Potential impacts of climate interventions on marine ecosystems

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KEY POINTS

- Climate intervention research is expanding as current mitigation efforts to limit warming below crucial targets are falling short.
- Substantial knowledge gaps exist on the potential impacts of climate intervention strategies on marine ecological systems.
- We review the potential impacts of climate intervention on marine ecosystems, including biotic and abiotic factors.

ABSTRACT

Rising global temperatures pose significant risks to marine ecosystems, biodiversity and fisheries. Recent comprehensive assessments suggest that large-scale mitigation efforts to limit warming are falling short, and all feasible future climate projections, including those that represent optimistic emissions reductions, exceed the Paris Agreement's 1.5°C or 2º warming targets during this century. While avoiding further CO_2 emissions remains the most effective way to prevent environmental destabilization, interest is growing in climate interventions – deliberate, large-scale manipulations of the environment aimed at reducing global warming. These include carbon dioxide removal (CDR) to reduce atmospheric CO_2 concentrations over time, and solar radiation modification (SRM), which reflects sunlight to lower surface temperatures but does not address root CO_2 causes. The effects of these interventions on marine ecosystems, both direct and in combination with ongoing climate change, remain highly uncertain. Given the ocean's central role in regulating Earth's climate and supporting global food security, understanding these potential effects is crucial. This review provides an overview of proposed intervention methodologies for marine CDR and SRM and outlines the potential trade-offs and knowledge gaps associated with their impacts on marine ecosystems. Climate interventions have the potential to reduce warming-driven impacts, but could also alter marine food systems, biodiversity and ecosystem function. Effects will vary by pathway, scale, and regional context. Pathway-specific impact assessments are thus crucial to quantify trade-offs between plausible intervention scenarios as well as to identify their expected impacts on marine ecosystems in order to prioritize scaling efforts for low-risk pathways and avoid high-risk scenarios.

PLAIN LANGUAGE SUMMARY

A rise in global temperatures from 1.5°C - 2°C or above historical levels threatens marine life, ecosystems, biodiversity, and the sustainability of fisheries. Recent studies highlight that current efforts to keep warming within critical limits are insufficient, and even optimistic future climate scenarios predict that the 1.5°C threshold established by the Paris Agreement will be surpassed. In this context, climate intervention strategies are being explored as ways to potentially reduce the worsening effects of climate change and complement, not replace, decarbonization efforts. These strategies aim to either remove carbon dioxide from the atmosphere or reflect sunlight back into space to cool the Earth. While currently an understudied area of research, these methods could have profound impacts on the ocean, including changes to sea surface temperature and nutrient cycling, which in turn affect the abundance, distribution, and diversity of marine life as well as the human communities that rely on marine resources. This review synthesizes current research on the climate intervention strategies that are most likely to have direct impacts on the marine environment, emphasizing knowledge gaps as they relate to the potential impacts on marine ecosystems and the need for improved predictive models.

1. INTRODUCTION

The unprecedented pace of physical and chemical changes in the atmosphere, ocean and land resulting from anthropogenic climate change is driving widespread environmental disruption, both on land and at sea (Malhi et al., 2020; Poloczanska et al., 2016). Changes in ocean conditions are already altering many essential services provided by marine systems, including commercial and subsistence fisheries, affecting food security, livelihoods, and overall wellbeing (Allison & Bassett, 2015; Blanchard & Novaglio, 2024; Cheung et al., 2010; FAO, IFAD, UNICEF, WFP and WHO, 2018). Ambitious international targets for greenhouse gas emissions have been established to prevent dangerous impacts of a warmer climate (IPCC, 2023) and to achieve ocean sustainability (IPBES, 2019; UN DESA, 2022). Despite concerted international efforts and progress in some regions towards these targets, global outcomes remain uneven, and key goals have yet to be achieved in the absence of consistent enforcement and accountability mechanisms (Rogelj et al., 2016; United Nations Environment Programme, 2022). An overshoot of the aspirational 1.5°C warming threshold established by the Paris Agreements is imminent, as are a number of climate tipping points that would be irreversible on human timescales, such as permanent loss of land ice and the resulting sea level rise that would displace hundreds of millions of people, as well as loss of terrestrial and marine species and ecosystems (McMichael et al., 2020; Penn & Deutsch, 2022). Given the immediacy of this climate crisis, a portfolio of possible solutions beyond reducing greenhouse gas emissions is being explored to limit long-term temperature increases above 1.5°C (McGrath et al., 2025).

Future climate scenarios, such as the Shared Socioeconomic Pathways (SSPs), have emphasized that cutting emissions alone is insufficient to avoid crossing the aspirational threshold of 1.5°C warming, and negative emissions are needed to achieve a more stable climate, involving the removal of gigatons of carbon dioxide (CO_2) from the atmosphere by mid-century (Loomis et al., 2022; O'Neill et al., 2017; Rogelj et al., 2016). How this industrial-scale carbon dioxide removal (CDR) will be achieved is not yet clear. The ocean is an obvious reservoir, as it contains the majority of the carbon in the Earth's surface environment and continuously takes up carbon from the atmosphere and land (Smith et al., 2024). Thus, there is a growing push to explore ocean-based carbon dioxide removal (hereafter referred to as marine CDR or simply CDR) through a variety of mechanisms that could, in theory, accelerate the natural sequestration of anthropogenic carbon into the ocean (Figure 1; Smith et al., 2024). At the same time, there is a growing effort to explore technologies that could cool the planet by modifying the solar radiation balance, termed solar radiation modification (SRM). The original and current vision for SRM was as a temporary implementation to stabilize or lower global temperatures while emissions are reduced (Long & Shepherd, 2014). Here, we review what is known and what needs to be determined for understanding the impacts of these two types of climate interventions on marine ecosystems.

Climate intervention is broadly defined as the deliberate large-scale manipulation of the environment to offset anthropogenic global warming by either reflecting sunlight back into space (i.e., SRM) or reducing the levels of atmospheric greenhouse gases (i.e., CDR; National Academies of Sciences, Engineering, and Medicine, 2021, 2022; Shepherd, 2012). Consideration of climate interventions has been controversial due to perceived concerns that furthering the development of these techniques could lead to a sense of complacency, with the belief that intervention technology can single-handedly address the problem without emission reductions, or that the international cooperation for global scale climate modification is not likely (Biermann et al., 2022; Fankhauser et al., 2022; Robock, 2012; Rogelj, 2023). Serious scientific consideration is necessary, however, as some form of CDR is almost certain to be essential to meet global temperature targets (alongside large-scale emissions reduction efforts) and SRM is the only known potential option for rapidly reducing surface temperatures.

The ocean plays a pivotal role in the development of the subset of CDR strategies that leverage and enhance the natural biogeochemical processes that transfer CO_2 from the atmosphere into the ocean (Figure 1; Boyd et al., 2022; Renforth & Henderson, 2017). Broadly, these CDR strategies can be separated into biotic and abiotic approaches. Biotic methods, such as the cultivation of biomass, rely on photosynthesis to capture CO_2 and ultimately store it at depth as organic matter or dissolved inorganic carbon (DIC). Abiotic methods, on the other hand, alter the carbon chemistry of the surface ocean to enhance its ability to absorb CO_2 and increase ocean DIC (National Academies of Sciences, Engineering, and Medicine, 2022). Millions of dollars in venture capital funding have been directed towards start-ups promising to amplify the ocean's carbon sink, though none are currently operating at the scale necessary to sequester climatically relevant amounts of carbon (Smith et al., 2024). Despite this increasing momentum, all CDR methods that involve ocean uptake or storage could impact marine ecosystems, and these impacts must be better constrained.

Of the proposed SRM technologies, stratospheric aerosol injection (SAI), where light scattering particles are injected into the stratosphere, and marine cloud brightening (MCB), where clouds are seeded over the ocean surface, have received the most attention (Figure. 1; Kravitz et al., 2013; United Nations Environment Programme, 2023). For SAI in particular, there are very few doubts that, if deployed at scale, it would partially or completely offset global mean temperature increases (Tilmes et al., 2018; Visioni et al., 2021), although there are questions over many regional or seasonal impacts and potential other effects it might have (Kravitz & MacMartin, 2020; Robock, 2020). On the other hand, there are still doubts over the feasibility and efficacy of global scale MCB, together with potential regionally uneven impacts (Haywood et al., 2023; Hirasawa et al., 2023; Rasch et al., 2024). Perception of SRM among those familiar is highly diverse and contentious for reasons including ethics concerns, general mistrust of technocratic interventions, lack of international governance and regulatory frameworks, risk of termination shock (i.e., rapid warming if SRM was suddenly halted), as well as unknown long- and short-term side effects and high uncertainty (Biermann et al., 2022; Morton, 2015; Parker & Irvine, 2018; Usman et al., 2022; Zarnetske et al., 2021).

Understanding the potential ecological impacts of climate intervention strategies requires a broader understanding of the underlying processes shaping marine ecosystems. Climate-driven shifts in net primary productivity (NPP) can have profound and multifaceted impacts on the food web by influencing the availability and quality of resources, as well as through impacts to the marine carbon cycle. Marine primary producers (such as phytoplankton, algae, seagrasses) use photosynthesis, which requires light and nutrients, to produce an excess of organic carbon beyond what is needed to sustain them. This excess organic carbon (i.e., NPP) can be consumed by secondary producers like zooplankton, which in turn are consumed by larger animals, transferring energy up the food chain through successive trophic levels (Ryther, 1976). Perturbations to plankton community structure can affect the efficiency of trophic transfer, leading to increases or decreases in the number of larger animals a given amount of NPP can support (Ryther, 1976; Stock et al., 2014). Similarly, shifts in plankton community structure affect the efficiency of carbon sequestration by modifying the biological pump, the suite of processes by which inorganic carbon is converted to organic matter and then transported to the deep ocean (Kwiatkowski et al., 2023; Siegel et al., 2023). Determining how these interconnected processes will be impacted by deliberate, large-scale intervention remains a key research priority.

Due to the nonlinearity of physical, biogeochemical, and especially ecosystem processes, there is a high level of uncertainty in how marine ecosystems would respond to climate intervention scenarios, building on the uncertainty of how marine ecosystems respond to climate change (i.e., Heneghan et al., 2021; Mariani et al., 2020; Petrik et al., 2022; Tagliabue et al. 2021). The majority of SRM studies have focused on the atmospheric or climatic response to the intervention, without direct analysis of ecosystem impacts (e.g., Kravitz et al., 2014; Tilmes et al., 2016), with some inferring potential changes to ecosystems from proxy metrics, such as temperature or NPP (e.g., Heck et al., 2018; Proctor et al., 2018; Tjiputra et al., 2016). Similarly, some modeling studies for CDR have assessed impacts to marine ecosystems indirectly through shifts in plankton biomass (e.g., Kwiatkowski et al., 2015), oxygen levels (e.g., Keller et al., 2014), or measures of carbonate chemistry such as pH (e.g., Gonzalez, 2016; He & Tyka, 2023; Wang, 2023). Given the uncertainties surrounding the relative impacts of ocean acidification, deoxygenation, extreme temperatures, and shifts in ocean circulation patterns, substantial further research is required to determine if the overall net effect of climate interventions will be more or less beneficial for marine ecosystems than non-intervention scenarios. Few studies to date have evaluated the impacts of CDR or SRM on food webs and fisheries, research on key ecosystem drivers such as temperature and NPP remains limited (Zarnetske et al., 2021), and while some work has explored some combined interventions (e.g., Jürchott et al., 2024; Keller et al., 2014), this area remains significantly understudied. Therefore, the overall objective for this review is to examine the potential mechanisms by which climate intervention might impact marine ecosystems in an effort to guide further research, contribute to future model forcing scenarios, and begin to develop a process-based framework that can compare marine ecosystem impacts across a diverse suite of potential pathways.

Here, we 1) review projected impacts to marine ecosystems in future climate scenarios without intervention; 2) introduce marine CDR and SRM strategies and outline the current state of knowledge and gaps about impacts on marine ecosystems; 3) synthesize available literature working to quantify the impacts of climate intervention on marine ecosystems; 4) outline key processes critical to understanding climate intervention impacts that are not currently captured by climate and ecosystem models and discuss how they could be incorporated; and 5) emphasize the value of co-design between climate modelers and end users to improve the relevance and utility of impact assessments. This review is intended to support interdisciplinary communication between the marine ecological, CDR, and SRM communities and to help identify and prioritize research pathways for understanding marine impacts of climate intervention strategies. We emphasize that holistically assessing ocean, ecosystem, and fisheries-based impacts is necessary for effective climate intervention planning and implementation due to the relevance for global food security, biodiversity, and ecosystem services.

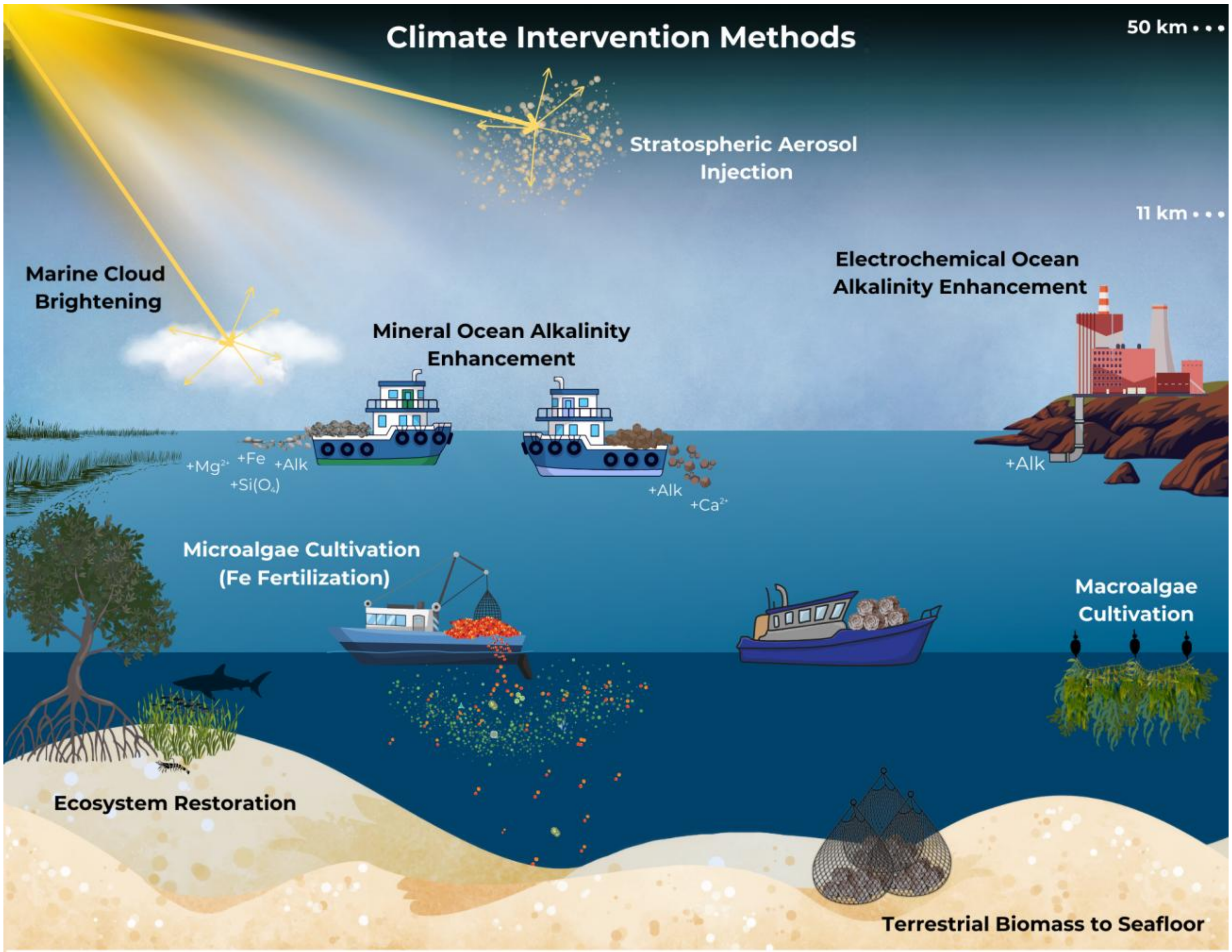


Figure 1. Illustration of the climate intervention methods reviewed in this paper to either increase the amount of sunlight reflected back into space (stratospheric aerosol injection and marine cloud brightening) or remove carbon dioxide from the atmosphere (ocean-based approaches for carbon dioxide removal).

