnitude among seaweed habitat groups, from 0.2 g C m^{-2} yr^{-1} for floating *Sargassum sp* rafts, to (highly productive) 536 g C m^{-2} yr^{-1} in Sargassum and kelp forests (Pessarrodona, Assis et al. 2022).

Because seaweed occur on shallow rocky shorelines and reefs (down to 90 m in oligotrophic waters in Madeira, Braga-Henriques, Buhl-Mortensen et al. 2022) where there is limited potential for storage of particulate or dissolved organic carbon, up to 80% of the organic carbon fixed by macroalgal beds is thought to be seasonally exported into the coastal and open ocean (Krumhansl and Scheibling 2012). Very few field studies have however verified seaweed carbon contribution to oceanic carbon stores employing carbon flux measurements. Recent work suggests that their mean contribution to particulate sedimentary organic carbon stores in receiving surrounding open ocean sediments may be 8.75 g C $m^{-2}yr^{-1}$, although no measurements exist from deep ocean sediments. This estimate is substantially lower than mean carbon sequestration rates measured in saltmarsh, seagrass, and mangrove sediments

(Queirós, Stephens et al. 2019). Yet, seaweeds have a much broader global extent than wetlands, with a current biome estimated at 6.06–7.22 million km^2 (Duarte, Gattuso et al. 2022). Thus, seaweed potentially contributes to oceanic particulate organic carbon stores more broadly, with seaweed eDNA found globally across the water column (Ortega, Geraldi et al. 2019).

The contribution of macroalgae to wetland carbon sequestration rates is also potentially underestimated, and only recent technological advances in stable isotope data analyses, particle tracking modeling and environmental DNA (eDNA) analysis have allowed for initial identification of seaweed material in wetland sediments and preliminary contributions to carbon pools (Ortega, Geraldi and Duarte 2020, Arina, Hidayah et al. 2023, Queirós, Tait et al. 2023). The associated sequestration rates in wetland sediments remain, however, unquantified. The open question also remains about how much particulate seaweed organic carbon finds its way into the deep ocean, below 1,000 meters, where it may remain locked away from degradation for thousands of years, and thus satisfy the permanence requirements in future blue carbon policy (Krause-Jensen and Duarte 2016, Krause-Jensen, Lavery et al. 2018). This potential flux remains to be quantified, with several research groups working on this question at present. Indeed, due to large remaining uncertainty in present seaweed NPP modeling estimates, lack of systematic measurements of the contribution to seaweed to ecosystem-level carbon cycling (Dolliver and O'Connor 2022, Hurd, Law et al. 2022), and a scarcity of field data estimating contribution to ocean carbon stores, seaweed is not typically represented in global carbon modeling studies as part of the particulate organic ocean carbon pool. Clearly there is a lack of consensus regarding their potential contribution to net oceanic uptake of CO_2 emissions.

The contribution of seaweed to the oceanic dissolved organic carbon (DOC) pool may be substantial (Wada and Hama 2013) but is poorly understood. DOC constitutes ~70% of the ocean's organic carbon (Santos, Burdige et al. 2021) and maybe 30-50% of the ocean's photosynthesis products (Carlson, Ducklow and Michaels 1994). A large fraction of this DOC may not be readily consumed, and instead be exported to the deep ocean where it may reside for more than 1,000 years (Carlson, Ducklow and Michaels 1994, Santos, Burdige et al. 2021). About 23-50% of carbon fixed by seaweed during photosynthesis may be lost to the environment as exuded DOC, but substantial variation exists between species and conditions (Newell and Lucas 1981, Paine, Schmid et al. 2021). A fraction of that exuded DOC is also not readily bioavailable due to its chemical composition, and thus likely has high residence times in the water column, which is of interest to blue carbon policy as a form of carbon storage. Our understanding of seaweed DOC production was recently reviewed by Paine, Schmid et al. (2021), who highlighted its relevance to ocean carbon cycling and storage as well as existing uncertainty.

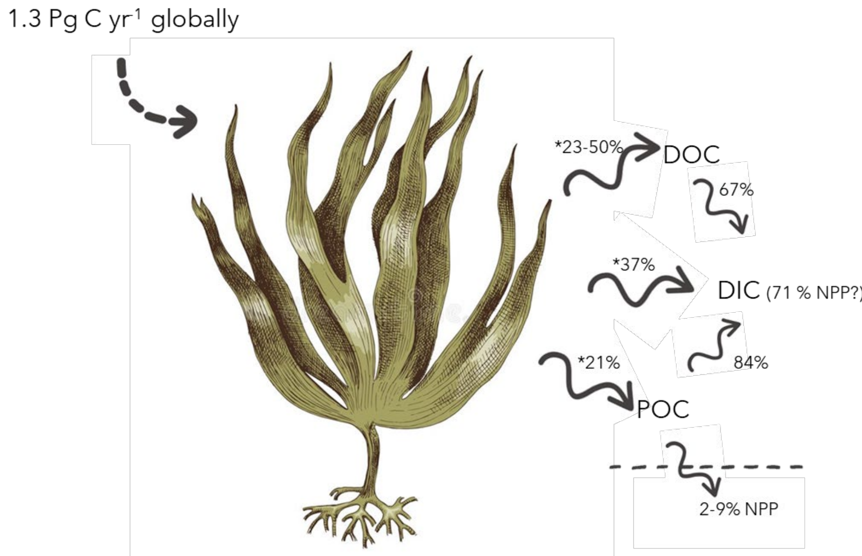

Figure 2: Types of carbon fluxes linked to seaweed. The total sum of % associated to fluxes does not add up to 100% as these reflect our best knowledge heretofore, based on different studies. Key sources include Duarte, Gattuso et al. (2022), Paine, Schmid et al. (2021), Queirós, Stephens et al. (2019),Krause-Jensen and Duarte (2016), Hardison, Canuel et al. (2010). Kelp diagram by Dreamstime.com.

A recent work proposed that seaweed contributions to the oceanic dissolved inorganic carbon (DIC) pool should also be considered (Perkins, Santos et al. 2022). DIC exists in the ocean in three chemical forms, two of which (carbonate and bicarbonate) are accounted for as alkalinity (in addition to a few other compounds) and these are less likely to return to the atmosphere than the third DIC species, aqueous CO_2 (Song, Wang et al. 2020). Whether DIC becomes part of the alkalinity pool or not is largely a reflection of the ocean's carbonate system state and is thus sensitive to greenhouse gas emissions (Doney, Balch et al. 2009).

A fraction of exported Particulate and Dissolved Organic Carbon (POC and DOC) that originated from seaweed will be re-mineralized into DIC by consumers (bacteria, fauna), and some of this will enter the long-term alkalinity pool (Santos, Burdige et al. 2021). Alkalinity production may thus be a key aspect of seaweed organic matter degradation. This has not been fully considered under a blue carbon lens but has the potential to be an important part of any blue carbon accounting, given the longevity of alkalinity in the deep ocean (Santos, Burdige et al. 2021) and the potential size of re-mineralized exported seaweed organic matter fluxes.

2.1.3 Other greenhouse gases

Our understanding of the effects of wild seaweed on other greenhouse gases is far from complete. Nitrous oxide (N_2O, defined in section 1.3.6) production from microbial grazing degradation of seaweed (Law, Rees and Owens 1993, Albert, Bruhn and Ambus 2013) has been observed in both natural and bloom conditions. This pathway partially explains why estuarine and coastal areas can serve as sources of N_2O (Bange 2006). At the same time, N_2O production related to seaweed may be limited by a local reduction of available ammonia and nitrate and the production of oxygen, all of which limit the microbial pathways that lead to nitrous oxide production. This limitation is expected to be especially true under farming conditions (in contrast to natural conditions), when biomass is removed from surface of seawater with every production cycle, limiting degradation *in situ* (Zhang, Boderskov et al. 2022). The nutrient balance of coastal waters can be a determinant of this effect. Eutrophic conditions create optimum conditions for bloom formation, which in turn lead to boom-bust cycles in biomass that support microbial degradation and the release of N_2O.

Anoxic sediments near seaweed ecosystems tend to be sources of methane due to methanogenesis (Roth, Broman et al. 2023). In the systems studied by Roth, Broman et al. (2023), methane emissions from seaweed and mixed vegetation ecosystems peaked in the summer and autumn, partially negating the biologically driven CO_2 uptake from these environments from the perspective of equivalent greenhouse gas emission. More work is needed to understand when and why seaweed systems emit CH_4, ensuring these are accounted for in any seaweed-based carbon project.

2.1.4 Identified issues

Several key matters have recently been highlighted as requiring consideration in the calculation of the efficiency of seaweed restoration and conservation as *via*ble blue carbon activities under standards that require detailed monitoring, reporting and verification. Matters that may reduce the efficiency of seaweed ecosystems to mitigate climate change mitigation include:

a) the amenability of POC and DOC export flux estimates to effective verification, leading to uncertainty in estimates of the carbon dioxide removal potential of activities (Gallagher, Shelamoff and Layton 2022);

b) potential relocation of nutrients from other photosynthetic organisms to enhanced seaweed biomass with the potential to affect overall ecosystem carbon cycling (Bach, Tamsitt et al. 2021);

c) the balance between potential calcification by seaweed epibiont communities (a CO_2-producing process) relative to carbon fixation by seaweed *via* photosynthesis (Bach, Tamsitt et al. 2021); and

d) potential production of potent GHG, N_2O and methane (Albert, Bruhn and Ambus 2013, Roth, Broman et al. 2023).

Additional matters that may increase the efficiency of these systems as climate change mitigation systems include:

a) the potential effects of albedo increase *via* enhanced rafting biomass and its ultimate effect of global radiative forcing and the greenhouse effect (Bach, Tamsitt et al. 2021);

b) prolonged periods of potential *via*bility of exported fragments (Frontier, de Bettignies et al. 2021).

Existing estimates for these effects are reputable if sparse in real life systems but may require consideration during potential future implementation of wild seaweed-based carbon projects. Many current seaweed habitats are particularly sensitive to climate change, as recently reviewed by Smale (2020). However, that sensitivity is highly variable across the world, with most assessed kelp habitat, e.g., in Europe, being presently in fair condition (or data deficient, Gubbay, Sanders et al. 2016), and seaweed habitats are potentially expanding in the Arctic and Antarctic (Bringloe, Wilkinson et al. 2022, Deregibus, Campana et al. 2023). Clearly, more globally distributed measurements are needed to allow us a better grasp of the contributions of seaweed DOC, POC and alkalinity fluxes to oceanic carbon stores and their potential role in the mitigation of climate change.

2.1.5 Future direction

Despite existing uncertainty in understanding, it is useful to consider the blue carbon value in investing in the conservation of seaweed due to its high biodiversity, economic and social value and given its broad global distribution, productivity rates and wide distribution of its detritus across the global water column. This is well aligned with an environmental push to limit and offset impacts on natural ecosystems (e.g., EU Habitats Directive; UN Ocean Decade for Restauration). From this perspective, benthic habitats are thought to be particularly sensitive. Identified challenges requiring further research to support the development of seaweed conservation as a *via*ble blue carbon activity include:

a) *Further development of the scientific basis underpinning project success*. Despite a long tradition, especially in Japan, to date almost 80% of seaweed restoration and afforestation efforts globally have been done in areas <1 ha in size and primarily driven by academics (>60%). Only three studies have worked with areas > 100 ha. The only currently known afforestation project in an area >100 ha failed, and so success has only been observed in small projects, focused on restoration of existing habitat, as reviewed by Eger, Marzinelli et al. (2022). That study concluded that this is still a research area in development and that known rates of success seemed to be improved by:

i) proximity of projects to natural populations to improve direct recruitment into the project area, or to other co-occurring species which may facilitate recruitment;

ii) ability to control grazer populations and competitors;

iii) the absence of environmental disturbance (such as storms);

iv) the striking of partnerships with governments and other actors to improve social license and limit impacts of other activities within project area.