2. PROJECTED IMPACTS TO MARINE ECOSYSTEMS WITHOUT INTERVENTION

Patterns of warming, deoxygenation, and acidification, driven largely by CO_2 emissions, are reshaping the global ocean (Barnett et al., 2005; Poloczanska et al., 2016). Observed shifts in the ocean's physical state, including alterations in salinity, density, and circulation (Doney, 2010; Henson et al., 2016), in turn affect nutrient transport to plankton, further deplete oxygen, and exacerbate acidification (Bopp et al., 2013; Doney et al., 2012; Silsbe et al., 2025; Van De Waal & Litchman, 2020). Without intervention, projected changes are poised to alter marine ecosystems (Kwiatkowski et al., 2020; Lotze et al., 2019; Tittensor et al., 2021). Earth system models (ESMs) are one of the primary tools available to represent the marine environment and carbon cycle under future scenarios (Petrik et al., 2022; Stock et al., 2011). Simulations from the Coupled Model Intercomparison Project (CMIP), a collaborative framework comprising many distinct Earth system models, have been used to examine changes to ecosystem drivers (e.g., temperature, pH, NPP) through the end of the century. Across both the CMIP5 (Bopp et al., 2013) and CMIP6 (Kwiatkowski et al., 2020) simulation ensembles, models agree on the direction and magnitude of surface temperature increases and pH decreases, both on average and regionally. Other ecosystem drivers have less certain projections.

For example, while the overall decline in upper-ocean oxygen and euphotic zone nitrate concentration has high model agreement across the CMIP5 and CMIP6 ensembles, regional patterns of change for both drivers remain uncertain, particularly in terms of scale and geographic distribution (Kwiatkowski et al., 2020). Oxygen and nitrate changes are understood to be largely due to warming-enhanced stratification, which inhibits vertical mixing and thus nutrient delivery and oxygen ventilation below the mixed layer, with modulation by temperature driven changes in remineralization rates. The projected decline in oxygen is expected to contribute to a compression of habitable zones for many marine fish species (Deutsch et al., 2015), among numerous biogeochemical and ecosystem impacts, especially when combined with other stressors (Gruber, 2011; Kim et al., 2023; Laffoley & Baxter, 2019; Steckbauer et al., 2020).

The future of net primary production, the base of the marine food web, is least certain. Models agree on increases of NPP in polar regions, driven by reduction in sea ice coverage and increased stratification (Kwiatkowski et al., 2020; Skyllas et al., 2019; Vancoppenolle et al., 2013). In other regions, on average, a future decrease in marine NPP is projected, consistent with satellite-based studies (Silsbe et al., 2025). However, the magnitude, spatial distribution, and sometimes direction of the simulated NPP trends have high disagreement (Nakamura & Oka, 2019; Kwiatkowski et al., 2020; Tagliabue et al., 2021; see section 5.1 for ESM uncertainty). Despite these uncertainties, modeling intercomparisons of marine consumers indicate that any future declines in phytoplankton biomass or productivity will lead to a greater degree of change in zooplankton and fishes, i.e., trophic amplification (Kwiatkowski et al., 2019; Lotze et al., 2019; Tittensor et al., 2021). Furthermore, NPP is an imperfect measure of energy at the base of the food chain, as the associated organic matter can be remineralized without being consumed by animals like zooplankton or fish (Dugdale & Goering, 1967). Thus, plankton biomasses and export production may be better metrics to assess ecosystem dynamics and carbon sequestration, but ESM projections of these variables are often more uncertain than NPP (Kwiatkowski et al., 2020; Laufkötter et al., 2016; Petrik et al., 2022).

Ocean acidification can be harmful for marine organisms that construct their shells or skeletons out of calcium carbonate ($CaCO_3$) minerals, which become more soluble in acidic environments (Hofmann et al., 2010; Raven et al., 2005). When the ocean absorbs excess CO_2 from the atmosphere, this CO_2 reacts with seawater in a series of temperature-sensitive chemical reactions, decreasing the potential hydrogen (pH) and the saturation states for the $CaCO_3$ minerals calcite and aragonite (see Section 4.4 for further discussion). Calcification then becomes much more energy-consuming, while existing $CaCO_3$ structures begin to weaken and could dissolve entirely. Calcifying primary producers (i.e., coccolithophores), as well as common zooplanktonic organisms (e.g., pteropods), are affected by pH and saturation state changes (Bednarsek et al., 2019; Krumhardt et al., 2017, 2019), as are coral, molluscs, and echinoderms (Doney et al., 2009). Multi-decadal declines in average surface ocean pH, as well as the saturation states of calcite and aragonite (Ω_{arag}) have already been observed at numerous locations globally, surpassing natural variability (Bates et al., 2014; Dore et al., 2009; Takahashi et al., 2014; Williamson & Turley, 2012). CMIP5 and CMIP6 models capture this historical trend and estimate that ocean pH could see an average decrease of up to 0.5 by 2100 while the aragonite saturation state could decrease by 0.9-1.4 depending on the scenario, resulting in potentially irreversible impacts on biogeochemical and ecological processes (Kwiatkowski et al., 2020; IPCC, 2023; Williamson & Turley, 2012). Coral reef net calcification and subsequent accretion beyond 2050 is not projected to continue at sustainable rates even under low emissions scenarios and could become negative under even moderate emissions scenarios (Cornwall et al., 2021). Numerous studies have examined aragonite undersaturation and its resulting decrease in coral reef calcification (Chan & Connolly, 2013; McMahon et al., 2013; Silverman et al., 2007). Previous literature suggests lower sensitivity to acidity fluctuations in more mobile marine species, although the cumulative impacts of warming and acidification are largely unknown and crucial to quantify (Baag & Mandal, 2022; Kroeker et al., 2013).

Disruptions to lower trophic levels as a result of a changing climate will likely result in shifts and alterations to the productivity of the marine food web and higher trophic levels (Heneghan et al., 2019; Munday et al., 2008). A comprehensive analysis combining multiple CMIP6 Earth system models and fisheries models as part of the Fisheries and Marine Ecosystem Model Intercomparison Project (FishMIP) found that, without the inclusion of fishing impacts, mean consumer biomass will decline ~19% by end of century under a high emissions scenario, while a low emissions, strong mitigation scenario will still result in a biomass decrease of 7% (Tittensor et al., 2021). Projected biomass changes reveal declines in equatorial regions due to nutrient loss and increases in the Arctic driven by reduced sea ice and extended productive seasons, highlighting a north-south gradient in climate change impacts (Tittensor et al., 2021). By combining estimates of fish stock status based on catch with a dynamic bioclimate envelope model, Cheung et al. (2022) projected biomass rebuilding rates of exploited species under different scenarios of future warming. Results indicate that, without substantial climate mitigation measures and robust fisheries management, the potential of biomass rebuilding to historically unexploited levels in many marine ecoregions will not be possible. Further, due to specific habitat requirements and limited geographical ranges, risk of extinction is disproportionately greater for endemic species, with one synthesis study estimating that 54% of marine endemic species assessed were at risk under numerous future warming scenarios (Manes et al., 2021).

The magnitude of potential loss to marine megafauna under future climate scenarios can be difficult to quantify as climate-induced changes often coincide with an uptick in anthropogenic activities (i.e., exploitation, gear entanglement, etc.) as well as shifts in prey distribution, thereby increasing uncertainty in predictions (Gulland et al., 2022; Simmonds & Eliott, 2009). The projected loss of marine mammal species richness and diversity across all emissions scenarios would disrupt ecosystem function and stability, with especially high species loss in the North Pacific and in higher emissions scenarios (Albouy et al., 2020). Additional impact or risk assessments specific to marine megafauna have linked behavioral impairments in sharks to ocean acidification (Rosa et al., 2017), changes in marine turtle dispersal patterns to intensifying storms (DuBois et al., 2020), and contractions in sea ice extent to breeding failure in penguins (Fretwell et al., 2023). At the same time, future warming is projected to hinder or even reverse the recovery of baleen whales to historical whaling activity (Tulloch et al., 2019). Such studies demonstrate not only the various interconnected components of a changing climate, they also highlight additional metrics that exist beyond a temperature target for parsing climate impacts at multiple spatiotemporal scales, metrics that ultimately define climate resilience. The current projections of marine ecosystem health in all emissions-driven future climate scenarios clearly indicate severe losses in biodiversity, function, and food security (Blanchard et al., 2017; Boyce et al., 2022) that increase with warming, motivating investigation of climate intervention.

3. CARBON DIOXIDE REMOVAL

Carbon dioxide removal (CDR) methods impact global atmospheric CO_2 concentrations directly and therefore aim to address the root cause of changes in global temperatures (McGrath et al., 2025). All of the CDR methods discussed here aim to redistribute carbon from the atmosphere into the ocean and keep it there as long as possible (Figure 1; Doney et al., 2024), but vary in their efficiency and durability (National Academies of Sciences, Engineering, and Medicine, 2021; Siegel et al., 2021, 2023). Biotic CDR techniques rely on photosynthesis to convert atmospheric CO_2 or surface ocean dissolved inorganic carbon (DIC) into biomass and then move that biomass to deeper water or sediments for storage. In contrast, abiotic CDR techniques, such as alkalinity enhancement, increase the capacity of the surface ocean to absorb atmospheric CO_2 by adding alkalinity and/or manipulating pH. The potential effects of these methods on marine ecosystems depend substantially on the details of the specific pathway proposed, how it is implemented, and the location of deployment.

3.1 Biotic methods for ocean-based CDR

Biotic CDR techniques aim to amplify an Earth system feedback that is believed to have cooled the climate during a period of high CO_2 in the geologic past by sequestering large quantities of organic carbon in marine sediments (Jarvis et al., 2011). Marine ecosystems may be impacted by biotic CDR techniques that cultivate marine biomass in the surface ocean as well as those that store biomass that originally grew on land. However, the mechanism and scale of impacts on marine ecosystems will depend on the proposed organic material type and placement site conditions. Biomass materials have a wide range of properties in ocean environments, which may be further modified by processing prior to placement (compression, extraction to remove valuable components, combination with minerals, or other treatment). The fate of biomass materials in the environment is also strongly sensitive to choices about containment and sinking methods (Raven et al., 2025).

3.1.1 Production and fate of biotic carbon

Similar to terrestrial afforestation, marine algae cultivation aims to convert CO_2 into organic carbon. Unlike terrestrial afforestation, however, the CO_2 for marine biomass growth comes from dissolved CO_2 in the surface ocean, which influences atmospheric CO_2 through the subsequent re-equilibration process. Marine biomass-based CDR strategies thus interact with marine ecosystems during cultivation as well as during breakdown and storage. Marine biomass sources can be roughly divided into those that rely on microalgae, such as nutrient fertilization or artificial upwelling for plankton growth, and those that rely on macroalgae, such as seaweed farming.

Microalgae cultivation involves fertilization of phytoplankton, increasing their biomass production (Figure 1). If microalgal biomass is exported below the mixed layer or into sediments, it could lead to net sequestration of carbon from the atmosphere (NASEM, 2022). The production of biomass in the surface ocean is limited by the availability of nutrients, largely nitrate, phosphate, or iron, depending on location. Up to 33% of the global surface ocean is primarily iron limited, with excess nitrogen and phosphorus available (Moore et al., 2001; Moore et al., 2013; Ustick et al., 2021). These regions have potential for large increases in productivity in response to a relatively small addition of iron. Iron fertilization allows phytoplankton to take up available nitrogen and phosphate, resulting in carbon uptake and possible export to the deep ocean, which can drive a depletion in surface ocean DIC that allows the surface ocean to absorb additional carbon dioxide from the atmosphere (De Baar et al., 2005; Martin, 1990; Martin et al., 1991). However, the export efficiency of organic matter from the surface to the deep ocean is highly uncertain and difficult to monitor, and the durability of sequestered carbon may be limited by the generally short return time of shallow waters to the surface if carbon is not exported well below the mixed layer (Buesseler et al., 2024; Doney et al., 2024; Jiang et al., 2024). Moreover, model simulations demonstrating the reduced durability and elevated leakage (i.e., the loss of stored carbon) of large-scale iron fertilization suggest it may not be as effective as alternative marine CDR approaches (Aumont & Bopp, 2006; Keller et al., 2014; Siegel et al., 2021; Tagliabue et al., 2023).

Macroalgae cultivation, or marine afforestation, refers to the farming or controlled growth of large marine algae and involves selecting suitable species, establishing cultivation sites, and managing the growth process (Figure 1; Ocean Visions and MBARI, 2022). Both *Sargassum* and kelp (*Macrocystis*), which are types of macroalgae (seaweed), have high growth rates and low nutrient requirements, making them relatively efficient sources of fixed carbon and good candidates for marine afforestation (Krause-Jensen & Duarte, 2016; Macreadie et al., 2021). For macroalgae, productivity is often macronutrient-limited (i.e., nitrate, phosphate) and would require different strategies from those of microalgae cultivation for large-scale fertilization (Williamson et al., 2022). Macroalgal carbon can be transferred to the deep sea via methods ranging from free sinking to farms that harvest the biomass, extract high-value products, and sink the residual (Elmer et al., 2023). In well-oxygenated deep ocean environments, most of the carbon in macroalgae biomass is respired over days to months (Filbee-Dexter et al., 2022; Pederson et al., 2021), releasing carbon as DIC that can be stored for decades to centuries or longer, depending on the depth and location of breakdown.

As a potential alternative to cultivating vast quantities of new marine biomass, terrestrial biomass sources such as waste products from agriculture and wood processing can also serve as inputs for CDR with marine storage, a process termed biomass sinking (Keil et al., 2010; Strand & Benford, 2009). These biomass sources rely on land-based nutrient systems that are relatively well understood and impacted at scale due to agriculture (e.g., Gregg & Izaurralde, 2010; Muth Jr et al., 2013), and thus they avoid the direct impacts to marine ecosystems from marine biomass cultivation. Strategies that rely on extant terrestrial infrastructure would also minimize disruption due to new infrastructure construction at sea. At the same time, increased demand for biomass materials could incentivize shifts in agricultural practices that could indirectly impact marine ecosystems through changes in nutrient or sediment runoff (Costello et al., 2009; Donner & Kucharik, 2008).

Many of the potential ecosystem impacts of biomass storage in the deep ocean depend on the degradation resistance ("recalcitrance") of the biomass used, in the context of the processes driving breakdown in the storage environment. Fresh biomass is composed of a spectrum of biomaterials ranging from highly labile (rapidly consumed) molecules like amino acids, sugars, and nucleic acids, which typically have lifespans of hours or less in the surface ocean, to recalcitrant biopolymers like lignin, which can persist in the environment for hundreds of years or longer (Hansell, 2013; Tegelaar et al., 1989). Breakdown rate and site circulation work together to influence the intensity of acute chemical change near the biomass storage site as well as the distribution of soluble breakdown products.

3.1.2 Ecosystem impacts of biotic CDR

The cultivation of macroalgae or microalgae at scale would represent a potentially massive perturbation of marine net primary productivity (NPP), with positive or negative impacts for food webs depending on scale and location (Oschlies et al., 2025). For example, augmented algal growth could drive shifts in local primary producer and zooplankton communities by shading the upper water column or by changing the availability of nutrients for phytoplankton, leading to food-web impacts and potentially impacting local fisheries (Gallo et al., 2025; Levin et al., 2023). Macroalgae can also act as natural fish aggregating devices by providing refuge from predators as well as substrates for other organisms to grow on, thereby aiding in their dispersal. Microalgae, particularly large diatom blooms, may also act as a potential substrate for colonization by microbes. Drifting macroalgae like *Sargassum* can travel vast distances, and even those macroalgae that grow attached to substrates can be extensively transported as fronds physically break off (Krause-Jensen & Duarte, 2016). By transporting "passenger" organisms living in seaweed offshore, macroalgal biomass may introduce species into new areas. Additionally, macroalgae can attract scavengers due to physical structure/substrate, food source and novel ecosystem, with complex effects on organism dispersal, genetic diversity of natural populations, and viral blooms (Alberto et al., 2010; Lyons et al., 2014).