The effectiveness of future projects at a larger scale will likely require a deeper integration of environmental policies that protect any deep-sea habitats and the seabed elsewhere that serve as potential sinks for seaweed carbon from disturbance (Epstein, Middelburg et al. 2022, Levin, Alfaro-Lucas et al. 2023).

b) *Future-proofing of projects*. Lessons from recent projects affected by extreme weather events highlight that such activities will only be successful at time-scales relevant to blue carbon (>100 years) if the effects of climate change are appropriately factored in (Eger, Marzinelli et al. 2020, Wood, Marzinelli et al. 2021). This is because both seaweed and the processes that lead to the uptake of their carbon into oceanic stores are sensitive to climate change pressures (Ravaglioli, Bulleri et al. 2019, Smale 2020). Recent developments in species distribution modelling for seaweed species are a step forward in this direction, but the performance of exiting models (statistical-based) is still lagging in terms of informing large scale project investment with confidence. Our understanding of the effects of climate change pressures on oceanic processes leading to long term carbon sequestration linked to seaweed are in their infancy.

c). *Development of regulatory mechanisms*. The requirements for the inclusion of seaweed conservation as part of blue carbon policy have been identified and reviewed (Krause-Jensen, Lavery et al. 2018, Eger, Marzinelli et al. 2020). There is exceptional research interest in this topic at present. The scientific evidence to support the design of blue carbon policy to account for the blue carbon contribution of seaweed is thus probably only a few years away.

2.2 Farmed seaweeds

2.2.1 Farmed seaweed and marine and human ecosystems

Seaweed has been farmed successfully in Asia and Eastern Africa since at least the 1940s, representing 51.3% of the global aquaculture production in 2018. The majority of this production originates in Asia, predominantly in China (FAO 2020, Chopin and Tacon 2021). A few studies have shown that when done correctly, seaweed farming can have a positive effect on the biodiversity of the surrounding environment (Radulovich, Umanzor et al. 2015, Lubsch and Lansbergen 2021, Theuerkauf, Barrett et al. 2022). Because of their capacity to absorb CO_2 and nutrients, it has been estimated that seaweed farming could help ameliorate eutrophication, hypoxia, and ocean acidification on a local scale in as many as 77 countries (Froehlich, Afflerbach et al. 2019).

The potential to ameliorate ocean acidification is thought to vary greatly among species (Stepien, Pfister and Wootton 2016). Seaweed aquaculture is also being assessed as a strategy to help curb environmental impacts of fish aquaculture through integrated multitrophic aquaculture approaches (Knoop, Barrento et al. 2022).

Seaweed products have multitude applications, including the constituents of cosmetics, food for human consumption, medicine, fiber materials with a number of applications in the clothing and building industry, biopolymers with some applications as replacement for plastic, the production of natural gases used as fuel and bio-oil, and supplements in cattle feed (Froehlich, Afflerbach et al. 2019). Seaweed products can provide a route for avoided emissions when their use avoids the uptake of new, fossil fuel intensive products, such as plastic polymers. The contribution of seaweed products to avoided emissions is, however, outside of this review, requiring careful consideration of markets and industrial processes involved in post-processing of biomass. Notwithstanding, seaweed has a large market potential outside of Asia and Eastern Africa where mature markets already exist (Msuya, Buriyo et al. 2014, FAO 2020), with emerging markets in Europe and America. Seaweed farming is also an activity with exceptional social value in countries such as Tanzania and Indonesia, where it represents one of few valuable routes for economic independence for women (Msuya, Buriyo et al. 2014). For these and other reasons, there is currently a large demand to expand seaweed aquaculture and to explore its potential to help meet global sustainability targets including mitigating climate change (Duarte, Bruhn and Krause-Jensen 2022).

2.2.2 The effect of farmed seaweeds on oceanic carbon

As for all blue carbon activities, seaweed farming stands to bring potential benefits *via* direct CO_2 capture into the farmed biomass, which may be used in lower carbon supply chains. During growth and harvesting, a potentially significant fraction of this biomass (Zhang, Boderskov et al. 2022) is also lost as detritus and exudates and may end up long term sequestered (>100 years) as described in 2.1. A more recent suggestion has been the targeted sinking of farmed biomass (Bernardino, Smith et al. 2010, Gray, Bisonó León et al. 2021).

The 2018 global production of seaweed was 31.8 Mt (fresh weight, "FW", FAO 2020). Using an estimated carbon content of dry weight of seaweed of 24.8%; and a 10% conversion of fresh weight to dry weight; Duarte, Bruhn and Krause-Jensen (2022) then equated this global production to a total uptake of 0.79

Tg C yr^{-1}, corresponding to a maximum flux of 2.89 Tg CO_2 yr^{-1} , if all harvested biomass was used towards sequestration (using up an estimated area of 1,983 km^2). Based on the current growth rate of the global seaweed aquaculture sector (6.2% yr^{-1}), a maximum uptake of 5.5Tg CO_2 y^{-1} could be globally driven by seaweed aquaculture by 2050 if all produced biomass was used towards sequestration, requiring an estimated area of 15,700 km^2 (Duarte, Bruhn and Krause-Jensen 2022). The 6th Assessment report of the Intergovernmental Panel for Climate Change suggests that low climate warming scenarios would require the delivery of substantial carbon dioxide removal of approximately 10Gt CO_2 yr^{-1}(~272 Tg C yr^{-1}) by mid to late century (Masson-Delmotte, Zhai et al. 2021). Achieving 1% of this rate (0.1 Gt CO_2 yr^{-1}, or 2.7 Tg C yr^{-1}) *via* removal of CO_2 from the upper ocean through the deployment of seaweed aquaculture could thus likely be met through the sectoral growth rate from now to 2050 (with consequentially large space and nutrient needs). But this would be the case only if all harvested biomass is considered. However, the type of use of biomass post-harvest is the key determinant of whether seaweed biomass production results in a significant net sequestration of CO_2, considering that CO_2 is produced at every step of current industrial seaweed production and processing methods, with variation expected in production methods, distance to port, means of transport to processing facilities, etc. (Czyrnek-Delêtre, Rocca et al. 2017).

With respect to artificial sinking of farmed seaweed into deep waters (> 1000 m), the fraction of that organic carbon that would enter long-term stores is very uncertain at present. From very few available field studies assessing decomposition of kelp at depth >1000 m, rapid decomposition rates of sunken kelp by seabed communities (5% day^{-1}) have been found (Smith 1983, Bernardino, Smith et al. 2010)). This suggests that over time-scales relevant to climate change mitigation (>100 years), the deep water column (not deep sediments) would be the long-term reservoir for this sunk (respired) carbon (National Academies of Sciences Engineering and Medicine 2021). However, given the variability of seabed habitats, those rates are also likely to be variable. Where biomass injection would take place seems to be a key determinant of sequestration time, with modelling estimates suggesting that injection depths greater than 1000 m would be required for any long-term storage (>100 years), and with variability expected between ocean basins based on global circulation patterns (Siegel, DeVries et al. 2021). The lack of field data in this area and the need for technological and regulatory development are critical challenges to the effective and safe delivery of farmed biomass at these very large depths, at an industrial scale (Ricart, Krause-Jensen et al. 2022).

In addition to harvested biomass, a report by the National Academies of Sciences, Engineering, Medicine of the United States National Academies of Sciences Engineering and Medicine (2021) further estimated that roughly 20% of standing farmed biomass is lost as particulate detritus though natural processes or due to harvest inefficiencies. A real-life estimate based on *Sacharina latissima* production (one of the most widely farmed species globally) in Sungo Bay, China (Zhang, Fang et al. 2012) put that value at 61%. Based on the global seaweed production in 2018 (FAO 2020), a potential additional term of 0.16

– 0.49 Tg C yr^{-1} in particulates could thus have entered the coastal and open ocean from production sites (using the same carbon content and fresh weight to dry weight conversions as Duarte, Gattuso et al. 2022). Field evidence for the fate of this detritus is by and large sparse (Gao, Gao et al. 2021). Based on Krause-Jensen and Duarte (2016) and Queirós, Stephens et al. (2019), and assuming presently poorly constrained rates of 8-11% of sequestration for this detritus in the seafloor, the final additional term of sequestered carbon from global seaweed farming would be 0.01-0.06 Tg C yr^{-1}. However, this number remains to be validated, especially in real life settings. Furthermore, as reviewed in 2.1, seaweed contributions to CO_2 inventories also include the production of dissolved carbon forms, both organic (i.e., dissolved organic carbon, DOC) and inorganic, the latter making some contribution to the oceanic alkalinity pool. Estimates for the production of exudates from farmed seaweed are not known at present and may only be speculated upon from values estimated from natural seaweed beds (Wada and Hama 2013).

The National Academies of Sciences Engineering and Medicine (2021) also assessed estimates for most of these terms as potential contributions of global seaweed farming to global oceanic carbon inventories (injection of all harvested biomass at depth, detritus production, DOC production, but not alkalinity production). They suggest that delivering a 0.1 Gt CO_2 yr^{-1} rate of removal of CO_2 from the upper ocean through the deployment of seaweed aquaculture (1% yearly of global excess CO_2 emissions) would require the deployment of seaweed aquaculture at a potentially challenging scale. Specifically, it would need to occupy an area equivalent to a belt of 0.5km across the whole coastline of the United States, or 73,000 km^2. This estimate is more than four-fold the area requirement predicted for the whole global aquaculture sector by 2050 (15,700 km^2) at the current sectoral growth rate of 6.2% yr^{-1} (FAO 2020, Duarte, Bruhn et al. 2022). This number poses the question of scalability for seaweed aquaculture as a blue carbon activity without consideration for pressures on marine space, competition with other sectors, and nutrient demand. That area corresponds, however, to just about 1% of the current model-based estimate of the natural seaweed biome (Duarte, Gattuso et al. 2022), or less than 0.2% of the global potential seaweed aquaculture suitable habitat (48 M km^2), if it could be assumed that offshore seaweed farming would be widely available the near future (Froehlich, Afflerbach et al. 2019).

It is also important to note that the estimates by the National Academies of Sciences Engineering and Medicine (2021) differ from other estimates presented here by considering that the detrital pathway (loss of detritus from grown biomass due to natural processes or harvesting inefficiency) represents a loss term in C accounting, and that any detritus production would be remineralized in the upper ocean. That assertion implicitly discounts the view that, as for natural kelp, farmed kelp detrital pathways represent an important contribution to the oceanic particulate carbon pool and is, therefore, at odds with Zhang, Fang et al. (2012), (2016), and Gao, Gao et al. (2021). The National Academies of Sciences Engineering and Medicine (2021) estimate for areal requirements for the seaweed aquaculture sector to deliver a 1% yearly removal of global excess CO_2 emissions by 2050 is, thus, likely to be pessimistic. These discrepancies between studies highlight that fundamental questions remain to be answered around the carbon dioxide removal value of seaweed farming, especially regarding the contributions of detrital pathways and dissolved carbon inventories.

2.2.3 Other greenhouse gases

Several climate-active gases may be emitted through seaweed farming and are worth considering in the determination of the efficiency of the industry as a climate change mitigation strategy. There is no consensus about whether or not seaweed aquaculture promotes N_2O production, requiring further investigation. N_2O production has been observed in both bloom conditions in nature as well as in experiments growing green seaweed *Ulva lactuca* (aonori), which is also commercially grown (Albert, Bruhn and Ambus 2013). However, N_2O emissions have also been suggested to be limited by seaweed farming, through a local reduction of available ammonia and nitrate stocks and the production of oxygen, all of which limit the microbial pathways that lead to N_2O production (Zhang, Boderskov et al. 2022).

The production of CH_4 from seaweed aquaculture has not been reported to the best of our knowledge. However, given the fact that natural seaweed beds and mixed vegetation ecosystems can serve as sources of CH_4 (Roth, Broman et al. 2023), it follows that artisanal, nearshore, benthic seaweed aquaculture could also be an CH_4 source. More research is needed to determine drivers and magnitude of such fluxes.

Whilst seaweed ecosystems may lead to methane emissions, the use of the red alga *Asparagopsis nodosum* and of other seaweed as additives in cattle feed have been found to cause a significant reduction of CH_4 production by cattle in several (but not all) studies, and there may be potentially important variations in the effects of different species of seaweed (Lean, Golder et al. 2021). Indeed, depending on the type (red, green, or brown) and species of seaweed used, the amount of supplement given (ca. 0.1 to 5% of the food intake), and the types of cattle considered (beef meat or diary), the reduction in CH_4 emissions from cows due to the seaweed addition can vary from fairly small to over 90% (e.g. Machado et al, 2014; Kinley et al. 2015; Maia et al. 2016; Molina-Alcaide et al. 2017; Roque et al. 2019; Roque et al. 2021). Several important considerations are thus needed prior to a wide scale application of this practice: 1) whether there is enough farmed seaweed to supply additives to a significant number of cattle to influence global carbon budgets cattle; 2) whether a high seaweed diet has detrimental effects on cows and subsequently on humans; and 3) can the active ingredient in seaweed by sufficiently preserved when processed into cattle feed.

Vijn, Compart et al. (2020) estimated that it would take about half of the global production of farmed seaweed to supplement the feed of all the cows in the US alone, assuming that seaweed represents 1%

of the cows' dry food intake. Effectively supplying all beef cattle globally would thus require seaweed production to vastly exceed the current global crop, if other supply chains were to be preserved.

The active ingredient in seaweed reducing CH_4 production appears to be bromoform ($CHBr_3$), which is toxic at high concentrations. The US, for example, has regulations about the safety ingestion levels for $CHBr_3$. This compound also volatilizes into the air readily, such that during the drying and processing of seaweed, a fraction of $CHBr_3$ will likely be lost, potentially reducing the effectiveness of the feed. Widespread farming of seaweed, for supplement to cattle feed or otherwise, will almost certainly increase the emissions of $CHBr_3$ and other halogen-containing organic compounds (halocarbons). These short-lived reactive gases influence the cycling of ozone in the atmosphere. Indeed, halocarbons (especially $CHBr_3$) are responsible for significant (ca. 10%) destruction of ozone in the polar stratosphere, which has direct impacts on global warming (Yang, Abraham et al. 2014, Tegtmeier, Ziska et al. 2015, Fernandez, Kinnison et al. 2017).

Micro- and macroalgae in the marine environment are major sources of $CHBr_3$ globally (Ziska, Quack et al. 2013), especially in the tropical regions, which play a significant role in the stratospheric halogen burden and contribute to ozone loss. This is due to enhanced upward transport from the ocean in the presence of deep atmospheric convection, making it crucial to consider the impact of widespread seaweed farming on halocarbon formation and its effect on stratospheric ozone. It has been estimated that even very large increases in the current area of the globe assigned to seaweed farming (well above the current rate of sectorial increase) would cause a very small (1%) increased in the global production of halocarbons by 2050 (Duarte, Bruhn and Krause-Jensen 2022), but this assessment was based on the production by *A. nodosum* specifically. Whilst this species is also the most promising seaweed so far regarding CH_4 emissions reduction in cattle, several other studies exist that have assessed the production of halocarbons in (other) naturally occurring and farmed seaweed species. Those studies show that the amount of $CHBr_3$ produced varies greatly between species (e.g., ~2.5-14.5 pmol halocarbon per g seaweed FW per h, Leedham, Hughes et al. (2013)) therefore suggesting that a more comprehensive assessment is still needed.

Since the agricultural, forestry and land use sectors drive 24% of global greenhouse gas emissions (Bashmakov, Nilsson et al. 2022), and agriculture is thought to be a key driver of rising CH_4 emissions, the use of seaweed as a cattle feed additive presents a potentially important avenue for seaweed farming to contribute to climate change mitigation *via* avoided emissions. However, further research is required. Balancing the rate of halocarbon production (and any effects on stratospheric ozone) with the potential benefits of seaweed with regard to effects on local CO_2 and other GHG emissions, and those of associated production chains, will thus likely be necessary to fully evaluate the effect of any seaweed-based carbon project on global warming.

2.2.4 *Identified issues*

All the estimates presented here on the potential effect of seaweed farming on oceanic carbon are built on the assumption that all farmed biomass is delivered to the seabed at depths greater than 1000 m at industrial scales. This assumes the existence of technology that is presently under development, untested, not available at scale, and raising ethical concerns (Ricart, Krause-Jensen et al. 2022). This assumption also neglects that all global seaweed production is currently used for marketable applications, such as human food consumption (Buschmann, Camus et al. 2017), not injection of biomass at depth. Short of an overhaul of the entire global seaweed market, currently published values estimating the potential size of the sector required to have meaningful effects on global CO_2 inventories (and associated sectorial areal and nutrient requirements) therefore likely represent substantial underestimations.

In addition, the main premise of all calculations presented is first and foremost that seaweed farming has a positive net effect on atmospheric CO_2 uptake, justified through seaweed's known large NPP rates. However, based on field measurements, the footprint of seaweed farms on local CO_2 uptake has both been shown to be positive (Jiang et al. 2013) and negative (Sato et al 2022), depending on location and conditions. Some variation between studies may be attributable to the use of different methodologies to estimate seaweed carbon fluxes, as well as to uncertainty in the estimation of physiological parameters (Buschmann, Camus et al. 2017). How these differences affect the estimation of the effects of seaweed farming at scales relevant to climate change mitigation (>100 years) remains unknown, posing as a present source of uncertainty for functioning seaweed farming-based carbon projects. Recent works have suggested that the identification of a consensual methodology to calculate the effect of natural and farmed seaweed on net CO_2 fluxes is still required (Bach, Tamsitt et al. 2021, Gallagher, Shelamoff and Layton 2022, Hurd, Law et al. 2022). Methods to provide these measurements exist and could potentially be incorporated into blue carbon methodologies. In practice, trade-offs would likely need consideration regarding the certainty and confidence achievable with a given method, and the cost of that approach in a voluntary carbon market project context. This is thus a picture of a research field in development.

Poor regulation or buy-in for good environmental practices linked to seaweed farming growth can lead to impacts on adjacent ecosystems. Particular issues raised include the physical biogeochemical alteration of the surrounding ecosystem potentially leading to shading; physical abrasion; and nutrient uptake; all of which can impact adjacent habitats of conservation and or commercial value (Campbell et al. 2019, Eklöf et al. 2006, Theuerkauf et al. 2021). Such effects vary with operational scale and intensity, as well as setting. For instance, there are contrasting, substantial differences in methods and expected impacts between artisanal, on-shore operations (e.g., the Western Indian Ocean) and large-scale, off-shore industrial operations (Eggertsen and Halling 2021, Msuya, Bolton et al. 2022). Existing knowledge and methodologies may provide the ability to improve the design of farms to limit such impacts, as recently shown (Aldridge, Mooney et al. 2021, Eggertsen and Halling 2021), offering guidance on how to develop future infrastructure.

2.2.5 Future direction

The current evidence basis suggests that there is indeed potential in farmed seaweed carbon project development, with numbers related to (potential) habitat available and carbon uptake fluxes particularly notable. The state of the science also indicates that there are fundamental gaps in knowledge related to our present ability to determine, with sufficient confidence, the full range of processes leading to carbon sequestration linked to this activity.

The climate change emergency requires us to consider any of these activities showing potential, in a safe manner. While at least one blue carbon scheme *via* local seaweed farming already exists (Yokohama Bay) the implementation of seaweed farming as a *via*ble activity in the international voluntary carbon market will require adhesion to verification processes, which necessitates clarification on the scientific principles underlying the contribution of this activity to an enhancement of carbon sequestration. Below we propose key areas requiring further investigation to enable that development:

1. Particularly important is the current disagreement about whether seaweed farming sites indeed enhance local CO_2 uptake (given that other marine ecosystem components also fix carbon). More research is needed. Appropriate siting of farms in areas of high nutrients could be one possible solution to limit any local potential competition for CO_2 and nutrients and ideally address eutrophication – modelling studies can already be used to provide initial guidance on this at design stage.

2. All current estimates for seaweed farming scalability as a blue carbon activity assume that farmed biomass is entirely used for sequestration – all 31.8 Mt of global production, none of which is currently used in that way. This use of biomass would require that the sector grows well beyond its current trend, if the existing seaweed market needs were to be met. No solution for delivering sequestration has been proposed other than the suggestion of sinking this biomass to the seafloor at depths greater than 1000 m, with technology currently lacking efficiency and safety assessments. Heretofore, no peer-reviewed studies have demonstrated that this is a *via*ble solution leading to sequestration. No peer-reviewed studies have risk assessed this activity either, which theoretically involves the drawdown of vast quantities of nutrients from the ocean surface within the sunk biomass, posing large questions for ecosystem impacts. Regulatory stumbling blocks remain as showstoppers too, as reviewed by Ricart, Krause-Jensen et al. (2022). Before the sinking of biomass can be considered a way to enable seaweed farming to be a *via*ble blue carbon

solution, the scientific validity of this approach needs to be demonstrated, and safety checks done at meaningful scales.

3. Questions also remain about how to estimate the various additional contributions that relate net primary productivity of farmed seaweed to carbon sequestration at a level relevant to carbon sequestration (>100 years), including: the size and fate of the detrital pathway; the size and fate of dissolved carbon contributions. These stand to be substantial, but this understanding is not well developed.

4. The upscaling of seaweed farming as a blue carbon activity with sufficient effects on global emissions (as reviewed by the National Academies of Sciences Engineering and Medicine (2021) will have substantial requirements for space. Seaweed is currently farmed in coastal waters, managed by each nation through Marine Spatial Planning processes (Ehler and Douvere 2009). Several governments around the world are interested in investing in seaweed farming and have proposed solutions resolving space issues include co-location (i.e., sharing of space between seaweed farming and other sectors, such as wind, Dutch Government 2014) and offshore siting of farms (Kim, Stekoll and Yarish 2019). Some investment is now needed to support ongoing sectorial technological development, with pilot projects ongoing in the Netherlands and in Finland (NSF1; OX2).

5. Last, life-cycle analysis reveals GHG emissions production throughout the life cycle of seaweed aquaculture, and the net benefit of seaweed aquaculture on emissions requires careful quantification including post-harvest processes.