Changes in the nutrient uptake in one region can have large impacts both locally and in downstream environments. For marine biomass cultivation, shifts in the relative utilization of different macro- and micro-nutrients (nitrate vs. phosphate vs. iron and other trace metals) by different types of primary producers can impact competition patterns among other types of organisms (Berger et al., 2023). Iron fertilization could further favor *Pseudonitzschia*, a diatom genus known to produce the harmful neurotoxin domoic acid (Silver et al., 2010). Changes in nutrient utilization patterns can also affect productivity in downstream locations due to "nutrient-robbing" (Berger et al., 2023). For example, much of the excess nutrients in the iron-limited Southern Ocean, a proposed location for large scale iron fertilization, fuel productivity in the subtropical gyres. Thus, CDR-induced nutrient shifts in the Southern Ocean might lead to less productivity and export in the subtropical gyres (Aumont & Bopp, 2006; Oschlies et al., 2010a; Sarmiento & Orr, 1991). Modeling studies have thus predicted a potential negative link between ocean iron fertilization and fisheries biomass on longer timescales, since a reduction of nutrients would ultimately lead to a reduction in fish biomass (Tagliabue et al., 2023).

Macroalgae production at scale would likely lead to the production of vast amounts of dissolved organic matter (DOM), both in the photic zone during growth and in deeper water during senescence and breakdown (Krause-Jensen & Duarte, 2016; Perkins et al., 2022). As they grow, certain types of macroalgae can release as much as 35% of their fixed carbon as DOM – especially relatively labile, soluble polysaccharides (Paine et al., 2021 and references therein) – and be an important food source for bacteria and other organisms. DOM is also released by other forms of biomass, including both phytoplankton (especially diatoms) and terrestrial biomass, although the chemical form of that DOM differs among biomass types (Wada et al., 2008).

The effects of biomass cultivation and storage on marine ecosystems represent a balance between the rate of release of chemical species and the physical processes that transport, mix, and dilute those signals in the ocean. During growth, dissolved and particulate marine biomass can be passively released (i.e., by fertilized microalgae sinking as part of the biological pump, or by allowing DOM release or macroalgae fronds to break off in regions of afforestation), and its downstream trajectory will depend on local currents. Since the sinking rates of biological material are generally slow, natural dispersal is expected to have relatively large effects on mid-water ecosystems where the biomass would decay (Omand et al., 2020; Ross et al., 2023). Changing the sinking rate and/or composition of exported organic matter would alter its availability to consumers like bacteria and zooplankton at midwater and deep pelagic environments, with potential negative effects on their food webs. Shallower water masses also generally upwell to the surface on relatively short timescales (Siegel et al., 2021), carrying the signal of biomass breakdown. Alternatively, biomass packages may be actively placed in specific regions (i.e., packaged macroalgae or terrestrial biomass being placed within specific zones on the seafloor), minimizing exposure time of biomass to the pelagic water column and maximizing sequestration time.

The breakdown of carbon biomass in the water column drives a variety of biogeochemical changes with implications for natural upwelling systems and artificial upwelling impacts. Artificial upwelling is a proposed CDR method where subsurface water rich in nutrients is transported to the surface to stimulate microalgae growth (Jürchott et al., 2024; Pan et al., 2016). Although the nutrient contents of biomass sources vary, breakdown will always release some amount of dissolved nutrients, DOM, and DIC to seawater. As the water masses containing CDR products upwell, they could thus stimulate NPP through elevated nutrient and DOM concentrations and modify the heterotrophic community (i.e., bacteria, zooplankton, fish). However, artificial upwelling also brings cold, dense, DIC-rich water to the surface, which can lead to surface cooling and indirect biogeochemical effects (Oschlies et al., 2010b), while also reducing the efficiency of carbon storage by returning the dissolved DIC that was intended to be sequestered in the deep ocean. Additionally, respiration in most of the modern ocean is powered by dissolved O_2, which means that enhanced biomass breakdown would reduce O_2 concentrations both locally and in downstream environments (Oschlies et al., 2025). Reduced oxygen concentrations in upwelled water may also lead to an expansion or intensification of low-oxygen zones in the upper thermocline (Levin et al., 2023), which could compress habitats for zooplankton and fish and subsequently lead to an increase in predation and competition (Deutsch et al., 2015). Even at small scales, localized O_2 limitation is a likely consequence of biomass placement in oxic zones, especially within physical piles with limited circulation.

On a global scale, the oxic respiration of additional biomass equivalent to 1 GtC yr^{-1} would enhance global O_2 consumption in the deep ocean by ~125 Tmol O_2/yr, a roughly 130% increase in the rate of ocean O_2 loss observed over the last several decades due to climate change (Schmidtko et al., 2017). Especially if additional O_2 demand is localized to a region, this decline has potentially severe consequences for both benthic and pelagic ecosystems (Deutsch et al., 2015; Kim et al., 2023; Laffoley & Baxter, 2019; Oschlies et al., 2025). In contrast, in oxygen-deprived environments, microorganisms can remineralize organic carbon through anaerobic metabolisms such as nitrate reduction, sulfate reduction, or fermentation (Froelich et al., 1979; Nissenbaum et al., 1972). Each of these metabolisms can release different species and have different effects on pH, DIC, and nutrient concentrations (Middelburg et al., 2020; Raven et al., 2024). Sulfate reduction is likely to be a particularly important pathway for microbial respiration in anoxic waters due to the abundance of sulfate in seawater. Microbial sulfate reduction produces relatively small effects on pH but relatively large potential effects on sulfide concentrations. Naturally occurring releases of sulfide (e.g., from organic matter-rich shelf sediments) are associated with toxicity to fish, although sulfide will oxidize relatively quickly back to sulfate in the presence of oxygen (Hamukuaya et al., 1998; Schunck et al., 2013). Several other potential products of microbial breakdown include greenhouse gases, such as N_2O and methane; their release would impact the overall assessment of climate outcomes for any intervention.

The location and method of biomass placement can significantly shape the ecological consequences of biomass-based CDR for both pelagic and benthic ecosystems. One option of growing interest is using relatively isolated, deep anoxic basins as sites for biomass storage (Raven et al., 2024). These basins, potentially including hypersaline brines as well as sulfidic environments like the Black Sea, represent relatively closed systems that isolate sequestered carbon due to their restricted circulation and also avoid direct impacts on benthic animals, which are naturally absent due to the lack of oxygen. Whether the products of these microbial reactions will impact ecosystems in the surrounding, oxygenated ocean will depend critically on the circulation at the selected site and the projected stability of its circulation and stratification into the future. Overall, the selection of a sequestration site for biomass on the seafloor or, in some cases, at mid-water depths, will be a key determinant of the potential impacts of biomass-based CDR on fisheries and other marine ecosystems.

Benthic ecosystems are fundamental for habitat structure, nutrient cycling, and provide numerous ecosystem services. On the seafloor, biomass materials will interact with benthic communities (i.e., organisms that live in or on the seabed) as a substrate, a potential food source, and a driver of geochemical change. The unique adaptations and ecosystem roles of these communities are poorly understood; thus, impact predictions are difficult. However, organisms adapted to low energy environments have a high likelihood of sensitivity to environmental disruption and slow disturbance recovery (Gollner et al., 2017). During biomass placement on the seafloor, these ecosystems are susceptible to physical disruptions from smothering by biomass or induced sediment flows (Levin et al., 2023). Benthic animal communities are also highly sensitive to oxygen availability. Local anoxia caused by seafloor respiration could cause aerobic heterotrophs (worms, nematodes, microbes) to be disrupted, migrate or suffocate (Barry et al., 2013; Oschilles et al., 2025), while microbial communities will adjust to favor organisms that are well adapted for low-oxygen, high-biomass conditions, with hard-to-predict consequences for ecosystem services and total carbon turnover (Orcutt et al., 2020). Examples of discrete natural biomass additions like whale falls and kelp falls are known to attract a complex community of grazers (Smith et al., 2015) and can impact the rate of breakdown for organic matter already in the sediments, which can affect the efficiency of carbon storage (van Nugteren et al., 2009; Zhu et al., 2024). Targeted seafloor placement may reduce the risks of direct interactions between biomass-based CDR and sensitive or protected benthic communities.

Protecting and restoring marine ecosystems and fisheries, such as increasing the number of whales or large fish, would provide obvious benefits to marine ecosystems and could be part of a marine CDR portfolio, though there are substantial uncertainties in the potential carbon fluxes (Figure 1; Bianchi et al., 2021; Collins et al., 2025; Mariani et al., 2025). Since marine food webs play a central role for the oceanic biological pump, these efforts could have implications for CDR, for example by increasing carbon export through fecal pellets and sinking carcasses

(Mariani et al., 2020), though some simulations have shown no impacts to carbon storage (Dupont et al., 2023). Similarly, while coastal ecosystems such as wetlands and mangrove forests may store significant biomass per unit area, they cover a thin sliver of the Earth's surface and may release methane, a potent greenhouse gas (Cotovicz Jr. et al., 2024; Zheng et al., 2018). Future research goals in this space include understanding the role of fish and fisheries in carbon sequestration (Collins et al., 2025; World Bank, 2024) and to assess whether carbon removal by ecosystem restoration could be substantial relative to the scale of negative emissions required in the coming decades (Smith et al., 2024).

3.2 Abiotic methods for ocean-based CDR

Ocean alkalinity has been a major control on atmospheric carbon levels throughout Earth's history, acting as the primary thermostat for the planet. Changes in the delivery of alkalinity by rivers and its eventual burial in marine sediments, driven by changes in atmospheric CO_2 and associated climate patterns, have provided a self-regulating negative feedback on atmospheric carbon concentrations over glacial-interglacial cycles and longer geologic timescales (Berner et al., 1983; Renforth & Henderson, 2017). Ocean Alkalinity Enhancement (OAE) is a climate intervention which seeks to increase the ocean's storage of dissolved inorganic carbon by adding large amounts of alkaline materials to, or electrochemically increase the alkalinity of, the ocean (Figure 1; NASEM, 2022).

Alkalinity is a complex chemical quantity, defined by the excess concentration of proton-acceptors over proton-donors, which allows a solution to buffer changes in pH (Wolf-Gladrow et al., 2007). Natural or anthropogenic alkalinity addition captures carbon by chemically shifting the speciation of the carbonate system away from CO_2, thereby enabling the ocean to absorb more CO_2 from the atmosphere. In the modern ocean only 1% of dissolved inorganic carbon (DIC) exists as aqueous CO_2 ($CO_2(aq)$ + carbonic acid). The rest is effectively trapped as bicarbonate (HCO_3^-) and carbonate (CO_3^{2-}), ionic forms of carbon which cannot directly exchange with the atmosphere. However, this partitioning of DIC between forms is highly sensitive to alkalinity, making alkalinity the primary control on the ability of the ocean to absorb atmospheric CO_2 (Dickson, 1992; Emerson & Hedges, 2008; Zeebe & Wolf-Gladrow, 2001).

Without alkalinity, DIC in seawater would primarily exist as CO_2, resulting in much less carbon being stored in the ocean and an order of magnitude more CO_2 in the atmosphere (DeVries, 2022). Additionally, because alkalinity buffers changes in pH (Dickson, 1981), OAE can sequester carbon into the ocean without exacerbating, and potentially locally mitigating, the well-established adverse impacts of ocean acidification on marine ecosystems (Butenschön et al., 2021; Doney et al., 2009, 2020; Falkenberg et al., 2020).

Alkalinity can be derived from numerous mineral, electrochemical, and industrial sources (Figure 1; Eisaman et al., 2023; National Academies of Sciences, Engineering, and Medicine, 2022). Seawater has naturally gained alkalinity over time through chemical weathering (i.e., rock erosion) and the dissolution of carbonate ($CaCO_3$) and silicate ($(Mg,Fe)_2SiO_4$) minerals (Berner et al., 1983). Minerally sourced OAE directly mimics this natural process by mining, grinding, and dissolving carbonate and silicate rocks into the ocean (Renforth & Henderson, 2017). Electrochemical methods (electrolysis, electrodialysis) can be more energy intensive, generating alkalinity by splitting seawater into a highly alkaline and highly acidic stream, then safely neutralizing the acid (Eisaman, 2024). Finally, some industrial waste products such as steel slags are highly alkaline and could be repurposed and commodified, but may contain various trace metals and other contaminants (Bach et al., 2019). Different sources of alkalinity come with varying degrees of potential ecosystem impacts, but must be balanced against other energetic and techno-economic tradeoffs (Caserini et al., 2022; Eisaman et al., 2023).

OAE can impact marine ecosystems through: 1) its intended direct perturbation to the carbonate system; and 2) the collateral influence of additional chemicals contained in some alkalinity sources. Direct perturbations to carbonate chemistry are independent of the source of alkalinity and scale only with the quantity of alkalinity added and rate at which it is delivered; however, secondary biogeochemical impacts vary with the specific source of alkalinity selected. Both pathways, the perturbations to carbonate chemistry and the secondary impact of biologically-active chemicals included in the alkalinity source, are most likely to impact marine ecosystems predominantly through shifting the community composition of primary producers (Bach et al., 2019; Henderson et al., 2008; Renforth & Henderson, 2017).

3.2.1 Ecosystem impacts of direct alkalinity addition on carbonate chemistry

The response of carbonate chemistry in seawater to alkalinity addition proceeds through two distinct phases, a highly concentrated but transient unequilibrated phase and a heavily diluted but long-term equilibrated phase. Both could affect phytoplankton abundance and community structure, but on different time scales. Initially, during the unequilibrated phase, the alkalinity addition will reduce the concentration of CO_2 and bicarbonate, while increasing the pH and the concentration of carbonate for a period of months to years (Bach et al., 2023; Feng et al., 2017). During this period, reduced substrate (HCO_3, CO_2) availability could limit calcification (Krumhardt et al., 2017), while reduced proton (H+) inhibition could enhance calcification (Bach et al., 2015). The net effect may enhance calcification by increasing the Substrate-Inhibitor-Ratio (HCO_3:H+; Bach et al., 2015) but could vary with species-specific impacts on calcifiers, which could depend on the baseline carbonate chemistry of the perturbed waters (Bednaršek et al., 2024). In situ OAE has been shown to enhance community calcification rates on a coral reef flat (Albright et al., 2016). Meanwhile, pronounced CO_2 reductions (<100 µatm) could inhibit photosynthesis and reduce primary production (Barcelos e Ramos et al., 2007; Riebesell et al., 1993) or delay bloom formation (Marin-Samper et al., 2024a), but may have varying impacts depending on phytoplankton size and physiology (Hansen, 2002; Hansen et al., 2007), confounding which species might be impacted most (Chrachri et al., 2018; Malerba et al., 2021). Recent microcosm (Ferderer et al., 2022; Guo et al., 2024) and mesocosm (Ferderer et al., 2023; Ramirez et al., 2025) experiments have demonstrated statistically significant changes in phytoplankton community composition and succession following unequilibrated alkalinity additions on timescales of days to weeks, with pronounced impacts on picoeukaryotes and mixed outcomes for diatoms. The time and space scale of this initial perturbation will depend on the rate/scale of alkalinity addition, timescales of gas exchange (weeks-months), and physical dilution/subduction of alkalinity (location dependent; see Zhou et al., 2024). However, for a single concentrated pulse of unequilibrated alkalinity, bulk productivity appears largely resilient (Marin-Samper et al., 2024a) and the perturbation would likely dilute, equilibrate and attenuate very quickly (Bach et al., 2019). This transient exposure to highly elevated pH is unlikely to impair phytoplankton growth rates beyond a tipping point from which the community can safely recover once the system has equilibrated (Delicroix et al., 2024; Oberlander et al., 2024; Ramirez et al., 2025).

During the equilibrated phase, alkalinity enhancement also leads to long-term, but smaller, perturbations to the carbonate system that may alter phytoplankton community structure on longer time scales. After the initial alkalinity enhancement, the CO_2 deficit in seawater is filled by the invasion of CO_2 from the atmosphere, largely restoring the speciation of DIC on time scales of several years (González & Ilyina, 2016; He & Tyka, 2023; Zhou et al., 2024). However, the combined effect of elevated alkalinity and DIC after equilibration will lead to a slight increase in pH and moderate increase in CO_3^{2-} and HCO_3^- relative to the unperturbed carbonate chemistry (Bach et al., 2019; Ilyina et al., 2013). This new, equilibrated chemical state could permanently increase calcification rates (Bach et al., 2015) and favor the growth and abundance of calcifying organisms (Henderson et al., 2008; Riebesell et al., 2017). For example, the Black Sea is a highly alkaline marginal sea and experiences remarkably large coccolithophore blooms (Kopelevich et al., 2014) compared to otherwise similar lower alkalinity marginal seas (Sorokin, 2002). A shift toward calcifiers could alter the carbon sequestration efficiency, as the process of calcification increases pCO_2 (Kwiatkowski et al., 2025), while the increased ballasting material could potentially increase export efficiency. However, early work has shown no evidence of changes in the stoichiometry, settling velocity or remineralization rates of sinking particles in an equilibrated mesocosm (Suessle et al., 2025). Finally, a sustained pH change could alter nutrient cycling through pH-dependencies on, for example, heterotrophic nutrient turnover (Taucher et al., 2021) and iron solubility (Liu & Millero, 2002); however, the observed impact on food quality (e.g. C:N; Ferderer et al., 2022; Paul et al., 2024), iron bio-availability (Gonzalez-Santana et al., 2024) and bulk productivity (Marin-Samper et al., 2024b; Ramirez et al., 2025) so far appears relatively low in response to equilibrated OAE.