2.3 Seabed management

2.3.1 The impact of seabed management on oceanic carbon

Marine sediments have been found to have a carbon storage capacity almost twice as high as terrestrial sediments, containing an estimated 87 Gt of organic carbon in the topmost 5 cm alone (Atwood, Witt et al. 2020). This makes them highly interesting candidates for long-term carbon storage conservation.

The assessment of global marine sedimentary carbon storage potential is complex, as the carbon storage potential of sediments is highly dependent on their physical characteristics and those of carbon input processes, which vary across scales. For instance, biological activity can have both stabilizing and destabilizing effects on the seafloor, depending on the organisms present in the local communities (Rühl, Thompson et al. 2020). The physical and chemical characteristics of the sediment are driven by natural and anthropogenic factors, which play a leading role in determining the carbon storage potential of any given location. Only 0.12 - 0.35 Gt of the ~87 Gt of organic carbon, calculated to be present in the top sediment layer at a global scale, are estimated to get buried to deeper depths each year (range based on varying estimates, dependent on environments considered and parameters included in the equation; (Epstein, Middelburg et al. 2022). This estimate is largely based on data from shallow shelf-areas, 58 - 70 % of which consist of course, sandy sediments (Emery 1968). Conversely, sediments in deeper waters are typically much finer-grained and less prone to be affected by natural turbulence and disruption (see e.g., (Diesing, Thorsnes and Bjarnadóttir 2021). Generally speaking, areas with fine-grained, cohesive sediments (supported by high organic load), with low resuspension potential, exposed to low turbulence and little biological and anthropogenic interaction support long-term carbon storage. Because geophysical conditions on the seafloor surface are dependent on the biological, hydrographic and anthropogenic pressures present, these characteristics can then be used as indicators of whether the environment in question is generally depositional and can therefore be a potential carbon sink.

Although 49% of marine sedimentary carbon stock is found within 200 miles of the coast (Atwood, Witt et al. 2020), seabed management typically occurs in coastal regions, within national waters (though the regulation of the High Seas will likely come online soon). In coastal regions, anthropogenic interactions with the seabed are also most common, though human interest in and interaction with the deep sea is ever increasing, not the least in the search for climate change mitigation strategies (Levin, Alfaro-Lucas et al. 2023). Each country may manage its national waters, and by extension seafloors, individually, though joint regions are often managed collaboratively in spatial planning agreements or at least in close communication between neighbouring countries, due to the undeniable connectivity of marine habitats (e.g., within the North Sea). Mining activities for example, are managed globally by the International Seabed Authority (ISA).

Intense anthropogenic utilisation of coastal regions, from fishing and aquaculture to energy generation and shipping, necessitates rigorous legislation to coordinate these activities and reconcile them with the protection of benthic habitats, even outside of protected areas. Because of the various commercial interests and comparatively easy access, near-coastal and shelf environments are better studied than deep sea environments. While a global estimate specific to shelf sea carbon storage has unfortunately not yet been attempted, regional estimates exist, e.g., for the Northwest European Continental Shelf (Legge, Johnson et al. 2020). And current efforts to quantify the carbon lost from sedimentary storage through anthropogenic disturbances are unfortunately still largely based on numerical model, with poor outcomes due largely to a lack of validation data (Sala, Mayorga et al. 2021).

Numerous anthropogenic activities that can be especially impactful on seabed properties and associated carbon stores:

1. Bottom contact fishing gears, including trawling and dredging, are highly disruptive to the sediment matrix. The gear effectively ploughs through the sediment surface, fluidising it (Foden, Rogers and Jones 2010, Epstein, Middelburg et al. 2022). The displacement of the oxygenated surface layer changes the biochemical properties of the sediment, causing deeper oxygen penetration depths, disrupting mineralization and burial processes, and promoting particle

resuspension (Tiano, Witbaard et al. 2019). This is particularly impactful for two reasons. Firstly, repeated trawling or dredging of the same areas is common practice, with recovery times increasing with each pass (Eigaard, Bastardie et al. 2017). Estimates of recovery times vary between a few days to 6.4 years in continental shelf areas (Hiddink, Jennings et al. 2017, Bruns, Holler et al. 2020) and go up to several decades at depths below 1000 m (Jones 1992). In soft sediments, each trawl scar can be several meters wide and deep (Bruns, Holler et al. 2020). Secondly, the expansion of affected areas is increasing, including deeper waters, as improvements in gear open up previously inaccessible habitats (Kroodsma, Mayorga et al. 2018). The areas fished are thought to account for 86 % of global sub-tidally buried organic carbon, potentially storing ~360 Gt of it in just the uppermost metre (Atwood, Witt et al. 2020). Of note, recent literature suggests that early estimates of carbon emissions from bottom trawling (Sala, Mayorga et al. 2021) may have been overestimated (Hiddink, van de Velde et al. 2023). The exact impact of seabed disturbance on sedimentary carbon storage remains unquantified, though areas rich in organic carbon are particularly vulnerable (Black, Smeaton et al. 2022).

2. Dredging of ports and shipping channels (i.e., maintenance dredging) causes similar environmental responses to those of seabed contact fishing gear, though deeper sediment layers are likely to be affected. While total carbon within the sediment has been found to be lower after a dredging event, this is mainly due to a reduction in inorganic carbon as the amount of organic carbon can also increase as a result of a dredging event (Wildish and Thomas 1985, Nayar, Miller et al. 2007).

Another difference between fishing and maintenance dredging is that sediment from shipping channels and ports is not merely displaced at random but removed and dumped in different locations. This subsequent dumping of the dredge spoil distributes particulate and dissolved carbon that had previously been in the sediment throughout the water column, making it prone to biological interactions and lateral transport. Preferential resuspension and transport of finer sediment fractions can then lead to an environment with larger mean grain sizes and lower carbon content (Morton 1977, Nayar, Miller et al. 2007). A novel approach introduced by (Sugimura, Okada et al. 2022) suggests using the dredged sediment to create new blue carbon ecosystems such as tidal flats and seagrass meadows. There is however little evidence so far on how successfully such a repurposing of dredged sediment (often from navigational channels, rich in heavy metals linked with high ship traffic) could be executed. A potential reduction of dredging in shipping channels and ports is unlikely in the future due to high commercial and societal importance of shipping.

3. Sand and aggregate extraction alters the seafloor topography and can, unless the environment is naturally turbulent enough, cause lasting changes in benthic faunal communities and sediment properties (Van Dalfsen, Essink et al. 2000, Foden, Rogers and Jones 2009,

Uścinowicz, Jegliński et al. 2014). Recovery times depend on the extraction methods used and local environmental conditions.

4. Deep-sea mining for nodules has many of the same effects as dredging in that the sediment surface layer is disturbed or even removed, leading to a fundamental alteration of the environment and carbon storage potential. According to experimental data, an area affected by disturbance retained only 54% of the carbon stock compared to nearby unaffected areas, even after 26 years (Stratmann, Lins et al. 2018).

5. The establishment and presence of large structures anchored in or on the seabed, such as wind farms, pipelines and undersea cables, causes disruptions of the sediment matrix and short-term resuspension events. Partially submerged structures such as wind farms in particular can also lead to long-term increases in suspended matter within the water (Coates, Deschutter et al. 2014, Grashorn and Stanev 2016, Dannheim, Degraer et al. 2019).Changes to hydrodynamic regimes add turbulence and vorticity, which in turn increase the likelihood of resuspension events and prevents lasting sedimentation, affecting sedimentary carbon stores (Grashorn and Stanev 2016).

2.3.2 Other greenhouse gases

Organic carbon beneath the oxygenated surface sediment layer can be microbially reduced to CH_4. This CH_4 can escape to the atmosphere *via* two general pathways of comparable magnitude globally: diffusive exchange and ebullition in the form of bubbles (Weber, Wiseman and Kock 2019). In both cases, most of the CH_4 escaped from the sediment is microbially oxidized within the water column (e.g., to CO_2) before reaching the surface. This is why shallow near shore environments tend to have the largest CH_4 fluxes on a per area basis.

Under certain conditions on the oceanic continental shelf, sedimentary CH_4 takes on the ice-like form of methane hydrates, or clathrates (Wallmann, Pinero et al. 2012), which when perturbed (e.g., by large temperature/pressure changes) may lead to sudden outgassing (Ruppel and Kessler 2017). While estimates vary, the global reservoir of oceanic sedimentary methane hydrates is thought to be substantial (many hundreds of gigatonnes C, Milkov 2005). However, the contribution of oceanic methane hydrates to atmospheric methane is very small and generally already included in the CH_4 emission estimate from the open ocean.

2.3.3 Identified issues

While the vertical downward transport of particulate and dissolved carbon through various means is also known as 'export', it should not be assumed that the seafloor is generally considered a carbon sink. There is current uncertainty about the extent of seabed areas estimated to be 'depositional' that can in effect be defined as long-term carbon sinks. Carbonaceous particulate and dissolved materials may be reintroduced into the water in various ways rather than being deposited, buried, mineralized and stored in the sediment. While the theoretically ideal conditions for sedimentary carbon storage can be deduced from studies and surveys, as described above, locating suitable areas offering such conditions can be difficult. Information on many of the driving factors as well as the make-up of the sediment itself is not yet available for most of the seafloor. Further study is needed to characterize larger areas of the seabed and to facilitate the selection and of areas dedicated to carbon storage, and the enforcement of their conservation. It is likely that at least some of the areas deemed suitable for sedimentary carbon storage are subject to multiple competing interests. Outside of national Exclusive Economic Zones, enforcing the exclusive use of selected areas as carbon storage reservoirs is problematic as the question of authority needs to be settled first in each instance. The capacity to protect such areas is therefore limited.

2.3.4 Future direction

Current scientific evidence suggests that it is possible to choose areas of the seabed that are likely to have high carbon storage potentials for protection. Whilst this can be estimated *via* modelling of near seabed conditions, quantifying this potential exactly remains problematic, especially in deep areas. Soft sediment deep-sea environments that are not yet affected by anthropogenic disturbances such as dredging and mining could be particularly promising options, and areas below 1000 m are generally assumed to be able to serve as long-term sinks (Williamson and Gattuso 2022).

The implementation of protected areas to prevent sediment disturbances could be a promising means of natural carbon retention in marine systems (Roberts, O'Leary et al. 2017), and the new High Seas Treaty may provide mechanisms to support such implementation. There are two main prerequisites that need to be met, to facilitate large-scale implementation of this approach. Firstly, depositional areas with large carbon storage potentials need to be identified. To this end, further extensive seabed survey is still needed to investigate, characterize, and chart the seabed. Secondly, enforcement will be needed to ensure compliance with newly protected seabed areas.