3.2.2 Ecosystem impacts of associated mineral and trace metal contaminants

Dissolved chemicals derived from the specific source of alkalinity could have additional, and much larger, ecosystem impacts than direct alkalinity addition, but depend on the source of alkalinity selected (Bach et al., 2019; NASEM, 2022). At one end of the spectrum, electrochemically derived compounds (OH^-, NaOH) are the most expensive, but purest source of alkalinity in that all associated chemicals are generally derived from the salt already in sea water and largely biologically inert at the scales they would be added (Eisaman et al., 2023; NASEM, 2022). At the other end of the spectrum, relatively cheap industrial waste products may contain high concentrations of trace metal contaminants (Chromium: Cr, Molybdenum: Mo, Nickel: Ni, Lead: Pb) with largely unknown impacts on biology (Bach et al., 2019). In between, mineral sources of alkalinity can contain high concentrations of biologically important nutrients (SiO_4, Fe) and/or alkaline earth metals (Ca^{2+} , Mg^{2+}) that may influence calcification (Bach et al., 2019). Two of the most common mineral sources of alkalinity are naturally occurring silicates such as olivine ($(Mg,Fe)_2SiO_4$) and industrially processed calcium carbonate derivatives such as quicklime (CaO).

Chemical byproducts of olivine OAE could cause large shifts in phytoplankton productivity and community structure. Olivine dissolves into silicic acid, magnesium and iron (Taylor et al., 2016). Magnesium is already highly concentrated in the ocean relative to what would be added, but may locally inhibit calcite (but not aragonite) production (Davis et al., 2000) at concentrated source locations. Silicic acid and iron, however, are critical nutrients that respectively limit the productivity of diatoms globally (Krause et al., 2019; Ragueneau et al., 2006) and all phytoplankton in High Nutrient, Low Chlorophyll regions (Aumont & Bopp, 2006; Tagliabue et al., 2017), and could thus have a much broader impact even once diluted beyond their source locations. Modeling suggests that sequestering ~1 GtC yr^{-1} (equivalent to approximately 10% of global CO_2 emissions in 2023) with olivine could triple the natural source of silicate (Beusen et al., 2009) and double the natural source of iron (Mahowald et al., 2005), with the resulting fertilization increasing global NPP by up to 40% and shifting community composition towards diatoms (Hauck et al., 2016; Kwiatkowski et al., 2023). The resulting ecosystem effects are broadly similar to those projected for iron fertilization strategies that cultivate microalgae, including enhanced productivity and altered food web structure. Further fertilization over an extended deployment (year-decades) will likely have diminishing additional ecosystem impacts as other macronutrients become limiting.

OAE using calcium-based minerals has fewer potential ecosystem impacts than using olivine. Calcium-based minerals contain relatively few impurities and can essentially be considered a source of only calcium ions (Ca^{2+}) and alkalinity. Global background calcium concentrations are high, meaning the relative perturbation will be small, but could still theoretically accelerate biological calcification (Stanley et al., 2005) and induce a shift toward coccolithophore dominance. Compared to olivine-sourced alkalinity, which promotes the growth of silicifiers and may inhibit calcification, a calcium-based mineral alkalinity source could favor a coccolithophore over diatom dominated phytoplankton community (Bach et al., 2019). However, recent cultured experiments using calcium-based and olivine minerals, respectively, found little effect on coccolithophore physiology (Gately et al., 2023) and no competitive advantage for diatoms relative to coccolithophores, even when directly competing (Hutchins et al., 2023).

Finally, any mineral additive or industrial by-product is liable to contain biologically active trace metals in high concentrations compared to the ocean (Hartmann et al., 2013). For example, in addition to iron, olivine can contain very high concentrations of nickel (Hutchins et al., 2023; Simkin & Smith, 1970), which can act as a nutrient or toxin depending on species and concentration (Glass & Dupont, 2017; Guo et al., 2022). Nitrogen-fixing cyanobacteria are most likely to benefit from elevated nickel (Dupont et al., 2010), while some species of diatoms (Xin et al., 2024), mussels (Millward et al., 2012) and crabs (Blewett et al., 2015) may experience toxic effects at high concentrations. However, there is little evidence of significant bio-accumulation (Nieminen et al., 2007). Additional trace metals (e.g., Cr, Mo, Pb) found in naturally occurring silicates (Beerling et al., 2018) and industrial by-products (Renforth, 2019) may be more harmful or likely to bioaccumulate (Wilson et al., 2019) and should still be assessed.

3.2.3 Impacts on marine food webs

The overall impact of OAE on food webs will likely hinge predominately on source-specific shifts in community composition of primary producers and how these bottom-up drivers propagate up the food-web (Mitra et al., 2014). However, direct impacts must also be considered, particularly at highly concentrated source sites. From a carbonate chemistry perspective, extended exposure (months) to high pH (>9.5) during the initial transient OAE response phase can adversely affect some fish species (Locke et al., 2009; Saha et al., 2002; Scott et al., 2005) and benthic epifauna (Jones et al., 2025), although such sustained conditions are improbable (Renforth & Henderson, 2017). Mesocosms of the subsequent, equilibrated phase have shown no sustained adverse effects on zooplankton community structure, productivity or nutritional quality (Sánchez et al., 2024), nor on various juvenile fish species (Goldenberg et al., 2024). From a mineral additive perspective, oxidation of Fe(II) to Fe(III) from olivine could consume dissolved oxygen at depth, but this could likely be mitigated by reasonable dispersal rates (Renforth & Henderson, 2017; Schott & Berner, 1985). Elevated mineral particulate from the direct deposition of alkaline source particles or the spontaneous precipitation of $CaCO_3$ induced by those particles (Paul et al., 2024) could impair visual predators like fish (Eisner et al., 2005) and seabirds (Baduini et al., 2001) or clog the mesh of filter feeding zooplankton (Guo et al., 2024). However, high particulate concentrations are expected to attenuate on the order of days (Balch et al., 2009). Finally, if naturally-sourced minerals dissolve slowly, they may become incorporated into organic particles, potentially reducing the nutritional quality of food available to zooplankton (Fakhraee et al., 2023).

3.3 Relative importance of marine ecosystem impacts across CDR approaches

It is not possible to rank the overall impact of a given intervention without assuming a normative weight on individual impacts, many of which do not have an obvious qualitative connotation. For instance, the creation of new biomass could be considered positive from an ocean productivity or fisheries perspective but negative from an ecosystem disruption perspective. Both perspectives are valid and likely to diverge between different interest groups. Instead of adjudicating those differences, we assess the importance of each individual impact pathway across different marine CDR interventions and report how well it is currently represented in models (Figure 2). Climate interventions with more impact pathways with higher relative importance and/or worse model representation (see Section 5; Figure 2 & 4) carry more high-risk uncertainty.

Biomass-based CDR approaches can influence marine ecosystems through cultivation, growth, leaching, transport and breakdown of new biomass. Nutrient impacts (Figure 2a) are largest for methods that directly add macro- and micro-nutrients to the surface ocean, including microalgae growth due to iron fertilization as well as some approaches for macroalgal cultivation (Aumont & Bopp, 2006; Moore et al., 2001; Ustick et al., 2021). However, nutrient impacts (e.g., nutrient robbing) may be mitigated for macroalgae cultivation if macronutrients (N, P) are added instead of micronutrients (Fe) because this may not deplete the downstream preformed nitrate concentration (Berger et al., 2023; Oschlies et al., 2010a). Terrestrial biomass storage at depth avoids direct perturbation to surface nutrients, but nutrients released from biomass breakdown can eventually upwell and impact surface-ocean environments, especially for well-ventilated (typically oxic) placement sites. The creation of new biomass (Figure 2b) from macroalgae cultivation in particular, and microalgal blooms to a lesser extent, produces physical habitat in the photic zone and can shade deeper organisms (Gallo et al., 2025; Levin et al., 2023). Dissolved organic matter (Figure 2c) can impact ecosystems in complex ways, as food, a nutrient, or an inhibitor; it is produced during the breakdown of essentially all biomass types, and it can be released in particularly large quantities during the growth and senescence of macroalgae and some other plankton (e.g., diatoms) (Krause-Jensen & Duarte, 2016; Perkins et al., 2022). As new biomass is physically transported (Figure 2d), the risk of smothering benthic ecosystems is highest for dense biomass that sinks efficiently and encounters seafloor animal communities (Omand et al., 2020; van Nugteren et al., 2009; Zhu et al., 2024). Macroalgal biomass is also particularly likely to transport passenger organisms (e.g. epibionts, viruses) during sinking and to attract opportunistic scavengers, although this impact is minimized for anoxic storage sites, where few exogenous organisms would survive (Figure 2e). During breakdown (Figure 2e), O_2 consumption may be a particularly important impact for any biomass type in naturally oxic water (Buesseler et al., 2024; Doney et al., 2024). Anaerobic respiration may occur within biomass piles at oxic sites, but sulfide release to the environment will be most substantial in anoxic environments (Hamukuaya et al., 1998; Middelburg et al., 2020; Schunck et al., 2013). Other effects of respiration are more evenly distributed across pathways but will be more acute for biomass types that degrade relatively quickly, releasing DIC and impacting downstream carbonate chemistry.

Perturbation from Intervention		Impact on Marine Ecosystems	Importance of Impact to:							Model Inclusion	
			Biotic CDR methods				Abiotic CDR methods			Earth System Models	Marine Ecosystem Models
			Macro Algae Cultiv.	Micro Algae Fert.	Terrestrial Bio -mass Storage		Carbonate Mineral OAE	Silicate Mineral OAE	Electro-Chemical OAE		
					Oxic	Anoxic					
a. Addition of micro- or macro- nutrients (Fe, N, P, Si)	i	Relieved nutrient stress increases magnitude of global NPP[1]	Med	High	Low	None	None	High	None	✓✓	✓✓
	ii	Downstream nutrient robbing shifts global distribution of NPP[1]	Med	High	None	None	None	High	None	✓✓	✓✓
	iii	Shift in balance of nutrients favors different primary producers[1]	Med	High	Low	None	None	High	None	✓✓	✓
b. Creation of new biomass	i	New physical habitat/refuge modifies surface ocean ecosystems	High	Low	None	None	None	Low	None	x	x
	ii	Self shading shifts vertical distribution and magnitude of local NPP[1]	Med	Med	None	None	None	Med	None	✓✓	✓
c. Dissolved organic matter and gasses released from biomass	i	Highly labile DOM provides food for bacteria and other organisms	High	Low	Low	Low	None	Low	None	✓	✓
	ii	Increased DMS production increases cloud formation and cools ocean[1]	Med	Med	None	None	None	Med	None	✓	✓✓
	iii	Increased N_2O production acts as greenhouse gas and heats ocean[1]	Low	Low	Low	Low	None	Low	None	✓	✓✓
d. Physical transport of biomass	i	Smothers benthic organisms	High	Low	High	Low	None	Low	None	x	x
	ii	Transports passenger organisms and viruses	High	Med	Low	Low	None	Med	None	x	x
e. Breakdown and respiration of biomass	i	Deep biomass attracts opportunistic scavengers/invasive species	High	Low	High	Low	None	Low	None	✓	✓
	ii	Aerobic respiration consumes O_2 and can create hypoxic or anoxic conditions, locally or downstream	High	High	High	None	None	High	None	✓✓	✓
	iii	Anaerobic respiration produces sulfide, which is toxic until oxidized	Med	Low	Med	High	None	Low	None	x	x
	iv	Respiration releases nutrients and can affect pH, locally or downstream	High	Med	Med	Med	None	Med	None	✓✓	✓
f. Equilibrated, long-term, shift in carbonate chemistry	i	Small change in CO_3^{2-}, HCO_3^-/H^+ impacts calcifiers[1]	Low	Low	None	None	Low	Low	Low	✓	✓
	ii	Change in calcifier composition alters export to mesopelagic	Low	Low	None	None	Low	Low	Low	✓	✓
	iii	pH dependent nutrient uptake/recycling alters NPP[1] and food quality	Low	Low	None	None	Low	Low	Low	✓	✓
g. Unequilibrated, transient shift in carbonate chemistry	i	Large decrease in CO_2 favours different primary producers[1]	None	None	None	None	Low	Low	Low	x	✓
	ii	Large increase in CO_3^{2-}, HCO_3^-/H^+ favours calcifiers[1]	None	None	None	None	Low	Low	Low	✓	✓
	iii	Large increase in pH may be harmful to some fish	None	None	None	None	Low	Low	Low	✓✓	✓
h. Addition of bioactive contaminants	i	Increase in Mg^{2+} disadvantaging calcifying organisms[1]	None	None	None	None	None	Low	None	x	x
	ii	Increase in Ca^{2+} advantages calcifying organisms[1]	None	None	None	None	Low	None	None	x	x
	iii	Increase in nickel possibly toxic to some mussels and crabs	None	None	None	None	None	Med	None	x	x
	iv	Increase in nickel benefits growth of nitrogen fixing phytoplankton[1]	None	None	None	None	None	Med	None	x	✓
	v	Increase in trace metals (Cr, Mo, Ni, Pb) can be toxic or nutritious to different phytoplankton[1], but unlikely to bioaccumulate	None	None	None	None	None	Med	None	x	x
i. Local increase in inorganic particulate	i	Increase in opacity of water could reduce PAR and NPP	None	None	None	None	Low	Low	None	x	✓✓
	ii	Increase in opacity of water could impair visual predators	None	None	None	None	Low	Low	None	x	✓
	iii	Particles could clog mesh of filter feeding zooplankton, impacting NPP[1]	None	None	None	None	Low	Low	None	x	✓
	iv	Poor food quality could harm grazers, impacting NPP[1]	None	None	None	None	Low	Low	None	x	✓
	v	More mineral surface induces precipitation, increasing particulate	None	None	None	None	Low	Low	None	x	✓

✓✓ Commonly included/ Well represented

✓ Occasionally included/ Partially represented

X Rarely included/ Poorly represented

Relative Importance of Impact

High | Med | Low | None

*Additional perturbations associated with deployment are discussed in **Section 3.3**

[1] Changes in magnitude/distribution of NPP/temp/ community composition impact higher trophic levels

Figure 2. The relative importance and representation of perturbations (**a-i**) and impacts (**i-v**) for CDR on marine ecosystems. The relative importance of each impact pathway to each CDR method is qualitatively described as high (red), medium (light red), low (light blue), or not relevant (blue). Additional impacts associated with deployment, which may vary within a given intervention, are discussed in Section 3.3. Relative importance represents an approximate, aggregate measure of the scale, uncertainty, and significance of a given impact. The general ability of Earth system (ESM) and marine ecosystem models (MEM) to simulate each impact pathway is summarized as a rough indication of inclusion across models used in prominent inter-comparison projects. MEMs are typically forced with temperature and plankton (net primary productivity or biomass) outputs from ESMs, and occasionally pH and/or photosynthetically active radiation (see **Section 5.3**). Model inclusion scores refer to 1) the ability to resolve changes in physics, chemistry and plankton in ESMs, and 2) the response of higher trophic levels in MEMs to ESM forcing are considered inclusive of corals and visual predators, in that they are the responsibility of the MEM. Perturbations can impact higher trophic levels *directly* through their physiological sensitivity (resolved in MEMs) to physical/chemical changes (resolved in ESMs) or *indirectly* through their predation response (resolved in MEMs) to physically/chemically driven changes in plankton (resolved in ESMs). Inclusion is scored with **two checks (✓✓)** if most *direct* and *indirect* pathways are reasonably well represented in most models, **one check (✓)** if only some *direct* or *indirect* pathways are reasonably well represented in some models, and a **cross (x)** for rare and/or insufficient inclusion. If there is only a *direct* impact on higher trophic levels, ESMs are scored exclusively on their ability to resolve the relevant physics/chemistry (e.g., **g - iii**). If there is only an *indirect* impact on higher trophic levels, ESMs must resolve the entire physical/chemical *and* plankton response to receive a check (e.g. **a - i**). MEMs are not penalized if the ESM does not resolve something that would otherwise be captured in the MEM forcing fields, meaning MEMs can receive checks even if the ESM does not (e.g. **g - i**). If there is a *direct* and *indirect* impact on higher trophic levels, ESMs and MEMs can receive a single check if they present a limited inclusion of either the *direct* or *indirect* impact pathway (e.g. **f - i**). These metrics are not intended to provide an exhaustive or specific review of all model configurations, but rather an informed, high-order assessment of model capabilities and processes that may need more development.