2.4 Shellfish farming, conservation and restoration

2.4.1 The impact of shellfish on carbon storage and fluxes

The natural incorporation of dissolved carbon into animals' shells and carapaces suggests that for a potentially substantial amount of time, a net negative carbon flux may be created (carbon from seawater into shell). Considering only shell material, the resulting carbon capture rates have been compared to those of forestry (Jansen and van den Bogaart 2020). However, the calcium carbonate deposition during shell and carapace formation is a net producer of CO_2 (Wang, Ge et al. 2016, Morris and Humphreys 2019). Furthermore, the potential dissolution of calcium carbonate, under unfolding ocean acidification, is likely to cause a decline in calcification rates between 10 and 25 % (Gazeau, Quiblier et al. 2007). In addition, each organism releases CO_2 and other carbon products throughout their lifetime through e.g., respiration and (pseudo-) feces production. Determining the resulting net carbon flux associated with such calcifying organisms is thus a complicated task, and existing estimates are dependent on which fluxes are included in the equations (Jansen and van den Bogaart 2020). Also, while (Tang, Zhang and Fang 2011) for example estimate an uptake of 3.79 ± 0.37 Mt C yr^{-1} and subsequent removal of 1.20 ± 0.11 Mt C yr^{-1} through shellfish harvesting (net amount not including soft tissue interactions), the long-term fate of the resulting 'waste' shell material is unknown, challenging the claim of 'removal'. Most of the waste material from shellfish food production is commonly either stored in landfills, left on land, or returned to sea without sustainable disposal or storage strategies, leading to various negative impacts on the environment and surrounding human populations (Shumway 2011, Silva, Mesquita-Guimarães et al. 2019).

Current alternative uses of calcium carbonates from shell material include pharmaceutical and medical applications (e.g., (Westbroek and Marin 1998, Chen, Jiang et al. 2016), food supplements or preservatives (Cho and Jeong 2018), animal feed (e.g. (Hamilton, Fairfull and Gowe 1985, Guinotte and Nys 1991), fertilizers and for water quality management (Huh and Ahn 2017, Lee, Kang et al. 2021), etc. (Bonnard, Boury and Parrot 2019). In many of these applications, the processing of the shells leads to increased bioavailability of the carbon fixed within, thus preventing long-term storage.

The promotion of long-lasting shell-material based products (e.g., building materials; (Silva, Mesquita-Guimarães et al. 2019, Águila-Almanza, Hernández-Cocoletzi et al. 2022), in which the calcium carbonates remain fixed for decades to centuries, would ensure prolonged storage of the carbon, and could therefore be encouraged.

In addition to the carbon directly incorporated into the animals' bodies and shells, the enhancement of benthopelagic coupling and carbon drawdown through their filter feeding must be considered. (Lee, Davies et al. 2020) measured active sediment deposition rates of 1.6 mg organic and 0.9 mg inorganic carbon per oyster per day. Therefore, shellfish beds can also be active contributors to sedimentary carbon reservoirs, though effects vary between species. However, this effect can vary across shellfish. For instance, some types of shellfish, termed 'biodiffusers', destabilize the seabed matrix, decreasing cohesion and organic content load of sediment (Montserrat, Van Colen et al. 2009). However, other species can create structures that bind sediment together and reduce near seabed turbulence (e.g., mussels and oysters). The stabilized sediment surface conditions and active vertical flux created by these shellfish beds can improve carbon retention. Environmental conditions will further matter. For instance, net carbon flux in and around natural oyster reefs can depend on depth. Experimental work showed that intertidal reefs exhibited net efflux of CO_2 (~7 t C ha^{-1} yr^{-1}) while subtidal reefs absorbed more than they emitted (circa −1.0 t C ha^{-1} yr^{-1}; (Fodrie, Rodriguez et al. 2017). Shellfish reefs bordering on the edges of salt marsh habitats can also have net CO_2 absorption (circa -1.3 t C ha^{-1} yr^{-1}) due to the high levels of organic carbon enrichment in surrounding sediments (Fodrie, Rodriguez et al. 2017).

2.4.1 Shellfish conservation

Efforts to conserve and restore shellfish habitats have increased in recent years, particularly in Europe (Lee, Davies et al. 2020). This drive is seen as a necessary response to combat 65 - 85 % loss of these habitats over the last century (Beck, Brumbaugh et al. 2011, Zu Ermgassen, Spalding et al. 2012). Monitoring of shellfish reefs for conservation purposes can be done through manual surveys and habitat mapping, remote sensing, or even with the use of unmanned aerial photography and subsequent detection using specialized machine learning approaches (Ridge, Gray et al. 2020).

Major threats to naturally occurring shellfish, that can be limited through direct management, are demersal fisheries (Halpern, Selkoe et al. 2007) and habitat loss (Airoldi, Connell and Beck 2009), and the design of MPAs and other OECMs can be major tools towards their management (e.g., (Sweeting and Polunin 2005, Nielsen, Nielsen et al. 2021). Habitat loss can be further countered *via* the introduction of artificial hard substrates, made most frequently from concrete (a carbon intensive substrate; (Fabi, Spagnolo et al. 2011) but also from other materials or mixed substrates such as in the case of the introduction of retired infrastructure (e.g., oil platforms) as artificial reef bases. Encrusting mollusks have been found to preferentially settle on vertical surfaces over horizontal ones, and on artificial substrates over natural ones (Spagnolo, Cuicchi et al. 2014). In some cases, such as in areas with established marine wind farms, both protection from fishing and habitat provision can be achieved in tandem: the submerged wind park structures can provide a substrate for encrusting species such as mussels and oysters (Degraer, Carey et al. 2020), while fishing in these areas is also restricted in some instances.

2.4.2 Shellfish aquaculture

The artificial cultivation of marine fish, crustaceans and shellfish has been practiced since at least 500 BC. As of 2020, 58.8 % of global marine aquaculture production has been in the form of mollusks (percentage by live weight, (Carranza and Zu Ermgassen 2020) and although aquaculture is practiced globally, China is the biggest producer of cultured shellfish with a market share of roughly 75 % (Fao 2022). Based on projections from 1961 to 2017, aquaculture is the currently fastest growing meat industry (mean growth rate of 3.1 % yr^{-1}; (Azra, Okomoda et al. 2021, Fao 2022). Future projections predict further growth in the coming years, following growing demand (Tacon 2020).

Despite some questionable practices (Sievers, Fitridge et al. 2017), shellfish aquaculture remains as one of the most sustainable aquaculture practices. There are a number of so-called Restorative Shellfish Maricultures (RSM) that aim to preserve and grow natural habitats as well as harvest shellfish sustainably (Carranza and Zu Ermgassen 2020). While traditional approaches were based on easily accessible near-coast locations and traditional substrates, more recent developments have moved off-shore towards long-line deployments as vertical floating substrates (Stevens, Plew et al. 2008). The environmental side-effects of shellfish agriculture are varied. Excretion of fecal matter into the surrounding areas can lead to increased nutrient loads, which promotes primary production in periods during which nutrients would normally be limited, but usually not to the point of eutrophication (Tzankova 2004, Ren, Ross et al. 2010).

Compared to wild communities, in which stock size may vary naturally inter-annually due to natural stochasticity around spawning and recruitment, cultured communities may be more predictable and thus assessable in terms of carbon budgets. However, net effects on carbon cycling will be particularly dependent on harvesting methods (e.g., the seabed is disturbed) and whole organism net carbon flux assessment (since calcification is a net producer of CO_2, see 2.4), and the fate of shell material post-harvest.

2.4.3 Other greenhouse gases

Aquaculture was estimated to account for over 1% of the global anthropogenic N_2O emission (less than 1% of the total global N_2O emissions) in 2003, and this contribution was forecasted to rapidly increase (Hu, Lee et al. 2012, Tian, Xu et al. 2020). Shellfish farming accounts for roughly a third of this sectorial rate (Tian, Xu et al. 2020). N_2O production arises from bacteria associated with aquaculture *via* both denitrification and nitrification. Filter and deposit feeders appear to have the highest rate of N_2O production among common aquaculture species, and this production may be reduced by keeping the system under optimal conditions (e.g., pH, temperature, dissolved oxygen, feed quality, etc.). Shellfish farming has the potential to emit significant amounts of CH_4 as well (e.g., (Yang, Tang et al. 2022). Detailed GHG inventories are thus clearly needed to determine whether a specific aquaculture operation can be explored in a carbon project context, and more work is needed.

2.4.4 Identified issues

Ocean acidification is potentially a key threat to shellfish habitats and aquaculture operations in the medium and long term. Selective breeding may be used to counter-act some of the effects of ocean acidification in aquaculture (Fitzer, McGill et al. 2019), and in the case of shellfish hatcheries and land-based facilities, chemical modifications to flow through seawater can be made during times of low pH, as well as repositioning of cultures (Barton, Waldbusser et al. 2015). However, use of these strategies in the conservation and restoration of natural reef habitats is limited. As the carbon retention potential of shellfish reefs is unclear, and natural reefs tend to be sources rather than sinks of carbon (Fodrie, Rodriguez et al. 2017), the widespread pressure from ocean acidification further undermines the reliability of managing shellfish habitats in a carbon project context.

If the byproducts of shellfish aquaculture (dead shell material) are not disposed of in a way that inhibits their degradation and eventual release of free carbon into the environment, the aquaculture practice does not contribute to carbon removal on long time scales. Shells have been valorized as a useful byproduct of shellfish aquaculture for direct use and as an inspiration for hybrid biomineral development (Morris, Wang et al. 2016).

2.4.5 Future direction

While some carbon can be taken up and stored in shellfish reefs for tens to hundreds of years, there is no conclusive evidence at this stage that shellfish aquaculture is an effective method of carbon removal. Non-bed building and non-filter feeding species such as limpets and abalone, both of which are widely cultured globally (Mau and Jha 2018), do not contribute to the net-carbon draw-down described in oysters and the like. Therefore, future exploration of shellfish management in a carbon project context should likely focus on reef-building, filter-feeding species. There are studies of mixed maricultures with algae and shellfish (i.e., Integrated Multi Trophic Aquaculture, or IMTA), in which negative net carbon fluxes have been measured (e.g. (Zhang, Yang et al. 2022). However, such systems and their effects are still poorly understood at this stage.

Conversely, the conservation of existing reefs and potential restoration of those negatively affected by anthropogenic drivers such as demersal fishing will prevent the release and subsequent emission of large amounts of carbon currently stored within them. Both measures promise good conditions for carbon uptake and storage at minimal cost to maintain the status quo (either current or that of the recent past, in the case of restoration).

2.5 Ocean Iron Fertilization and related approaches

Open ocean fertilization aims to enhance the biological productivity of ocean ecosystems and hence the sequestration of carbon *via* the addition of nutrients. There are two general approaches: 1) the intentional addition of micro (iron) or macro (nitrogen (N), phosphorous (P)) nutrients to the surface ocean; and 2) enhanced upwelling of nutrient-rich deep water to the surface. These are described in detail below:

2.5.1 The effects of ocean fertilization on oceanic carbon

Of all the open ocean fertilization approaches considered, the addition of dissolved iron (i.e., Ocean Iron Fertilization, "OIF") appears to have one of the highest sequestration potentials (>1 Gton C/yr globally if fully implemented (or 10% of global emissions by 2050, Williamson et al. 2021). This idea was in part inspired by the observation of high iron levels during previous ice ages, when dissolved CO_2 and temperature in the ocean were much lower (Martin, Fitzwater and Gordon 1990, Khatiwala, Schmittner and Muglia 2019). To date, 13 scientific experiments have been performed *in situ* to investigate the effects of intentional OIF, each on a timescale of weeks and a spatial scale of tens of kms. Seven of these experiments took place in generally iron-depleted waters in the Southern Ocean (so called High nutrient (N, P), low chlorophyll regions). These studies observed clear drawdowns of CO_2 in water and increases in chlorophyl during the days to weeks following the addition of iron, albeit with substantial variability among the studies. What is much less known is the fate of carbon initially drawn down over longer timescales. Indeed, out of the 13 experiments, very few showed clear export of carbon to the deep ocean. Much (ca. 90%) of organic carbon stimulated by the iron addition likely returns to the atmosphere quickly, following biological respiration, with no evidence of sequestration demonstrated.

In practical terms, the relative iron requirement by phytoplankton is much lower than that of macronutrients including N and P. Thus, the addition of iron may be logistically more practical – only a very small fraction of the global iron production would be needed for large scale application of OIF (Williamson, Boyd et al. 2022). One potential co-benefit of OIF may be increased fish biomass but evidence of this is tangential at best – stemming from e.g., increased salmon stock following volcanic eruptions (which deposit natural iron to the surface ocean; Parsons and Whitney 2012). OIF has been regulated under the London Protocol since the late 2000s and no further public scientific experiments have been carried out over the last decade. Other approaches exist that share the aims as OIF. Our understanding of those approaches, and of their effects on oceanic carbon, remains very limited, and future work is required to assess their *via*bility and safety:

Macronutrient addition (Nitrogen (N), Phosphate (N)). The addition of these macronutrients shares many of the similar characteristics and criticisms as OIF, and thus will only be discussed briefly. Artificial macronutrient additions have been rare (limited to two P -only addition experiments). Theoretical estimates for global CO_2 removal capacity based on this activity are on the order of a few GT CO_2 / year (Williamson, Boyd et al. 2022). There is, however, extensive literature on the impact of naturally present nutrients on ocean biology and carbon uptake. It is thought that artificial N addition could reduce the amount of natural N_2 fixation by bacteria and denitrification at depth, resulting in negative feedback. Such 'stability' of the N cycle implies that P, rather than N, is the critical control of the biological pump on climate relevant timescales (>100 years). The only P addition experiments performed so far, however, showed no enhanced biological productivity, illustrating large uncertainty in this approach. An obvious drawback in macronutrient addition relative to OIF is the fact that phytoplankton requires orders of magnitude more N and P than iron. Thus, large and probably unsustainable amounts of N and P would be needed to meet globally relevant carbon removal rates. Finally, additions of these macronutrients, like OIF, are also regulated under the London Protocol.

Artificial upwelling: The deep ocean contains much higher concentrations of macro- and micronutrients and CO_2 than the surface ocean. This is largely a result of continuous sinking of organic matter from the surface and subsequent remineralization (breaking down of organic matter down to smaller components; Martin, Knauer et al. 1987). Upwelling (upward vertical transport) of deep, cold, nutrient-rich water occurs naturally in some regions of the ocean (e.g., near eastern boundaries of oceans and some islands as well as along the equator). This generally results in high biological productivity and sustains much of the world's fisheries, e.g., in Chile (Pauly and Christensen 1995). Artificial upwelling as a blue carbon activity has been proposed, through mechanically driven flows of large amounts of nutrient-rich deep seawater (e.g., *via* pipes), thereby inducing greater biological productivity and CO_2 drawdown. However, a main drawback of this approach is that deep water also contains high CO_2 concentration, negating much (or perhaps all) of enhanced biological drawdown of CO_2 following fertilization (Yool, Shepherd et al. 2009). Indeed, the few artificial upwelling experiments done to date have not demonstrated any significant carbon drawdown (Williamson, Boyd et al. 2022). Furthermore, similarly to OIF, there are also concerns about changes in the ecosystem, including potential stimulation of harmful algae blooms, deoxygenation in midwater, ocean acidification, and emissions of other greenhouse gases following artificial upwelling. There is also a thermal dimension in this proposed approach. Artificial upwelling of deep, cold water will initially cool the surface ocean (and atmosphere and land), which will be seen as beneficial. But doing so alters the natural temperature gradient between the surface and deep ocean, thereby potentially disrupting the natural thermohaline circulation (large scale circulation of ocean currents driven by density differences in water masses). A model study projected that large scale enhanced upwelling will result in an increase, rather than decrease, warming in the long run following the initial cooling (Kwiatkowski, Prange et al. 2015). In sum, pumping deep ocean water to the surface, the basis for Ocean Thermal Energy Conversion (OTEC), may be a *via*ble method for electricity generation on a local scale. But deploying this method on a large scale to draw down carbon is unlikely to succeed.

2.5.2 Identified issues

There are several concerns about the efficacy and environmental impacts of OIF and similar approaches:

1) 'Nutrient stealing' downstream: upon addition of sufficient iron, phytoplankton will likely grow until they become limited by another nutrient (e.g., N or P). This process removes N and P from the waters and possibly limit phytoplankton growth downstream that would otherwise have occurred naturally (Watson, Boyd et al. 2008);

2) Shifts in ecosystem (Williamson et al. 2012): most of the OIF experiments resulted in a change in the ecosystem composition, with fast-growing species (e.g., diatoms) often favored. Such ecosystem changes (e.g., increased harmful algal bloom) may be undesirable;

3) Increased acidification of ocean interior (IPCC AR6, Chapter 5): increased ocean uptake of CO_2 by enhanced biological production and sinking of this carbon to depth will result in faster ocean acidification at depth;

4) Depletion of midwater oxygen: increased sinking of organic matter will likely lead to greater respiration and consumption of oxygen;

5) As a result of deoxygenation at depth, it is possible that OIF will increase the emissions of other green gases, especially N_2O and to a less extent CH_4. These gases have much stronger greenhouse gas potential than CO_2 and could offset the climate benefit of carbon drawdown.

2.5.3 Future direction

Due in part to regulation by the London Protocol, some fundamental questions about OIF remain, including effectiveness as a method of carbon removal and environmental impact. A common criticism for OIF is the observation that only a small fraction of carbon initially drawn down gets sequestered into the deep ocean (De Baar, Boyd et al. 2005), while most of the carbon drawn down as a result of iron addition is recycled within the surface layer and respired to become CO_2 (Boyd, Jickells et al. 2007). However, our degree of understanding in this is low. Several aspects require further exploration to consider the upscaling of OIF as carbon projects and to evaluate their effectiveness.

It is possible that when iron addition occurs in regions of natural deep-water formation (e.g., near the Antarctic, where surface water cools, becomes denser, and sinks into the deep ocean), the efficiency of sequestration is higher. How quickly the surface water is transported to depth is a likely source of variability in the effectiveness of existing OIF experiments, especially comparing those conducted in the Southern Ocean vs. those in tropical waters. Further experimentation, along with improvements in ocean modeling and paleoceanographic studies, could provide some insights into this topic.

Additions of a different form of iron (e.g., iron salt or ligand-bound iron) could be more efficient than the typical addition of ferrous sulfate in the previous 13 experiments because more of this micronutrient would be recycled within the surface ocean instead of getting oxidized and sinking out to the deep ocean as particles (Oeste, de Richter et al. 2017). So far, our understanding in this regard is largely limited to laboratory measurements and theoretical arguments. Further insights may also be gained by studies of the impact of other sources of iron on ocean biogeochemistry (e.g., desert dust, coal ash, volcanic ash).

Lastly, given the dynamic nature of the ocean, small scale experiments (both in time and size) are not suitable for tracking the fate of carbon from fertilization to sequestration in the deep ocean. Further OIF experiments over longer temporal scales and larger spatial scales would be necessary to evaluate the climatic and environmental impact of OIF more comprehensively. As mentioned above, OIF is considered a 'geoengineering' approach and is regulated internationally under the London Protocol, which prevents large scale ocean pollution. Therefore, any further ambition to upscale OIF would necessitate substantial safeguarding requirements to meet regulatory approval within compliant countries. The London Protocol was amended in 2013 to theoretically allow researchers to apply for exceptions to conduct OIF experiments. A large scale, multi-year experiment has been planned by the Republic of Korea in the Southern Ocean (Yoon, Yoo et al. 2018), but the progress on this is so far unclear.

2.6 Alkalinity enhancement

Natural processes such as the dissolution of carbonate sediments and the degradation of organic matter can facilitate the uptake of atmospheric CO_2 through the creation of more alkaline seawater environments with higher uptake capacity. Artificial alkalinization of the ocean has thus been proposed as a helpful means to counter-act ocean acidification and accelerate local CO_2 uptake (Renforth and Henderson 2017). This can be achieved by industrial means, through the addition of large amounts of ground minerals, rich in calcium or magnesium cations, to the ocean surface and in coastal waters (e.g., as ground olivine, $Mg_{2(1-x)}Fe_{2x}SiO_4$, Meysman and Montserrat (2017)). Although capacity is seemingly well suited to upscaling, such geoengineering activities can be costly, cause important localized impacts, and be met with important regulatory barriers (Renforth and Henderson 2017). Alternative, biological means of producing alkalinity have captured the attention of the international community and are currently being explored as potential carbon dioxide removing strategies.

2.6.1 The effects of alkalinity enhancement on oceanic carbon

CO_2 exists in the ocean in three chemical forms, two of which (carbonate and bicarbonate ions, CO_3^{2-} and HCO_3^- , respectively) are accounted for as part of the oceanic alkalinity pool (which also includes other chemicals). Carbonate and bicarbonate ions are less likely to return to the atmosphere than the third species of CO_2 in the ocean, aqueous CO_2 (Song, Wang et al. 2020), potentially for >100 years (National Academies of Sciences Engineering and Medicine 2021). The alkalinity pool is a key part of the ocean carbonate system, the balance of which determines the net direction of processes regulating the production and uptake of CO_2 in seawater, and its CO_2 exchanges with the atmosphere. Because of its dependance on the ocean carbonate system, alkalinity enhancement produces the important effect of potentially also helping to counter ocean acidification (Taylor, Lichtschlag et al. 2015).

Alkalinity production is well known to occur naturally in marine systems through rock mineral weathering. This is a slow mineral dissolution process, through which alkalinity is produced as bicarbonate in aqueous form, allowing the ocean to store more CO_2 (Meysman and Montserrat 2017). Mineral dissolution may be accelerated by smaller particle sizes because of the proportionally larger reactive surface area. Several types of naturally occurring minerals have been tested with the purpose of delivering seawater alkalinity enhancement. Recently, focus has been drawn to fast weathering silicate minerals, especially olivine, due to its global availability and fast dissolution rate. Achieving a large reactive surface area requires milling minerals to fine particles in production sites (Renforth and Henderson 2017), and this industrial process leads to CO_2 emissions that need to be balanced out against the final alkalinity production rates achieved (Renforth 2012). Minerals can then be released in different sites. Open waters require smaller particles to lengthen the residence time of minerals within the (near surface) mixed layer and thus CO_2 uptake, but this leads to higher CO_2 emissions at processing thus reducing the effectiveness of the approach. Greater interest for application of larger sized particles has thus been shown, with a focus on areas where high bed shear stress (ocean shelf), wave action and tidal rolling (beaches) lead to seabed transport and further grinding of deposited particles (Meysman and Montserrat 2017). Montserrat et al. 2017 is one of the most established studies assessing olivine weathering experimentally. They used natural seawater with different compositions, and indicated consistent alkalinization, followed by CO_2 invasion from the atmosphere into the seawater. The efficiency of the process was not estimated due to difficulties in estimating olivine dissolution rates, saturation effects and secondary reactions. Their estimate of CO_2 sensitivity, specifying how much CO_2 is taken up from the atmosphere for each mole of alkalinity that is released from the seabed, is 0.84 ± 0.1 (mol of DIC mol^{-1} of Total Alkalinity)