While carbonate or electrochemically derived OAE should not substantively impact the creation, transport, or breakdown of new biomass, the impacts of silicate-based OAE are considered equivalent to microalgae fertilization (Figure 2a-e) due to the large iron and silicate content of mafic minerals (Taylor et al., 2016). Olivine contains roughly 1 mol of iron and 5 mol of silicate per 20 mol of alkalinity, meaning the ~3 Pg of olivine required to abiotically sequester ~1 GtC would add as much as 230 Tg of iron to the ocean (Hauck et al., 2016). Much of that iron would be quickly scavenged onto particles in its colloidal form (Liu & Millero, 2002), but if only 0.1% remains bioavailable, it would double the natural deposition of bioavailable iron from dust (Mahowald et al., 2005), in addition to tripling the natural source of silicate (Beusen et al., 2009).

All interventions, both biotic and abiotic, that add DIC to the ocean could impact marine ecosystems through shifts in the carbonate system on long-term (Figure 2f) and transient (Figure 2g) timescales. On long-term timescales (decades-centuries) an equivalent amount of CDR from biotic interventions that primarily store carbon as DIC following the oxic respiration of created biomass (micro- and macroalgae) could lead to larger carbonate shifts, albeit in the opposite direction (acidification), than from abiotic (OAE) interventions which buffer DIC-driven shifts in carbonate chemistry through the addition of alkalinity. However, to successfully sequester carbon, shifts in carbonate chemistry driven by oxic respiration would need to occur deeper in the water column, where they could be decoupled from marine ecosystems in the surface ocean. On shorter, transient time scales (hours-days), these shifts are primarily relevant to abiotic, alkalinity-based interventions that can drive a more rapid (seconds-minutes) and concentrated shift in carbonate speciation than biotic interventions, which are levelized across the timescales of biological productivity, remineralization and export (days-weeks). Still, the importance of these transient impacts to abiotic marine CDR are considered low, as large perturbation to carbonate chemistry should dilute and equilibrate before causing permanent biological harm (Bach et al., 2019; Delacroix et al., 2024; Oberlander et al., 2025; Ramírez et al., 2025). The impacts of bioactive contaminants (Figure 2h) are limited to minerally sourced alkalinity and industrial by-products and vary based on source material. The impacts of elevated inorganic particulate (Figure 2i) are limited to solid-source OAE and vary based on grain size.

Finally, different interventions will have different impacts associated with the mechanical and practical constraints on how they are deployed. However, because these impact pathways are not necessarily linked to the biogeochemical pathways through which carbon is captured or stored, they could vary substantially within a given intervention (and thus are not included in Figure 2). The potential impacts of different deployment mechanisms are discussed below. A full life cycle analysis of how materials are sourced and transported is not considered here but is critical in the cost-benefit assessment of any project.

For biotic methods, the largest direct practical impacts are related to the ship-based dispersal of nutrients for microalgae fertilization or deposition of biomass. Increased ship traffic could have wide ranging impacts related to the effects of ship noise on marine mammals (Erbe et al., 2019), introduction of invasive species (Costa-Areglado et al., 2025) and/or chemical pollutants (Kurniawan et al., 2022) in ballast water, increased risk of whale encounters including ship strikes (Nisi et al., 2024), increased emissions (Deng & Mi, 2023), and potential for habitat destruction during port development (Bulleri & Chapman, 2010). These impacts would likely be larger for microalgae fertilization than biomass storage due to the breadth and remoteness of siting locations, e.g., the Southern Ocean (Oschlies et al., 2010a) compared to the Black Sea (Raven et al., 2024). Microalgae fertilization through artificial upwelling would require the construction and ship-based mobilization of hundreds of thousands of 500+ m long pipes across the open ocean (Koweek, 2022). Macroalgae cultivation is more likely to occur coastally, obviating the need for a large shipping fleet, but could increase impacts associated with small vessel traffic and increased coastal infrastructure in more ecologically sensitive coastal regions (Campbell et al., 2019).

For abiotic methods, alkalinity can be added to the ocean in solid form via open-ocean ship-based dispersal, coastal benthic deposition, or river liming, and in dissolved form through either non-electrochemical reactor-based processes, such as accelerated weathering of limestone (Renforth & Henderson, 2017) or electrochemical systems (Eisaman et al., 2023). The deployment impacts of ship-based dispersal are analogous to biotic methods, but substantially larger due the mass of material needed. For instance, while 1 tonne of iron could stimulate the export of order 1,000 tonnes of carbon into the deep (<250 m) ocean (Buesseler et al., 2004), 1 tonne of olivine, which has a molecular weight of ~140 g/mol and 4 mol of alkalinity/mol, could sequester only 0.25 tonnes of carbon (ignoring the fertilizing contribution of Fe/Si). To evenly disperse 1 Pg of olivine per year (~0.03 Pmol of alkalinity), enough to sequester about 3% of global emissions, it would take approximately 100 dedicated tankers (Köhler et al., 2013). Alternatively, river liming, which eventually sequesters carbon in seawater, has long been used for ecosystem restoration in acidified freshwater systems, but can increase turbidity and induce benthic smothering (Clair & Hindar, 2005). Similarly, coastal deposition of sediments has long been used to reduce coastal erosion through beach nourishment, but is also known to degrade beach and benthic habitats (Saengsupavanich et al., 2023).

On the other hand, dissolved sources of alkalinity could substantially increase coastal infrastructure depending on the degree to which they are incorporated into existing coastal outfalls. Specifically, electrochemically derived alkalinity can be produced by passing a saline feedstock through an electrochemical reactor to produce either dissolved alkalinity (OH^-) or filter out alkaline precipitates ($NaOH$, $Mg(OH)_2$, $Ca(OH)_2$) (Eisaman et al., 2023). Depending on the source of the feedstock (seawater, industrial or desalination byproduct), there could be a range of impacts on the ecological communities. If the feedstock is sourced from seawater, rather than existing desalination or industrial byproducts, additional seawater intakes can lead to the entrainment, impingement and exclusion of marine organisms (Missimer & Maliva, 2018) and further expose entrained organisms to large temperature and pH swings in the reactor. Similarly, accelerated weathering of limestone requires pumping large amounts of seawater through an acidified reactor (Rau & Caldeira, 1999).

4. SOLAR RADIATION MODIFICATION (SRM)

4.1 Stratospheric Aerosol Injection

Stratospheric aerosol injection (SAI) is a proposed intervention technique that would involve injecting aerosols or their precursor gases into the stratosphere to reflect sunlight back into space, thereby reducing the amount of solar radiation reaching the Earth's surface (Figure 1; Crutzen, 2006; Robock et al., 2009). Gaseous precursors would be released at high altitudes by aircraft, which would convert to particles forming a layer that would scatter and reflect sunlight (Richter et al., 2022). In theory, SAI aims to mimic the cooling effect observed after large volcanic eruptions (Sigl et al., 2015; Tilmes et al., 2015). For example, as a result of the sulfur-rich eruption of Mount Pinatubo in 1991, global temperatures temporarily decreased by about 0.2-0.5°C over the next one to two years (Canty et al., 2013; Soden et al., 2002). The duration for which aerosols from SAI would persist in the stratosphere and cool temperatures depends on several factors, including the specific aerosol composition, particle size, altitude of injection, timing of injection, and meteorological conditions (Kravitz & MacMartin, 2020). Aerosols would remain suspended in the stratosphere for a few years, necessitating continuous replenishment (Smith, 2020).

SAI deployed at scale would result in a reduction in global mean temperatures, and with it a reduction in many temperature-associated ecological and societal impacts (Tye et al., 2022). For a wide variety of physical climate effects, a world with high greenhouse gases and SAI would have fewer impacts than a world with high greenhouse gases alone (Irvine et al., 2016; Visioni et al., 2021). However, SAI will likely result in uneven changes in regional climate or precipitation patterns (Jiang et al., 2024), a delay in the recovery of polar stratospheric ozone (Tilmes et al., 2021), and potential changes in marine and terrestrial ecosystem structure (Zarnetske et al., 2021). The complex interactions between aerosols, atmospheric chemistry, and climate are not fully understood (Moch et al., 2023). Furthermore, abrupt termination of SAI, due to technical, political, or economic reasons, could result in rapid and potentially severe temperature increases, known as “termination shock” (Jones et al., 2013). The sudden shift in climate conditions, either from termination or implementation, could have detrimental consequences for ecosystems and societies (Hueholt et al., 2024; Parker & Irvine, 2018; Trisos et al., 2018).

4.2 Marine Cloud Brightening

Marine cloud brightening (MCB), like SAI, aims to cool the climate by increasing the reflection of sunlight to space (Figure 1). However, instead of injecting aerosols into the upper atmosphere, MCB proposals envision pumping sea water through specially designed sprayer systems to generate a fine mist of sea salt aerosol particles. These particles would reflect sunlight directly and seed cloud droplets to brighten low-lying marine clouds and potentially extend their coverage as well (Alterskjær et al., 2013; Latham, 1990; Latham et al., 2012). The addition of aerosol particles under the right conditions would quickly brighten clouds by increasing the number of cloud droplets that form and decreasing their average size; for the same amount of liquid water within a cloud, this would increase the effective surface area available to reflect sunlight (Twomey, 1974, 1977). This "instantaneous" microphysical cloud brightening would then be either compounded or diminished by subsequent macrophysical cloud adjustments. As clouds with smaller droplets are less efficient at forming precipitation, aerosol-induced drizzle suppression can extend the lifetime of clouds or increase their areal coverage and liquid water content, enhancing cooling beyond the Twomey effect (Albrecht, 1989; Yuan et al., 2023; Chen et al., 2024). However, smaller droplets are more easily evaporated at cloud top, driving enhanced turbulent mixing of hot, dry air into the clouds, dissipating them and thus counteracting the Twomey effect (Ackerman et al., 2004; Toll et al., 2019; Glassmeier et al., 2021).

The effect of international shipping on clouds is considered to be the best present-day analogue to a hypothetical MCB deployment (Christensen et al., 2022; Robock et al., 2013). Ship tracks, curvilinear cloud perturbations that can be traced back to pollution from smokestacks of individual ships, were first identified in the scientific literature in the mid-1960s and their implication for the possibility of deliberately cooling the Earth was immediately noted (Conover, 1966). Regional-scale cloud brightening has been observed within major shipping corridors, providing compelling evidence that MCB could be effective at producing a cooling effect in at least some conditions (Diamond et al., 2020). Cloud changes following stringent regulations in ships' sulfur pollution implemented in 2020 (Diamond 2023; Watson-Parris et al., 2022) have been investigated as a contributing factor to accelerated warming in the early 2020s (Gettelman et al., 2024; Jordan & Henry, 2024; Quaglia & Visioni, 2024), in what could be considered an unintentional trial of reverse MCB (Yuan et al., 2024).

The overall efficacy of MCB is uncertain, however, due to the aforementioned cloud adjustment mechanisms that reduce cloudiness. Ship track studies have even shown net darkening in a substantial minority of tracks due to the loss of liquid water (Chen et al., 2012; Coakley & Walsh, 2002). The strength and even direction of MCB forcing can therefore be sensitive to assumptions about the magnitude of sea salt aerosol being injected (Hoffmann & Feingold, 2021; Wood, 2021) and background meteorology (Zhang & Feingold, 2023). Unlike SAI, the forcing from MCB would be spatially heterogeneous, as the most susceptible clouds only exist in certain locations. This heterogeneous forcing pattern opens a number of questions and concerns about local effects where the seeding would take place, and remote effects from changes in oceanic and atmospheric circulation patterns (Hill & Ming, 2012; Jones et al., 2009; Stjern et al., 2018). Balancing the needs for large enough spatial coverage of seeding to produce a desired magnitude of cooling, the existence of susceptible clouds in target seeding locations, and minimization of remote effects has proved particularly tricky for developing plausible deployment strategies in global climate models (Haywood et al., 2023; Hirasawa et al., 2023; Wan et al., 2024). These issues, including how they would affect marine ecosystems, have recently been highlighted as priority research areas for assessing the feasibility of MCB (Diamond et al., 2022).

4.3 Physical & biogeochemical impacts of SRM in marine systems

Despite ongoing assessments of SAI's feasibility and impact on the physical climate system, insights as to its effects on the marine carbon cycle and net primary productivity are limited. To date, Earth system models are the primary tool utilized to investigate how implementing SAI could impact the global net carbon sink in both terrestrial and marine environments (Kravitz et al., 2013; Tjiputra et al., 2016; Yang et al., 2021). Cooling caused by SAI implementation increases CO_2 solubility in seawater, leading to a strengthened global carbon uptake (Cao, 2018; Cao & Jiang, 2017; Plazzotta et al., 2019; Tjiputra et al., 2016; Zhao et al., 2024). Using different simulations, Tjiputra et al. (2016) observed that SAI enhances carbon absorption by the ocean relative to reference scenarios (RCP4.5 and RCP8.5), except in the Arctic due to increased sea ice recovery. This study also reported that SAI can boost the biological pump in certain regions, increasing carbon transfer from the surface to depth, and maintaining the strength of the meridional overturning circulation. Thus, SAI may aid in lowering atmospheric CO_2 levels, albeit with a significant drawback of increased acidification in deeper waters (Tjiputra et al., 2016). While SAI is also expected to limit the reduction in global ocean oxygen content caused by a warmer climate (Cao, 2018), it will inevitably alter ocean circulation and biological processes which could impact oxygen distribution at regional scales (Keller et al., 2014), with potential implications on species compositions, migration patterns, and habitat availability (Cheung et al., 2013).

Theoretically, MCB could directly decrease NPP by exacerbating light and/or temperature limitations on growth or indirectly increase NPP by relieving nutrient limitation from warming-driven stratification. Using a one-dimensional water column model to look at the effects of shading alone on phytoplankton productivity, Hardman-Mountford et al. (2013) found that shading redistributed productivity vertically but did not change the integrated NPP. Partanen et al. (2016) assessed biogeochemical changes in an ESM of intermediate complexity (three-dimensional ocean circulation model with a simplified one-layer atmosphere) with MCB radiative forcing taken from an atmospheric circulation model (Partanen et al., 2012) and found that cooling and shading from MCB slightly decreased global ocean NPP as compared to a moderate unabated warming scenario. This net reduction in NPP was primarily caused by a decrease in temperature-dependent biological growth rates, with a smaller contribution from increased light limitation, and was partially offset by an increase in nutrient availability from weaker stratification. Temperature and light effects were largely confined to the forcing locations but NPP changes were more spatially heterogeneous downstream of the forcing locations due to changes in nutrient transport. Lauvset et al. (2017) compared SAI and MCB implementations in a global climate model (with coupled ocean-atmosphere circulations) and found that both methods limited global declines in ocean NPP as compared to unabated warming but are less effective than mitigation, in part because of their reduction in downwelling sunlight. The strong correspondence between forcing location and environmental and biogeochemical responses seen in the more idealized modeling framework of Partanen et al. (2016) is not found in the fully coupled model results of Lauvset et al. (2017), highlighting the importance of understanding how intervention-driven ocean-atmosphere circulation changes modulate ecosystem responses compared to local temperature and radiation effects alone.