Because of the industrial processes involved in mineral ocean alkalinity enhancement and possible impacts from mineral leachates upon release, focus on natural habitats supporting alkalinization as an alternative to mineral weathering has gathered interest. For instance, a not yet peer-reviewed modelling study (Fakhraee, Planavsky and Reinhard 2022) suggests that within reasonable environmental conditions for mangroves and seagrass, their restoration could significantly enhance alkalinity production in the sediment, thereby increasing seawater alkalinity and promoting the local uptake of CO_2 from the atmosphere. Specifically, at a potentially very high rate of 10t CO_2 ha^{-1} yr^{-1} (no estimate of CO_2 sensitivity given). This perspective has been verified, for instance in Red Sea mangroves, where alkalinity production was estimated to cause an uptake 1264.77 g CO_2 m^{-2} yr^{-1} (Saderne, Fusi et al. 2021). In seagrass habitats, carbonate chemistry has been found to fluctuate with the diel cycle. However, Chou, Fan et al. (2021) found that in a semi-enclosed lagoon in China, where particulate carbon export is limited and seawater residence time is high, high pH and low pCO_2 are observed across the day, accompanied by seabed alkalinity production and carbonate dissolution (a CO_2 uptake process). They still note, however, that alkalinity at the site is lower than in surrounding waters, possibly due to the local dissolution of carbonates (this study also did not provide an estimate of CO_2 sensitivity). Indeed, through a flux of produced O_2 to below ground rhizomes, seagrass is thought to stimulate oxidative mineralization of (their own) organic matter buried within the sediment (Santos, Burdige et al. 2021). This process produces CO_2, which increases in porewater, dissolving sedimentary carbonates and raising the bicarbonate ion concentration, which may then escape into the overlying water column, raising alkalinity (Santos, Burdige et al. 2021). Porewater flushing of alkalinity from seagrass meadows, mangrove and even saltmarsh may then contribute to a raised alkalinity pool in the ocean.

Macroalgae, in turn, do not occur in soft sediment beds, and so, unlike in wetlands, they do not drive seabed alkalinity production. However, macroalgae are known to have strong effects on seawater carbonate chemistry (Hirsh, Nickols et al. 2020). Young and Gobler (2018) suggested that the positive effects of green macroalga *Ulva sp*. on local bivalve calcification were produced as a result of increased alkalinity by the alga, resulting from nitrate uptake, although in this study alkalinity was estimated indirectly using carbonate chemistry system calculations and not direct measurement. Other studies have reported similar results without measuring alkalinity (Hamilton, Elliott et al. 2022). Indeed, laboratory experiments on seaweed *Ecklonia radiata* degradation have also observed alkalinity production, as a possible result of sulfate reduction and ammonification (Perkins, Santos et al. 2022). The mechanism described occurs in two phases, with highly reactive dissolved organic carbon forming first as a result of seaweed degradation, and some of this then becoming DIC in a second phase, with 33% of degraded carbon entering the oceanic alkalinity pool. The latter occurred as a result of sulfate reduction (Perkins, Santos et al. 2022).

2.6.2 Identified issues

Fakhraee, Planavsky and Reinhard (2022) explain their potentially high rates of CO_2 uptake modelled for mangrove and seagrass through the enhancement of organic matter deposition known to occur in such habitats. They propose this leads to subsequent shoaling of oxygen penetration depth within the sediment, promoting microbial anaerobic respiration in sediments (that is, microbial iron and sulfate reduction), exacerbating carbonate dissolution, a CO_2 uptake process. However, studies such as Van Dam et al (2021) demonstrate through comprehensive field data collection, that the balance of processes that compose the carbonate system across the sediment and water column is not yet well constrained, and that in many cases, the opposite mechanism to that described by Fakhraee, Planavsky and Reinhard (2022) may be observed. Indeed, Van Dam, Zeller et al. (2021) found that iron and sulfate reduction only accounted for 1% of local $_{CO2}$ uptake in tropical seagrass ecosystems, with local calcification explaining 95.8% of local CO_2 production, which was larger than net particulate organic carbon deposition by one order of magnitude, and was estimated overall at a net rate of 6.1 mol m^{-2} yr^{-1} (2.7t CO_2 $ha^{-1}yr^{-1}$). This pattern appeared in part to be driven or exacerbated by temperature. The Fakhraee, Planavsky and Reinhard (2022) model also ignores the contribution of bioturbation (the effects of the activity of burrowing animals on the structure of coastal and marine sediments and the composition of its pore-water), which is enhanced in areas of high organic carbon deposition such as mangrove and seagrass ecosystems (Sarker, Masud-Ul-Alam et al. 2021). The net effect of bioturbation on CO_2 uptake or production rates, through effects on the alkalinity and other carbon pools, will vary with the composition of local sediment communities and their particular ecology (Rao, Malkin et al. 2014, Bernardino, Sanders et al. 2020). An ongoing failure to reconcile modelled processes with the complexity of the drivers of the carbonate system within real-life seagrass and mangrove habitats thus suggests that their net effect on alkalinity production (and thus CO_2 uptake) remains uncertain.

Recent observational work (Stepien, Pfister and Wootton 2016) has also suggested that the effects of seaweed on alkalinity production vary greatly between (39) species, with many species causing total alkalinity reduction as part of their carbon concentrating mechanism, that is, a process of promoting their own CO_2 uptake, and thus, photosynthesis. This effect was found to decouple total alkalinity from DIC, which may invalidate total alkalinity estimation based on standard carbonate chemistry calculations (e.g., using CO_2SYS) in the presence of seaweed. This was shown, in those cases, to lead to a localized increase in seawater CO_2, with total alkalinity decreasing by as much as 1100 µmol/kg seawater, during 24-hour assays. That study shows that the effects of seaweed on ocean alkalinity may not be easily generalizable, and a careful choice of species being consequently required where carbon dioxide removal strategies are concerned. The findings of Stepien, Pfister and Wootton (2016) may also partly explain some of the discrepancy observed between field studies assessing the value of seaweed ecosystems as net CO_2 sinks, as explored in previous sections.

2.6.2 Future direction

The science underpinning biologically driven alkalinity enhancement is more recent and less certain than mineral weathering research (Oschlies, Stevenson et al. 2023). Contradictory evidence has emerged from different systems, and between field and modelling studies, suggesting that there is yet poor consensus on the potential usefulness of mangrove and seagrass systems to support alkalinity enhancement, related to uncertainty in biogeochemical pathways linked to the alteration of the carbonate system. Seemingly, differences in the setting of habitats appear to be important and identified differences between systems should form the basis for future studies seeking to provide clarity on where seagrass or mangrove habitats can be used to this end.

The science underpinning the use of seaweed for alkalinity enhancement is also immature. In this case, fundamental differences between species in the deployment (or absence) of carbon concentrating mechanisms may provide guidance on species selection (Stepien, Pfister and Wootton 2016) for future applications providing local alkalinity enhancement. Heretofore, greater knowledge of species-specific biogeochemical effects on the oceanic carbonate system are still required. As important pathways will be observed not only as seaweed grows (in natural or farmed habitat) but also as seaweed degrades (Perkins et al 2022), a full balance of processes affecting the carbonate system across the life cycle of populations, occurring in parallel in real-life settings, is still lacking and needs elucidating. This may be easier to constrain within seaweed farming settings when the life cycle of the canopy is more or less fully controlled. It is noteworthy that CCMs have also been hypothesized for seagrass (Miller and Kelley 2021), and so the mechanisms underpinning differences between field studies on seagrass may to some degree remain obscured by poor understanding. Indeed, as in seaweed with CCMs (Stepien, Pfister and Wootton 2016), decoupling of parts of the carbonate system has also been observed in dense seagrass habitats (Miller and Kelly 2021). Both lines of evidence thus suggest that: 1) CCMs can complicate

Table 1: Summary of evidence for new potential blue carbon projectivity types beyond wetlands. '4. Optimal conditions' inform about suggested project settings under which carbon removal may be most successful and limiting ecological impacts, based on the current scientific understanding of ecological processes underpinning it (1), any identified issues (2), and existing need for future research (3).

Activity	1. Current understanding of potential [uncertainty level]	2. Issues potentially affecting effectiveness	3. Future direction	4. Optimal conditions
Wild seaweed conservation, restoration, and afforestation	Large CO_2 fixation fluxes, exceeding rates by wetland plants. [low] Broad global biome, exceeding coastal wetlands. [moderate] Potentially substantial contribution to long-term oceanic carbon stores recognized: deep ocean sedimentary carbon stores; dissolved organic carbon pool; alkalinity pool. [high] Scientific principles underpinning success of projects [moderate].	Positive: increased Earth albedo and reduced radiative forcing; prolonged viability of released biomass during water column transit upon release. Negative: climate change at trailing (equatorial) edge of distributions; balance of contribution to carbon sequestration rates against CO_2 producing processes in seaweed ecosystem; uptake of nutrients otherwise used by other autotrophic ecosystem components; potential N_2O production. challenging verification of contribution to sequestration rates.	Refinement of scientific basis underpinning project success. Refinement of/agreement on field techniques allowing for effective assessment of carbon fluxes (and those related to other greenhouse gases) affected by seaweed ecosystems (project boundaries). Development of tools allowing for successful identification (in space and time) and quantification of long-term sinks for seaweed carbon. Demonstration of success rate of projects at large scale (>100 ha). Development of tools to improve confidence in projections that allow for (required) futureproofing of projects (e.g., improve species	High nutrient availability Well established connectivity between source populations and ocean interior (>1000m). Low probability of extreme weather events (storms, heat waves). Temperate ocean and high latitudes (lower magnitude of long-term warming over time; lower loss of carbon content during transport). Work with species from biogeographical distribution center to leading edge, as climate change unfolds. Proximity to wild populations. (restoration better than afforestation) Ability to control grazer and other pressures.

			distribution models). Development of partnerships to improve social license for, and success rates of projects.	
Seaweed farming	Large CO_2 fixation fluxes, exceeding rates of wetland plants. [low] Broad global biome, providing potential for sectoral growth. [moderate] Required offshore sector growth (i.e., farming beyond coastal waters) possible at scale. [high] Viable contribution to long-term oceanic carbon stores hypothesized: deep ocean sedimentary carbon stores and water column (particulates and dissolved carbon forms). [high] Market uptake of harvest determines final application and net effect on emissions. [moderate]	(In addition to matters identified in seaweed conservation section): Effectiveness of suggested solutions for long-term sequestration of harvested biomass unknown. Methodologies for verification of contribution to oceanic carbon stores not widely accepted. Production of other greenhouse gases during farming (halocarbons and N_2O) detected in some species and understudied in others. Competition for nutrients and light with other autotrophic species in farming habitat may limit net effect on ecosystem. Post-harvest processing of biomass leads to emissions.	Demonstrate viability of proposed solution for sequestration of carbon from harvested biomass. Optimize project design to maximize connectivity of (naturally) exported carbon into oceanic carbon stores and limit impacts on adjacent ecosystems. Market and supply chain development beyond Asia and Eastern Africa towards applications that lead to avoided emissions via lower emissions supply chains (i.e., the use of seaweed in any industrial, commercial or domestic process that reduces the carbon demand of that process). Testing of offshore viability, especially through co-location solutions. Develop regulatory frameworks to ensure safety and efficiency of proposed approaches.	Environments where nutrient limitation is not a concern, especially if addressing local eutrophication (coastal cf. offshore). Co-location designs may reduce space constraints and environmental impacts (e.g., integrated multitrophic aquaculture). Growth of species with lower rates of calcifying epibionts. Temperate ocean and higher latitudes to maximize exported carbon sequestration rates. Lower latitude focus on avoided emissions. Work with species from biogeographical distribution center to leading edge to promote NPP.