4.4 Marine ecosystem impacts of SRM

Importantly, despite alleviating temperature increases, SRM would not address the ongoing process of ocean acidification, which poses a significant threat to calcifying organisms globally, such as coral reefs. One metric associated with the impacts of acidification on calcifying organisms is the aragonite saturation state (Ω_{arag}). Ω_{arag} is a measure of how likely calcium carbonate in the form of aragonite is to precipitate or dissolve in seawater (Goldsmith et al., 2019). The saturation state of carbonate minerals is given by:

$$\Omega = \frac{[Ca^{2+}] \cdot [CO_3{}^{2-}]}{K_{\text{sp}}}$$

where $[Ca^{2+}]$ and $[CO_3{}^{2-}]$ are the calcium and carbonate ion concentrations, and K_{sp} is the apparent solubility product (Jiang et al., 2015). For values of $\Omega_{\text{arag}} > 1$, calcium carbonate undergoes precipitation. Before the industrial revolution, almost all shallow water reefs were surrounded by waters with high saturation states ($\Omega_{\text{arag}} > 3.3$). However, the spatial extent of reef suitable habitat meeting this threshold has reduced over time and is expected to reduce significantly in the future (Figure 3a; Hoegh-Guldberg et al., 2007; Kleypas et al., 1999). Kwiatkowski et al. (2015) showed that SAI implementation under RCP 4.5 resulted in lower values of tropical Ω_{arag} when compared with RCP 2.6 and RCP 4.5 due to the greater solubility of CO_2 in the cooler oceans, which is also true for recent SAI simulations (Richter et al., 2022; Figure 3b). Compared to the no intervention future, the $\Omega_{\text{arag}} = 3.3$ saturation zone is smaller under intervention, with fewer reefs within the region bounded by the $\Omega_{\text{arag}} = 3.3$ contour (Figure 3b). The benefits of SRM relative to RCP 2.6 and 4.5 scenarios also diminish as the sensitivity of bleaching threshold to Ω_{arag} increases. Therefore, an important area of research would be to understand the synergy between temperature and acidification impacts under SAI to avoid inadvertent exacerbation of risks to coral reefs.

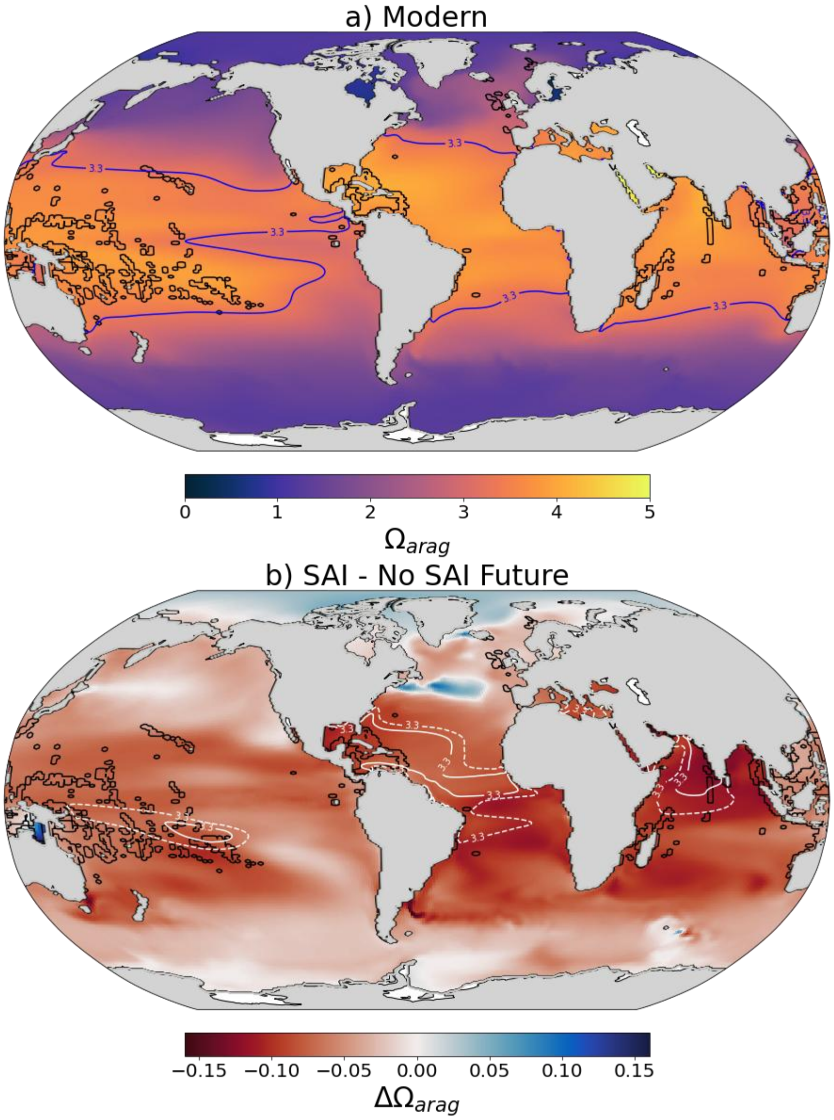


Figure 3 a) Mean of aragonite saturation state (**Ω** $_{arag}$) at the ocean surface over the period 2015-24 under future climate scenario SSP2-4.5. The black lines show the boundaries of existing coral reefs and the blue contour line represents **Ω** $_{arag}$ = 3.3 contour, the historical **Ω** value for healthy shallow water coral reefs. b) Mean of the anomaly of **Ω** $_{arag}$ between the intervention (SAI) and no intervention (SSP2-4.5) scenarios over the last decade of SAI deployment in the model simulations (2060-69). The white contour lines represent **Ω** $_{arag}$ = 3.3, with the SSP2-4.5 (dashed white line) covering relatively more reefs than SAI (solid white line). SAI results in an anomalous decrease in surface ocean aragonite nearly everywhere, relative to a simulation with rising atmospheric CO_2 and no intervention.

The potential of MCB to reduce the toll of heat stress on coral reefs has been of particular interest (Latham et al., 2013, 2014). Condie et al. (2021) modeled MCB over the Great Barrier Reef in northeastern Australia as a reduction in degree heating weeks using a coral reef meta-community model. They found that while MCB was the single most effective intervention of those tested specifically for temperature reduction, its benefits were limited unless paired with other measures like increasing control of coral predators and introducing thermally tolerant coral species. Simulations pairing MCB with the introduction of thermally tolerant corals did show benefits, but these were limited by a marked increase in predation by crown-of-thorns starfish unless targeted control measures were also put in place, highlighting the complexities of ecosystem responses to intervention even in relatively simple model arrangements. As bleaching is influenced by both heat and radiation (Falkowski et al., 1998; Hoegh-Guldberg, 1999; Saito et al., 2008; Tagliabue et al., 2021), the potential for additional benefits of MCB in terms of shading the surface and shifting the distributions of direct radiation deserves further study.

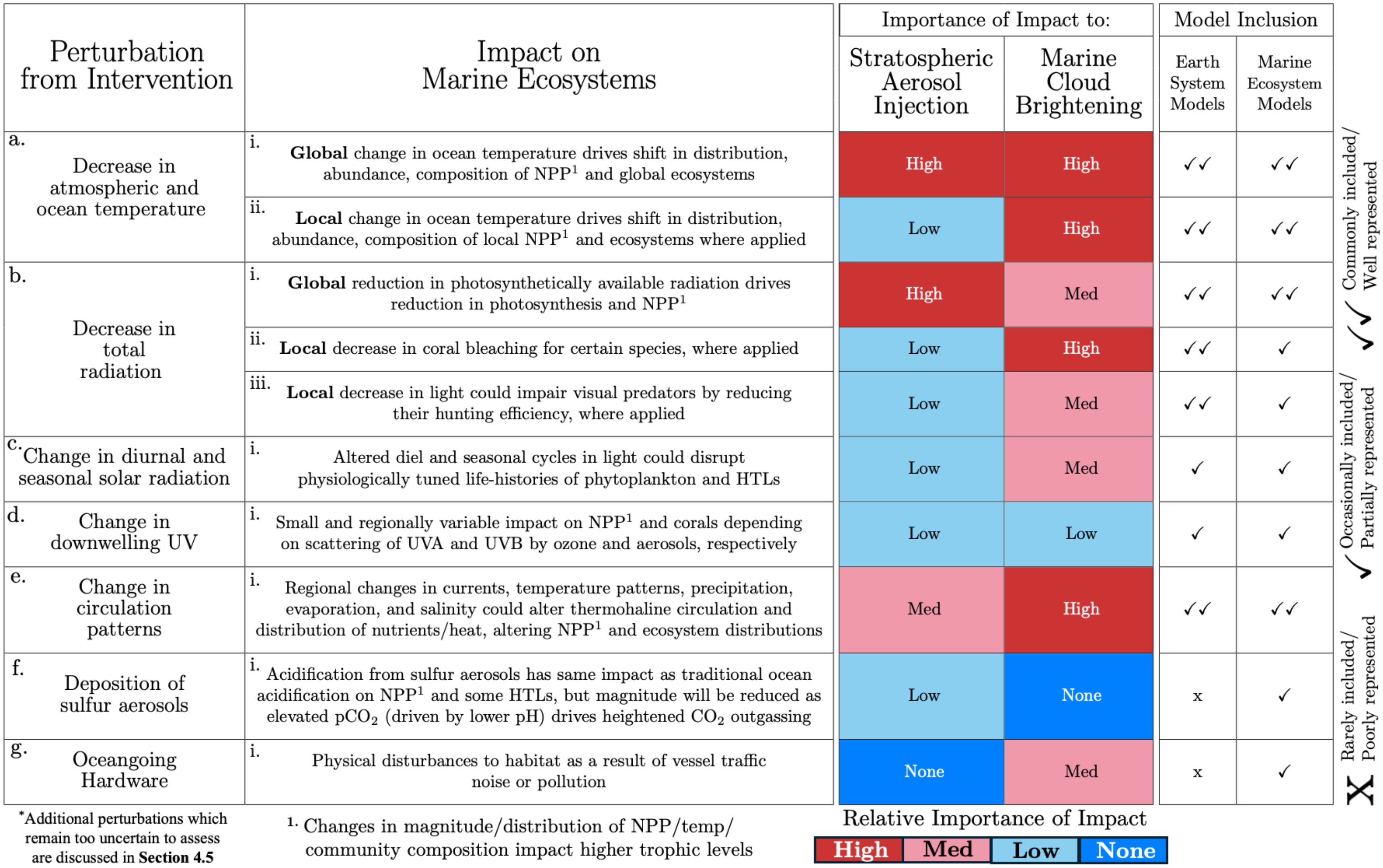
As of publication, MCB has not yet been tested extensively in the field, though there have been small-scale experiments. For example, in Australia urgent climate protection efforts are currently being tested and implemented on the Great Barrier Reef, as roughly half of the coral cover has disappeared, predominantly due to bleaching in the last few decades (Dietzel et al., 2020; Sovacool et al., 2023). Field testing for MCB and artificial shading (i.e., fogging) have been considered (Hernandez-Jaramillo et al., 2023, 2024), along with a portfolio of climate protection pathways that include nature-based ecosystem restoration as part of the Reef Restoration and Adaptation Program (RRAP) launched in 2018 by the Australian Government. Fogging efforts, designed to mimic the effects of sea fog, are more localized in time and space than MCB and could be deployed much sooner. However, fogging lacks the capability to reduce temperatures over broader regional or temporal scales (Sovacool et al., 2023), and is instead designed for rapid deployment to individual reefs before and during marine heat waves.

The influence of SAI or MCB on food webs and fisheries is unknown, since no studies to date have quantified potential impacts. However, given observed impacts of climate change on marine ecosystems, it is likely that if either SRM strategy were implemented, the distribution and biodiversity of marine ecosystems would shift on multiple spatiotemporal scales as a result of reduced temperatures, although at what rate and in which direction would vary regionally and be dependent on the deployment strategy. Previous studies have demonstrated that marine species are sensitive to temperature changes and larger pelagic organisms may migrate to different areas in search of more suitable habitat, which would then alter local fishing patterns and catches (e.g., Cheung et al., 2013; Duffy et al., 2016; Kleisner et al., 2016; Pinsky et al., 2021; Poloczanska et al., 2013; Ramirez et al. 2022). Alterations in diurnal and seasonal solar radiation from an intervention further warrant consideration, as they may disrupt species with life-history traits closely tied to light-driven diel or seasonal cycles. However, the geographical distribution of marine life is not only constrained by temperature. If certain habitats experience reduced nutrient availability as a result of changes to currents and circulation patterns under SRM, it could further affect the community composition and abundance of species that rely on specific feeding grounds. SRM would also result in a decrease in light availability which could impair visual predators by reducing their hunting efficiency and disrupting light-dependent reproductive cues.

4.5 Relative importance of marine ecosystem impacts across SRM approaches

Because the impacts of SRM on the marine environment remain woefully understudied, identifying, let alone quantifying, potential effects on marine ecosystems is inherently challenging. Nevertheless, existing literature, combined with expert judgment, was used to identify potential ecosystem impacts of SRM, assess their relative relevance to SAI and MCB, and evaluate their current representation in models (Figure 4).

The importance of temperature- and radiation-driven impacts on marine ecosystems varies by intervention strategy and spatial scale. SAI induces global cooling and results in modest, diffuse reductions in solar input making its ecological relevance high at the global scale but relatively low locally (Figure 4a–c; Kravitz et al., 2012). These modest changes are expected to have only moderate effects on NPP and are unlikely to cause substantial ecological disruption. In contrast, MCB produces more spatially heterogeneous effects due to localized deployment and variable atmospheric transport, leading to greater ecological relevance at the regional scale. Local reductions in downwelling solar radiation from MCB could be on the order of 10% (Figure 4b), potentially leading to stronger impacts on net primary productivity and ecosystem structure in targeted regions.



Perturbation from Intervention		Impact on Marine Ecosystems	Importance of Impact to:		Model Inclusion	
			Stratospheric Aerosol Injection	Marine Cloud Brightening	Earth System Models	Marine Ecosystem Models
a. Decrease in atmospheric and ocean temperature	i.	**Global** change in ocean temperature drives shift in distribution, abundance, composition of NPP[1] and global ecosystems	High	High	✓✓	✓✓
	ii.	**Local** change in ocean temperature drives shift in distribution, abundance, composition of local NPP[1] and ecosystems where applied	Low	High	✓✓	✓✓
b. Decrease in total radiation	i.	**Global** reduction in photosynthetically available radiation drives reduction in photosynthesis and NPP[1]	High	Med	✓✓	✓✓
	ii.	**Local** decrease in coral bleaching for certain species, where applied	Low	High	✓✓	✓
	iii.	**Local** decrease in light could impair visual predators by reducing their hunting efficiency, where applied	Low	Med	✓✓	✓
c. Change in diurnal and seasonal solar radiation	i.	Altered diel and seasonal cycles in light could disrupt physiologically tuned life-histories of phytoplankton and HTLs	Low	Med	✓	✓
d. Change in downwelling UV	i.	Small and regionally variable impact on NPP[1] and corals depending on scattering of UVA and UVB by ozone and aerosols, respectively	Low	Low	✓	✓
e. Change in circulation patterns	i.	Regional changes in currents, temperature patterns, precipitation, evaporation, and salinity could alter thermohaline circulation and distribution of nutrients/heat, altering NPP[1] and ecosystem distributions	Med	High	✓✓	✓✓
f. Deposition of sulfur aerosols	i.	Acidification from sulfur aerosols has same impact as traditional ocean acidification on NPP[1] and some HTLs, but magnitude will be reduced as elevated pCO$_2$ (driven by lower pH) drives heightened CO$_2$ outgassing	Low	None	x	✓
g. Oceangoing Hardware	i.	Physical disturbances to habitat as a result of vessel traffic noise or pollution	None	Med	x	✓

✓✓ Commonly included/ Well represented

✓ Occasionally included/ Partially represented

✗ Rarely included/ Poorly represented

*Additional perturbations which remain too uncertain to assess are discussed in **Section 4.5**

[1] Changes in magnitude/distribution of NPP/temp/ community composition impact higher trophic levels

Relative Importance of Impact

High	Med	Low	None

Figure 4. The relative importance and representation of perturbations (**a-g**) and impacts (**i-iii**) for SRM on marine ecosystems. The relative importance of each impact pathway to each SRM method is qualitatively described as high (red), medium (light red), low (light blue), or not relevant (blue). Additional impacts which remain too uncertain to assess are discussed in Section 4.5. Relative importance represents an approximate, aggregate measure of the scale, uncertainty, and significance of a given impact. The general ability of Earth system (ESM) and marine ecosystem models (MEM) to simulate each impact pathway is summarized as a rough indication of inclusion across models used in prominent inter-comparison projects. MEMs are typically forced with temperature and plankton (net primary productivity or biomass) outputs from ESMs, and occasionally pH, photosynthetically active radiation (PAR) and/or other output (see **Section 5.3**). Model inclusion scores refer to 1) the ability to resolve changes in physics, chemistry and plankton in ESMs, and 2) the response of higher trophic levels to ESM forcing fields in MEMs. Higher trophic levels are considered inclusive of corals and visual predators, in that they are the responsibility of the MEM. Perturbations can impact higher trophic levels *directly* through their physiological sensitivity (resolved in MEM) to physical/chemical changes (resolved in ESM) or *indirectly* through their predation response (resolved in MEM) to physically/chemically driven changes in plankton (resolved in ESM). Inclusion is scored with **two checks (✓✓)** if most *direct* and *indirect* pathways are reasonably well represented in most models, **one check (✓)** if only some *direct* or *indirect* pathways are reasonably well represented in some models, and a **cross (x)** for rare and/or insufficient inclusion. If there is only a *direct* impact on higher trophic levels, ESMs are scored exclusively on their ability to resolve the relevant physics/chemistry (e.g., **b - ii, iii**). If there is only an *indirect* impact on higher trophic levels, ESMs must resolve the entire physical/chemical *and* plankton response to receive a check (e.g. **d - i**). MEMs are not penalized if the ESM does not resolve something that would otherwise be captured in the MEM forcing fields, meaning MEMs can receive checks even if the ESM does not. If there is a *direct* and *indirect* impact on higher trophic levels, ESMs and MEMs can receive a single check if they present a limited inclusion of either the *direct* or *indirect* impact pathway (e.g. **c - i**). These metrics are not intended to provide an exhaustive or specific review of all model configurations, but rather an informed, high-order assessment of model capabilities and processes that may need more development.