Seabed management	Carbon storage potentials and flux rates spatially variable, dependent on local physical, chemical, biological and anthropogenic influences. [high] Provision of high carbon storage potential, but on unknown time scales. [moderate] Seabed carbon budgets reasonably well-known in coastal and shelf areas, less well-known in deeper and off-shore environments. [moderate] Effects of seabed disturbance not fully understood. [moderate]	Numerous stakeholders competing over seabed use, with competing interests and pre-existing claims to areas of interest for carbon storage management purposes. Poor regulation. Small-scale spatial variability in carbon uptake and retention potential Limited knowledge of seafloor characteristics Unknown duration of carbon retention	Study of promising deep sea depositional areas, to assess baseline carbon storage potential and retention time frames. Establish quantitative assessment of anthropogenic sediment disturbance on carbon flux and storage potentials on large spatial scales, ideally growing empirical evidence base. Legal protection of areas of interest has to be established and enforced, or supported by recognized verification process.	Low hydrodynamic energy and biological and anthropogenic disturbance levels Reducing conditions/hypoxia High carbon input to seabed Few competing interested parties in the chosen areas
Shellfish farming, conservation, and restoration	Shellfish production promotes the fixation of free carbon into mineralized forms, and uptake of organic carbon into the seabed by actively inducing downward flus of suspended particulates. [moderate] Calcification produces CO_2. [low] Aquaculture industries are already growing and projected to grow further in the future.	Uncertainty in estimates of resulting net carbon fluxes, dependent on parameters included in the calculation and final fate of shell substrates and soft tissues. Lower pH environments through increased CO_2 uptake rates may hinder optimal conditions for shellfish farming and conservation of wild shellfish communities in the future.	Optimize processing and recycling of shell material after tissue harvest to promote long-term carbon fixation. Improve net carbon flux estimates through further study and inclusion of conflicting parameters in calculations. Preserve natural shellfish habitats to prevent release of already fixed carbon stores.	Utilization / processing of whole organisms in aquaculture applications. Integrated multitrophic aquaculture may reduce environmental impacts. Non-extreme water pH levels. Reefs placed within protected areas.

	[low]	Competing interests from demersal fisheries.		
Ocean Iron Fertilisation (OIF)	Large initial drawdown of CO_2 in surface water when deployed in high nutrient (N, P) low chlorophyll regions [moderate], but long-term sequestration rate of C probably much lower. [high] Overall potential of activity at removing carbon may be significant. [high]	Consumption of macronutrients that otherwise would have stimulated in biological production downstream; changes in ecosystem composition; depletion of midwater oxygen; increased ocean acidification; emissions of other greenhouse gases (N_2O, CH4)	Further OIF experiments on longer time and in large spatial scales are needed to: determine the fate of carbon that is initially drawn down due to fertilization as well as ecosystem impact. Form of iron added may also be optimized.	Carbon sequestration from OIF may be more efficient in regions of natural deep water formation. OIF currently regulated by London Protocol. Regulatory approval may be needed to perform further OIF experiments/activities.
Macronutrient (N, P) addition	N addition may result in reduced natural production of reduced nitrogen, resulting in a negative feedback; P addition theoretically and logistically superior to N addition, but has not been shown to drawdown carbon in field experiments. [high] Overall low potential [moderate]	Broadly the same issues as OIF.	Compared to OIF, macronutrient addition is theoretically and logistically less advantageous.	Macronutrient addition is currently regulated by the London Protocol. Regulatory approval may be needed to perform further experiments/activities.
Artificial upwelling	Enhanced upwelling of nutrient-rich, cold deep water stimulates biological carbon drawdown in surface waters, which is however negated by outgassing of CO_2 from these waters. [moderate]	Changes in ecosystem composition; depletion of midwater oxygen; increased ocean acidification; emissions of other greenhouse gases (N_2O, CH^4); upwelling will also initially cool the surface ocean,	Enhanced upwelling may be a viable method to generate electricity in certain regions but is unlikely to be an effective carbon drawdown strategy.	

	Overall low potential. [low]	but at a cost of likely disrupting the thermohaline circulation.		
Alkalinity enhancement	Mineral based alkalinity enhancement is thought to have high capacity for upscaling, based on a mature evidence basis. [moderate] High longevity of carbon sequestered into the oceanic carbon pool. [low] Some species of seagrass, mangrove and seaweed may also cause localized alkalinity production, so net effect of other types of projects may need to be assessed. [high]	Preparation of minerals for weathering is a CO_2 emitting industrial process, that needs to be factored in against the effectiveness of different deployment strategies. Carbon concentrating mechanisms in seaweed (and potentially in seagrass) vary between species and could cause alkalinity decrease in some cases. Measurements of carbonate system parameters in seagrass and seaweed habitats suggest standard calculations (and modelling based on these) is unreliable in such settings. Empirical data is required to verify potential. This may extend also to mangroves. Different pathways link seaweed to the carbonate system in living and degrading biomass. Full life-cycle estimates are yet to be produced.	Assessment of the potential production of Ni (toxic) in mineral-based alkalinization is still needed, especially when optimal effectiveness is expected in applications in coastal areas. A better understanding differences between species of seaweed and seagrass (and mangrove) on the carbonate system requires further field validation with carbonate chemistry parameter measurements (not estimation) due to potential uncoupling of processes.	Mineral-based alkalinization deployment where natural processes lead to seabed transport and further grinding of deposited particles: a) in areas of high bed shear stress (ocean shelf) b) where wave action and tidal rolling (beaches)

expected effects of these organisms on oceanic alkalinity; 2) standard carbonate system calculations may not be applicable in habitats formed by dense photosynthetic organisms, such as seagrass and seaweed, requiring local, direct measurement of involved parameters, when exploring alkalinity enhancement value; and 3) modelling studies based on standard carbonate system calculations may not provide an effective route to explore the potential upscaling value of such habitats toward alkalinity enhancement.

3 Discussion

This review aimed to provide a thorough assessment of the peer-reviewed literature (and a small number of additional sources) documenting potential greenhouse gas effects delivered by seascape carbon flux management beyond wetlands as blue carbon project activities. We innovated on the existing collection of reviews on this topic (Black, Smeaton et al. 2022) by focusing on carbon project developers and the scientific community as end users of this document. We provided expert weighting of the certainty/uncertainty in the existing evidence bases for different types of projects; identified potential issues that may require a discount in carbon uptake rates in project activities (as well as any other issues linked to climate active gases beyond CO_2); suggested where future research is needed; and identified conditions/settings under which potential future projects could be successful, based on the evidence reviewed (Table 1).

We highlighted that significant seascape blue carbon fluxes exist in nature that justified a reconsideration of what other types of projects could become blue carbon project activities, beyond wetlands. This review also emphasized that these fluxes did not fall under currently recognized methodologies, such as those linked to the biological carbon pump, seaweed habitats and the management of the seabed in the open ocean. The review further pointed out that these fluxes required the consideration of carbon donor and carbon sink habitats, which are not necessarily co-located (not just those where the full blue carbon cycle occurs within one habitat, such as mangrove, seagrass and saltmarsh). The scientific community has accepted this concept (Smale, Moore et al. 2018, Queirós, Stephens et al. 2019, Queirós, Tait et al. 2023).

The review proposed that fluxes involving carbon arriving at the managed habitat from elsewhere (i.e., allochthonous) would also be accounted for in carbon projects if they added to sequestration within the managed habitat and if such rates could not be possible without the carbon project. The review also stated that habitats providing such organic carbon fluxes elsewhere would also require different accounting. Under this lens, seascape carbon projects would include allochthonous carbon fluxes (which are not currently included) as well as autochthonous carbon fluxes (which are already accounted for, e.g., in mangrove and seagrass carbon projects), if evidence could be provided to demonstrate additional sequestration.

We found substantial differences in the size and maturity of evidence pertaining to the estimation of such fluxes within different types of potential, new carbon project activity types we reviewed (as summarized in Table 1). A key issue that emerged was how to set the boundaries of the managed ecosystem to determine which fluxes can or should be accounted for. This may require the development of substantially different methodologies than those currently in existence for mangrove and seagrass carbon projects. Some of the types of projects reviewed here suggest that life-cycle analysis of carbon fluxes may be needed, especially when carbon is exported to land (for example shellfish and seaweed farming). From a scientific viewpoint, verification of such new carbon projects under a seascape framework is also challenging. *Via*ble management of such fluxes may require the deployment of carbon tracing methods allowing for the identification of connectivity between source and sink habitats, at least at setup phase. Multiple biotracing techniques could be used in tandem, such as isotope, eDNA and biomarker analyses, and carbon flux measurements, some of which may still require technological development.

With marked differences in the settings for the various types of potential projects reviewed here, it is likely that such developments would take some time. However, the information contained in this review may provide some guidance on which types of projects may be ready for deployment in the near future, at least from a scientific point of view. Based on the evidence reviewed, we suggest that seaweed conservation and farming projects, and seabed management may be those for which carbon project methodologies can be more easily developed. This is because we are gaining a better understanding of the connectivity between source and sink habitats, supporting the possible development of projects within a seascape framework (Queirós, Tait et al. 2023). Nonetheless, some key gaps in evidence remain, which could be seen as current stumbling blocks requiring further research, including the need for further research on the possible production of climate active gases by some seaweed species, potential differences between seaweed species on effects on seawater carbonate chemistry; avoiding species that support dense communities of calcifying epibionts, defining the boundaries of the managed carbon source system, and providing verification methodologies where the baseline evidence for carbon fluxes is sparse (seabed management; seaweed carbon sinks). For such projects, which do not include deliberate sinking of seaweed, it is likely that sufficient evidence may exist in the very near future to justify carbon projects, with science and regulatory landscapes developing at fast pace, and limited safeguarding issues identified here that could be seen to be current showstoppers, at least to development. With some additional research, it is likely that these could become the first tier of next generation potential carbon project activities.

For the other types of potential projects we reviewed (shellfish farming, nature-based alkalinity enhancement and ocean fertilization), significant knowledge gaps persist, particularly in terms of safety and scientific demonstration of greenhouse gas benefits of potential project types. Based on the evidence we reviewed, we find that is its inadvisable for these project types to be considered as potential candidates for carbon project development in the immediate future, as substantial research is still required (see Table 1). Therefore, we do not anticipate that such projects will be easily implementable based on the current evidence.

With an urgent need to reduce greenhouse gas emissions and mitigate climate change, some of the potential project types reviewed (seaweed conservation, farming and seabed management) may provide a way to maximize the global ocean's blue carbon capacity in the future. Such projects will not replace the immediate need to decarbonize the global economy, but they are smaller steps in the right direction, and may provide important access to income generation and the growth of the blue economy, in coastal communities and beyond.

This Issues Paper presented the peer-reviewed scientific literature review focused on current scientific knowledge, assessing uncertainty, limitations, and optimal conditions under which specific activities can lead to blue carbon opportunities with low uncertainty to their atmospheric benefits. In future, carbon project development criteria would need to be developed to apply to to blue carbon activities that were found in this report to have the greatest potential as future carbon project types. Such work would need to assess the conditions under which such activities could be developed into carbon project activities and receive carbon finance to support their development.

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