These reductions may also disrupt the behavior of visual predators by impairing their hunting efficiency. Although small relative to the natural diel cycle, the magnitude of local radiative changes required for MCB to achieve a meaningful global effect, given its limited deployment area, could be an order of magnitude greater than those of SAI (Figure 4d), with implications for seasonal productivity and broader ecological responses (Partanen et al., 2016; Lauvset et al., 2017).

Potential alterations in ocean circulation patterns (Figure 4e), and subsequent shifts in heat and nutrient distribution, are highly relevant to assessing the impacts of MCB on marine ecosystems. As MCB may produce uneven results regionally, changes in circulation may amplify or redistribute the intended cooling effects, introducing additional uncertainty in ecosystem responses. Alternatively, SAI induces more spatially uniform cooling than MCB, which may be less likely to result in major disruptions in ocean circulation patterns. Nevertheless, atmospheric and oceanic circulation changes are tightly coupled with large-scale and regional-scale impacts (Barreiro et al., 2008; Povea-Pérez et al., 2024). The effects of SAI and MCB on ocean circulation are highly uncertain and quantification has only been attempted by a handful of studies (e.g., Fasullo & Richter, 2023; Goddard et al., 2022). Based on current understanding and existing gaps in knowledge, relative importance scores of high for MCB and medium for SAI were assigned due to the potential for impacts to ocean circulation through broader climate feedbacks.

The potential deployment of SAI and MCB would each involve specific infrastructure and machinery, which may result in varying associated impacts to the marine environment. For SAI, a pathway for deployment would involve substantial investment in new high-altitude aircraft, with potentially thousands of missions annually from global bases to maintain target radiative forcing (Smith & Wagner, 2018). This operational scale increases the potential for interactions with the surface ocean chemistry and such impacts have not been studied. The long-term deployment of SAI may lead to sulfate aerosol deposition in surface waters, particularly in mid-latitudes due to stratospheric circulation (Kravitz et al., 2009; Visioni et al., 2020). Although sulfate deposition is not a driver of global changes in ocean acidification (Doney et al., 2007; Hunter et al., 2011), there might be a small likelihood of local impacts to ocean chemistry that could be worth exploring further (Figure 4f). For MCB, oceangoing hardware is central to the feasibility and risk profile, influencing everything from effectiveness to the environmental footprint (Figure 4g).

Certain additional impacts are not included in Figure 4, as current understanding is too limited to allow for a non-speculative assessment. For instance, uncertainties remain around the deposition and redistribution of sea salt from MCB deployment. While localized evaporation during seawater extraction may slightly enhance fluxes in some regions and lead to concentrated salt deposition elsewhere, the overall impact on ocean salinity is likely negligible, particularly given the oceanic origin of the salt. Even under industrial-scale deployment scenarios of order 100 Tg annually (Rasch et al., 2024), the expected relative salinity perturbation would be well under 0.1%. While the implementation of SAI and MCB would be characterized by significant uncertainty, it is almost certain to introduce impacts from increased aircraft and vessel traffic, including noise, pollution, and, depending on fuel source, elevated emissions of greenhouse gases and particulates that may impact marine ecosystems, especially sensitive pelagic species.

5. CURRENT LIMITATIONS IN MODELING MARINE ECOSYSTEM IMPACTS

Both global and regional scale modeling techniques allow us to assess the potential impacts of future climate scenarios and climate intervention on marine ecosystems, and to inform regional and global climate policy (Blanchard & Novaglio, 2024; IPCC, 2023). Marine ecosystem impact assessments are accomplished by simulating the physical and biogeochemical ocean state (including plankton productivity) using a global or regional ocean model, then using these simulation results to drive marine ecosystem models, which simulate higher trophic levels not resolved in the ocean models (Blanchard et al., 2024; Ortega-Cisneros et al., 2025; Stock et al., 2011; Tittensor et al., 2018). Marine ecosystem models are diverse in their intended purpose and complexity, and simulate ecosystem processes like growth, reproduction, competition, and predator-prey interactions, giving rise to emergent properties such as species distributions, food webs, fisheries production, and biodiversity (Heneghan et al., 2021; Murphy et al., 2025; Steenbeek et al., 2021). Because ecosystems depend heavily on temperature and food availability, the details of how well these drivers are represented, or if key biogeochemical or plankton dynamics are represented at all in the driver simulations, is critical for assessing model limitations and uncertainties.

5.1 Earth system model limitations

Earth system models (ESMs), global climate models that simulate and couple the atmosphere, ocean, land, cryosphere, and include ocean and land biogeochemistry, are used for global and regional impact assessments of climate change and climate intervention on marine ecosystems (Schoeman et al., 2023; Stock et al., 2011). ESMs were designed to capture climate dynamics and the carbon cycle, and as such have a relatively simplistic representation of lower trophic level marine food webs, focused on their role in biogeochemical cycling (Séférian et al., 2020; Stock et al. 2011). Substantial ocean biogeochemical model development is needed to capture many key processes that might be affected by global and regional SRM and marine CDR climate interventions (Figures 2 and 4).

Despite the goal of ESMs to capture carbon cycling processes in the ocean, there are wide variations across the ESMs in the representation of lower trophic level processes that drive ecosystems, such as NPP (Tagliabue et al., 2021) and carbon export production (Henson et al., 2022; Laufkötter et al., 2016; Planchat et al., 2023). This uncertainty is driven by both limitations in historical observations, which are more difficult in the ocean than on land, and major uncertainties in key lower trophic level processes. For example, both globally integrated historical levels of NPP and the future trends are highly uncertain across the ESMs contributing to CMIP, with no reduction in uncertainty from CMIP5 to CMIP6 (Kwiatkowski et al., 2020). One driver of NPP projection uncertainty is the parameterization of nitrogen fixation by diazotroph phytoplankton (Bopp et al., 2022), which constrains nutrient availability in “tropicalized,” high recycling biomes that are expected to expand under global warming. Similarly, another is the parameterization of temperature-dependent remineralization of organic matter in the mesopelagic (midwater) zone and how this recycling contributes to NPP in a warming climate (Rodgers et al., 2024). The typically assumed fixed C:N:P stoichiometry of phytoplankton in ESMs contributes additional uncertainty to projections of NPP, carbon export and zooplankton production in an increasingly stratified and nutrient-limited ocean (Kwiatkowski et al., 2019; Kwon et al., 2022; Teng et al., 2014). Another driver is the large variation in representation of zooplankton and their grazing, which drives uncertainty in higher trophic level productivity, and also carbon recycling and export both in the present and future (Clerc et al., 2024; Petrik et al., 2022; Rohr et al., 2023).

Besides issues of overly simple representations of phytoplankton and zooplankton, ocean biogeochemical models in general, and ESMs in particular due to simplifications made for computational efficiency, have further challenges in representing key ecological processes relevant for both SRM and CDR. These include the carbonate chemistry dependencies of plankton growth and calcification (Seifert et al., 2025), the interaction of zooplankton with anthropogenic particles (Fakhraee et al., 2023), and limited or absent representation of calcifying plankton (Planchat et al., 2023) and coral reefs (Bouttes et al., 2024; Mongin et al., 2021), which can be negatively impacted by acidification but could benefit from elevated alkalinity thereby highlighting the presence of competing drivers under future scenarios (Bach et al., 2015; Krumhardt et al., 2017, 2019; Kwiatkowski et al., 2025). Magnesium/calcium ratios affected by minerally sourced OAE are also rarely included in BGC models (but see Gangstø et al., 2008), yet may shift calcifier preference toward aragonite over calcite, as observed through geologic time (Bach et al., 2019; Davis et al., 2000; Ries, 2010). Other poorly represented processes relevant for CDR include benthic ecosystems, macroalgae and associated "passenger" organisms, microbial responses, and the impacts of trace metals and dissolved organic compounds. On a global scale, ESMs have neglected feedbacks from climate interventions on atmospheric CO_2 concentrations and terrestrial carbon uptake. The need for emissions-driven simulations is increasingly recognized (Sanderson et al., 2024), which capture these feedbacks and yield more realistic estimates of intervention outcomes for both marine and terrestrial ecosystems (Oschlies, 2009; Palmiéri & Yool, 2024; Lenton et al., 2019).

Another challenge in leveraging global ocean future projections for marine ecosystem impacts is the coarse resolution of global models. The bulk of marine ecosystem productivity is in coastal regions (Watson & Tidd, 2018). Most fish catch occurs over continental shelves where productivity is high and distance to shore is shorter, and thus the fuel needed to fish is minimized. However, the coarse resolution of ESMs (25-100km) means that these coastal regions of high productivity are poorly represented in terms of key physical processes like upwelling and river dynamics (Liu et al., 2021; Small et al., 2015), both providers of nutrients driving high coastal production. Despite these challenges, ESM development has been moving toward a focus on better marine ecosystem representation, with more flexibility on lower trophic level processes (Krumhardt et al., 2024; Long et al., 2021; Petrik et al., 2022; Stock et al., 2011). Current efforts in regional downscaling of future climate projections (Jacox et al., 2023; Pozo Buil et al., 2021) will allow for making climate projections usable for fisheries management and decision making, but are also hampered by computational demands and lack of observational data.

5.2 Regional ocean modeling limitations

While the applications of regional models for impact assessments and coastal management have expanded over time (Fennel et al., 2022), including in the context of climate intervention (Anschütz et al., 2025; Butenschön et al., 2021; Ou et al., 2025), several limitations still hinder their broad adoption for climate intervention applications (Buesseler et al., 2024; Ho et al., 2023). Many challenges are shared with global models, such as simplified representations of biogeochemical cycles and ecological processes as well as difficulties in bridging to higher trophic levels. Regional biogeochemical models, however, offer the advantage of resolving finer spatial scales than global models, down to kilometers or even hundreds of meters. These scales are needed to capture processes relevant to CDR, such as rapid subduction of surface waters by submesoscale currents (Wang et al., 2025). Nonetheless, their accurate representation remains challenging both theoretically (McWilliams, 2016) and because of the high computation costs, which can often rival those of global applications.

While regional models are able to capture local processes, activities such as monitoring, reporting, and verification (MRV) require connecting regional results to global scales, for example to address long-term durability of CDR and downstream biogeochemical impacts (Buesseler et al., 2024; Doney et al., 2024). While “downscaling” of large-scale simulations to regional models is the norm, the reverse, “upscaling” of regional model results to basin and global scales, e.g., by two-way coupling, has been used for physical processes (e.g., Combes & Matano, 2014; Schwarzkopf et al., 2019) but remains underexplored and challenging for biogeochemical processes (Harvey, 2000). Finally, advancement and validation of regional models for climate interventions is hindered by a lack of dedicated field studies as well as long-term time series data. However, a number of recent projects targeting regional interventions, such as the Reef Restoration and Adaptation Program in Australia, are designed to integrate field and modeling components, which could help improve these limitations.

5.3 Global Marine Ecosystem Model (MEM) limitations

Extensive development and intercomparison of global MEMs have provided critical insights into the effects of climate change and fishing on higher trophic levels, while also revealing important structural limitations, many of which are documented in Figures 2 and 4. No fewer than nine global marine ecosystem models have been developed to quantify and project the impacts of climate change and fishing in the open ocean (Tittensor et al., 2018, 2021). Their global scope means these models generally represent marine organisms by functional traits such as body size (Heneghan et al., 2021) or as distinct functional groups (Christensen et al., 2015), with little species or taxonomic resolution (but see Cheung et al., 2008 and Coll et al., 2020). Thus, these models are best-suited for global-scale assessment of climate impacts on higher trophic levels, but not for regional-scale or local studies where effects on individual species are of interest to fisheries managers and other regional stakeholders (Coll et al., 2024; Ortega-Cisneros et al., 2022).

Even in the open ocean, to our knowledge only five global models (DBEM, Cheung et al., 2008; DBPM, Blanchard et al., 2012; EcoOcean, Coll et al., 2020; FEISTY, Petrik et al., 2019; APECOSM, Maury, 2010) disaggregate marine biomass based on ocean depth (epipelagic versus mesopelagic; Maury et al., 2010) or pelagic and benthic zones (Blanchard et al., 2012; Cheung et al., 2013; Petrik et al., 2019), or across multiple depth zones (Coll et al., 2020; van Denderen et al., 2021). At the same time, the majority of global models aggregate consumer biomass across ocean layers based on body size or trophic level (Heneghan et al., 2021). This limits the ability of these models to provide information on CDR impacts on benthic communities present on the continental shelf, or mesopelagic and bathypelagic organisms in deeper waters.

Another shortcoming is that not all global MEMs explicitly consider species interactions (which account for carbon propagation through marine food webs) and species movement (which accounts for organic carbon transport). Even when ESMs resolve depth and functional differences in plankton biomass or NPP, this information is often lost to the MEM when forced (Figure 2a-iii, b-ii, g-iii; Figure 4b-ii). This, in turn, limits the ability of MEMs to mechanistically approximate where carbon may be absorbed, transported, and re-released by the living ecosystem under changing climates (Atwood et al., 2015; Roman & McCarthy, 2010). When they do, as in the case of EcoOcean, there is high uncertainty associated with dispersal capacity of organisms and how they adjust this dispersal to suitable or unsuitable environmental conditions (Coll et al., 2020). In coupled applications, MEMs are commonly forced by ESM output variables such as temperature and primary production (or plankton biomass), and occasionally also pH, light and other variables. Accordingly, ESM perturbations can affect MEMs either directly (through dependencies on physical or chemical conditions) or indirectly, by altering lower trophic productivity that propagates up through the food web. As a result, the ability of MEMs to simulate ecosystem responses is constrained not only by internal processes, but also by how accurately ESMs represent the physical and biogeochemical conditions that shape direct and indirect ecological pathways.

Finally, across the current suite of global marine ecosystem models, ESM temperature and lower trophic level variables such as primary production, plankton biomass and carbon export overwhelmingly drive projected climate change impacts on higher trophic level biomass (Heneghan et al., 2021), even for the few global marine models that include drivers beyond temperature and lower trophic level forcings, such as pH, oxygen or sea ice concentrations (Tittensor et al., 2021). Therefore, although all of these models can estimate the effects of temperature changes from SRM or other intervention strategies on fish biomass, most are unable to project any direct impacts of ocean acidification, deoxygenation or changes in nutrient concentrations on marine animal biomass.

5.4 Regional MEM limitations

Several ecosystem modeling frameworks exist to conduct simulations of climate change impacts on marine ecosystems at a regional scale (reviewed in Tittensor et al., 2018, Ortega-Cisneros et al., 2025). These models are built to answer questions relevant to the specific regions for which they are developed and have mostly focused on the impacts of fishing (Stock et al., 2023; Coll et al., 2024). Regional models have higher taxonomic resolution than global models, with several groups modeled as individual species or pooled into functional groups with a few species of similar life history and ecological characteristics (Kaplan et al., 2019; Koehn et al., 2016). Most regional MEMs represent benthic communities (focusing on target species, e.g., Eddy et al., 2017), but not often the ecosystem processes associated with benthic carbon dynamics. These models could thus be used for regional-scale assessments of climate intervention impacts on functional groups relevant to fisheries management and decision-making, but may need refinement to model impacts on groups such as benthic or zooplankton communities.

As for the global MEMs, the most common drivers used in regional climate applications are sea temperature, NPP, and plankton biomass (Ortega-Cisneros et al., 2025). In addition, regional MEMs have been extensively used to test impacts of ocean acidification (Marshall et al., 2017; Olsen et al., 2018; Zunino et al., 2021), deoxygenation (Morell et al., 2023; de Mutsert et al., 2016; Niiranen et al., 2013), and nutrient-driven changes (Bauer et al., 2018; Piroddi et al., 2021) on marine ecosystems and the services they provide. Hence, these models could be applied to represent the pathways by which different interventions affect marine organisms at the scale of the local ecosystem. There is limited capacity to generalize or extrapolate findings more broadly or to other regions. Regional MEMs often rely on assumptions or external data for processes occurring outside the region of interest, which can introduce inaccuracies and may not capture broader-scale interactions (Birkhofer et al., 2015). Therefore, for the purposes of evaluating impacts of climate intervention methods, especially CDR, interpreting regional MEM output requires familiarity with the strengths and limits of the projections.

6. ENGAGEMENT AND COLLABORATION ENABLE BETTER CLIMATE INTERVENTION RESEARCH

Results from climate models can provide information that is pertinent to a wide range of audiences, including policymakers, managers, and the general public. A substantial challenge in determining the ecosystem impacts of climate intervention is the discontinuity that exists between communities developing climate projections (the 'producers') and the 'users' of the modeling products, specifically those responsible for the creation and dissemination of impact analyses aimed at supporting policy development (e.g., Dilling & Lemos, 2011; Lemos et al., 2012; Schoeman et al., 2023). This gap in communication and understanding, amplified by differences in technical language, disciplinary norms, and risk perception, can lead to serious consequences when models are applied beyond their original intent (van den Hurk et al., 2018; Morrison, 2021).

Because models are simplified representations of complex systems, decisions regarding what to include, how to configure experiments, and how to interpret outputs are guided by the specific purposes of their developers (Morrison & Lawrence, 2023). When these upstream modeling decisions are misaligned with downstream user needs, there is a risk that model outputs may be inaccurate, misleading, or even harmful when used for applied purposes such as policy development (Harvard & Winsberg, 2022). This potential for misuse due to representational inadequacies in climate models has not gone unnoticed: Briley et al. (2021) assessed CMIP5 models to determine whether they had adequate representation of lake-atmosphere-land interactions for use in the Laurentian Great Lakes region, where these processes are significant drivers of local climate. The authors found that over half the models were inadequate in how they represented lakes, making them unfit for use to answer questions for the region. Lehner et al. (2019) evaluated the adequacy of runoff output fields in models, and concluded with a heavy warning to exercise caution when using the model output for certain purposes given discovery of instances where models produce the right answers for the wrong reasons. Similarly, Nissan et al. (2019) found that the metrics provided in modeling can fail to adequately capture the phenomenon that downstream users care about, and lead to the mischaracterization of risk.

In response to these challenges, many have recommended the adoption of co-production and co-design practices throughout the modeling process (Hewitt et al., 2021; Koren et al., 2022; Reed et al., 2023). These approaches aim to ensure that decisions about model structure, experimental design, and uncertainty communication are informed by diverse user perspectives, increasing the fitness of the models for their intended use and reducing the likelihood of misuse. Critically, this collaborative design process enhances the accessibility, relevance, and legitimacy of modeling products (Dilling & Lemos, 2011), especially in applied contexts like ecosystem impact assessments of climate intervention strategies.

Effective co-design requires early and sustained engagement between modelers and users, enabling knowledge gaps to be jointly identified, uncertainties to be holistically assessed, and modeling outcomes to be translated into actionable insights (Figure 5; Enquist et al., 2017; Sheppard et al., 2011). While many existing engagement efforts, such as regional climate modeling feedback in Germany (Huebener et al., 2017), stakeholder interviews in Australia (Wiseman et al., 2010), and visualization focus groups in Canada (Newell et al., 2021), have occurred after model development, the future of climate modeling will benefit from transforming the users into co-producers. This shift can foster shared ownership of the modeling process and strengthen the perceived usability, credibility, and legitimacy of resulting products for climate policy, management, and governance (Cash et al., 2003; Ahmed & Palermo, 2010).

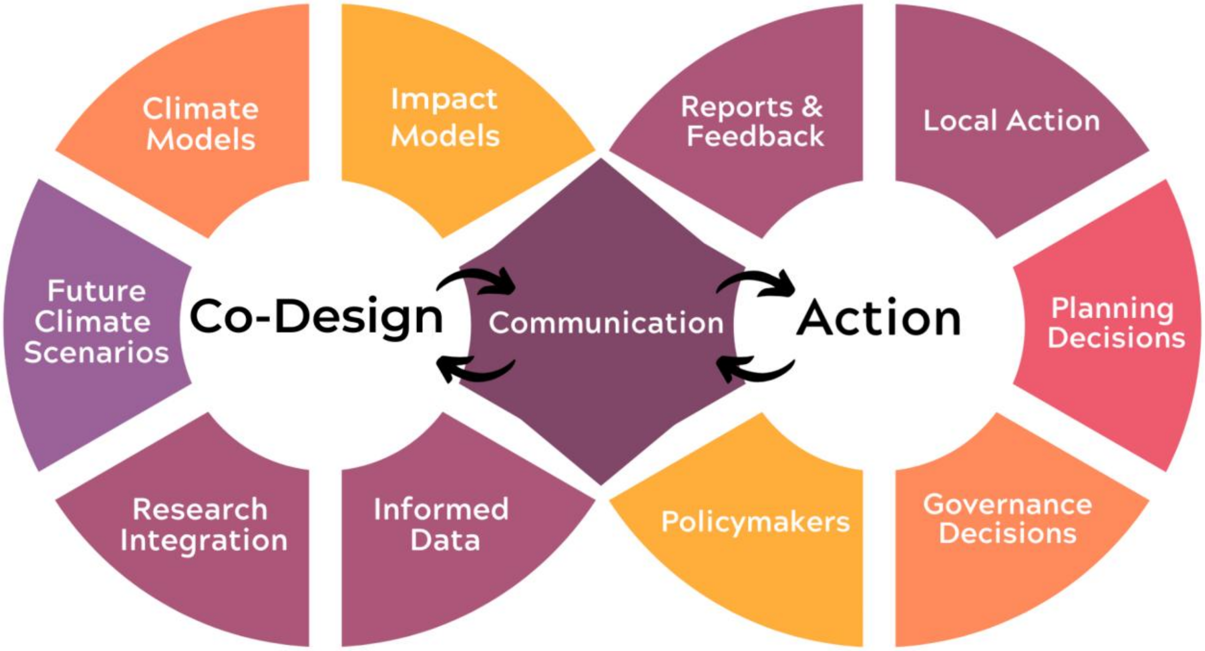


Figure 5. Integration of climate and impact models into policy, planning and governance decisions relies on a feedback process at all stages between the producers and the users. Through reporting, local actions, and planning sessions, the models become better informed for the producers and more applicable to the needs of the users. This feedback process is an iterative loop that relies on open, adaptable, and interdisciplinary communication to connect the co-design to the action.

7. DISCUSSION

For climate intervention and the world's oceans, the challenge the world faces moving forward is determining how to make decisions that balance sustainable fisheries, carbon sequestration potential, and the risks to marine ecosystems. These determinations should be made while more broadly balancing the likely impacts of an intervention, the risks and potential economic or ecological co-benefits, against the baseline global scenarios of more severe climate disruption. This challenge involves balancing non-monetary goals, such as preserving biodiversity, with maintaining ecosystem services and managing the costs of environmental damage. As CDR at climatically-relevant scales becomes increasingly needed to avoid surpassing warming thresholds, the critical question is how it can be implemented with minimal harm to ecosystems, ensuring that biodiversity, ecosystem services, and food security are not adversely affected. SRM deployment and governance face similar considerations including grappling with the potential to reduce climate risks versus the likelihood of introducing new risks (Felgenhauer et al., 2022). Addressing these challenges requires the involvement of experts from numerous fields including but not limited to economists, social scientists, and ecologists, as their expertise is crucial in navigating the many unknowns in this space, and transferring lessons learned from resource management to climate management. Effective interdisciplinary engagement will necessitate both strategic, long-term approaches and tactical, short-term actions in ecosystem management.

Even with current and proposed nature-based and technological solutions, CDR is only capturing a fraction of the carbon necessary to meet climate goals. Approximately 0.27-0.68 GtC per year of CDR is already occurring, but nearly all of this is from nature-based strategies with low (<100 yr) durability such as forest restoration rather than through novel technology (Smith et al., 2024).

Moreover, this estimate is subject to substantial uncertainties related to measurement methods, and recent analyses suggest that self-reported forest-based CDR may overestimate actual carbon removal due to issues with additionality (i.e., whether the carbon removal would have occurred without the intervention; Badgley et al., 2022; Probst et al., 2024). Limiting warming to 1.5°C with no overshoot necessitates the removal of anywhere between 420 and 1100 billion tons of CO_2 cumulatively from all possible sources, which would require between 200-500 years at current rates of CDR (Smith et al., 2024). However, this carbon removal target continues to rise due to limited decarbonization. Challenges to industrial-scale carbon capture and storage include high costs, land use requirements, lack of infrastructure and regulatory frameworks, among many others. Furthermore, only some forms of CDR can achieve long-term carbon sequestration, as the effectiveness of these strategies depends on the processes involved, including how and where the carbon is captured and stored, and the long-term stability of those storage methods (Siegel et al., 2021; Smith et al., 2024).

Despite already substantial impacts to ocean warming and the balance of the carbon cycle, the ocean remains an important lever for carbon sequestration. Approximately 2-3 billion tons of carbon released into the atmosphere per year are currently absorbed by the ocean, equal to about one third of present human emissions, with novel technologies aiming to increase this capacity (Friedlingstein et al., 2023). For example, kelp cultivation has been estimated to provide potential sequestration of as much as 0.03 GtC yr^{-1} (~0.3% of 2023 global emissions), which could require approximately 73,000 square kilometers in an optimal growth area to achieve (National Academies of Science, Engineering, and Medicine, 2022), while some anoxic basins may have the capacity to store several GtC in total (Raven et al., 2024). Designating areas for CDR activities presents an extensive list of considerations and tradeoffs, including interactions between ongoing climate change and a suite of climate interventions.

While terrestrial CDR projects have been undergoing testing and development for decades, marine CDR pilot projects are beginning to ramp up, with some efforts aiming to be fully operational within the next 5 years. For agencies to effectively evaluate proposals and permit applications for new CDR projects, transparency surrounding the expected positive or negative cumulative impacts is crucial. Shifts in community composition as a result of even local-scale disturbances could influence food webs (van Denderen et al., 2021; Krumhardt et al., 2022), carbon export (Boyd & Newton, 1995), and trophic transfer efficiencies (Boxhammer et al., 2018); it is important to understand if and how such shifts could propagate beyond their localized response. Additional key factors to assess are the spatial extent of the proposed project, expected duration, and whether the anticipated benefits justify the scale of intervention needed to make any significant dent in the global carbon balance. Multiple organizations have taken on the challenge of establishing accounting procedures and environmental safeguards, and reporting standards for CDR (AGU, 2023; Reinhard et al., 2023; Reykjavik Protocol, 2024), but there is still a critical need for a structured and globally accepted code of conduct to govern CDR research moving forward to ensure responsible, equitable, and transparent research practices (Loomis et al., 2022). Complementing such governance frameworks is the equally important task of developing robust monitoring, reporting, and verification (MRV) systems to quantify carbon removal as well as detect and attribute unintended ecological consequences across trophic levels. This review highlights potential ecosystem impact pathways and current modeling limitations, offering a basis for MRV standards that can connect observed ecological changes to specific climate interventions. Monitoring priorities could include shifts in key ecosystem drivers discussed throughout, such as plankton community composition, oxygen concentrations, or carbonate chemistry (e.g., aragonite saturation). Ultimately, future MRV efforts should emphasize flexible, iterative approaches capable of adapting to emergent ecosystem responses.

The landscape of research on climate intervention is currently fragmented across the communities investigating specific interventions. Individual interventions are generally investigated and understood in isolation (e.g., scenarios focus on the specifics of plausible deployment for MCB or OAE alone, not MCB and OAE together; Baur et al., 2022; Lockley et al., 2019). As a result, there is a large and potentially hazardous gap in our understanding related to the multitude of intersecting complexities in implementation, interactions between interventions, and potential conflicting mechanisms and impacts. Given that intervention schemes respond to different aspects of climate change, work on unique temporal and spatial scales, and will likely have different consequences, there will likely be motivation to adopt a combination of interventions to respond to climate change (MacMartin et al., 2018). Therefore, providing adequate and usable information appropriate for decision making on combined interventions and their synergistic impacts is critical. For example, implementing SRM and biotic CDR simultaneously will likely have compounding impacts as, for example, shifts in the intensity and spectrum of sunlight due to SRM could impact photosynthesis and carbon fixation in the surface ocean. The consequences of the interventions may be complex and interactive, and this complexity requires equivalent interdisciplinary research (Moustakis et al., 2025). Holistic evaluation of the interactive and complex landscape of impacts is essential for the development of sound climate intervention policy (Diamond et al., 2022).

The growing interest and investment in climate intervention research represents a potentially pivotal moment where organizational and policy frameworks transition to intentionally – as opposed to accidentally – modifying the climate. Lessons learned in ecosystem management could offer insights on how to accomplish climate management. The current climate transition is analogous to the paradigm shift in fisheries management after the collapse of major fisheries, such as Northwest Atlantic cod and Pacific salmon (Beamish, 2022; Myers et al., 1997), caused by overfishing and habitat loss. Fisheries management has since undergone significant changes to prevent future overexploitation and ensure the recovery of fish populations in some areas. Frameworks have also been developed to help navigate the complex processes of stakeholder engagement, balancing of competing interests, and safeguarding natural resources (e.g., Dobush et al., 2021; Mease et al., 2018; Pomeroy & Douvere, 2008). These strategies could be adapted to the climate sphere and expanded upon to embrace the higher-level complexity and expanded scope of the global climate crisis. For example, large-scale climate interventions like SAI have global implications, affecting nearly all aspects of human life, thus dramatically expanding the pool of stakeholders and issues that need to be considered. In contrast, interventions that focus their acute impacts in a local area or political jurisdiction, present a more contained set of challenges, though no less complex. Co-designed research into climate management strategies that engages CDR researchers, modelers, ecologists, and resource managers gives humanity its strongest path forward to protect marine ecosystems in the face of climate change.

8. CONCLUSIONS

Climate intervention strategies are currently being explored as potential tools to reduce the risks of climate-driven impacts, including extreme temperatures, biodiversity loss and threats to food and water security. While these strategies could help limit some of the most severe effects of climate change, they also carry substantial risks of disrupting regional climates, ecological processes, and biogeochemical cycles in marine environments, with uncertain consequences for fisheries and marine biodiversity. Critically, current models do not fully capture these complex ecological responses, and few studies have directly quantified the impacts of climate intervention on marine food webs or fisheries. This review comprehensively identifies the primary processes through which climate intervention could impact marine ecosystems, then qualitatively assesses each process with respect to its potential to perturb marine ecosystems and how well it is simulated in existing models. The identified overlap between high-impact processes and poor model representation provides a roadmap for determining which technologies have the most high-risk uncertainty and which research directions are most urgent. Moving forward, a combination of rigorous research, model development, transparent risk assessment, and inclusive governance are essential to fully understand the potential impacts of these interventions on both ecosystems and society.

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CONFLICT OF INTEREST

In addition to her primary role as UCSB faculty, MR serves as the Chief Science Officer for, and holds equity in, Carboniferous, Inc., a seed-funded startup company exploring potential applications of deep marine biomass sequestration; the work presented here was conducted and funded entirely via research grants at UCSB.

DATA AVAILABILITY STATEMENT

The climate model data used to generate Figure 3 are freely available from the Climate Data Gateway at NCAR (www.earthsystemgrid.org/) for updated SAI simulations (ARISE-SAI-1.5) and SSP2-4.5 (Richter & Visioni, 2022). Global coral reef boundaries displayed on Figure 3 are also freely available from the UN Environment Programme (UNEP-WCMC, WorldFish, World Resources Institute, & The Nature Conservancy, 2021) and can be accessed through the UNEP-WCMC GIS portal.

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