Ocean alkalinity enhancement: Insights and considerations for application

Authors
Steffen Swoboda[1,2]*, Sandy Avrutin[1], Michael Fuhr[3], Mirco Wölfelschneider[1,4], Andrew W. Dale[1], Andreas Oschlies[1]

[1]GEOMAR Helmholtz Centre for Ocean Research Kiel, Kiel, Germany
[2]Helmholtz Centre for Environmental Research (UFZ), Leipzig, Germany
[3]Yale Center for Natural Carbon Capture, New Haven, USA
[4]Leibniz Centre for Tropical Marine Research (ZMT), Bremen, Germany

* sswoboda@geomar.de

Abstract

Reaching net-zero emissions by 2050 requires both steep reductions in anthropogenic CO_2 and active carbon dioxide removal (CDR) to offset hard-to-abate sources, with CDR projected to account for roughly 15–20% of current greenhouse gas emissions. Ocean alkalinity enhancement (OAE), which accelerates mineral weathering to increase oceanic CO_2 uptake, is among the most promising marine CDR strategies, yet large-scale deployment remains largely conceptual and a consolidated implementation framework is lacking. Here, we develop a comprehensive OAE process chain structured around five themes - initial planning and considerations, preceding steps, deployment, monitoring, reporting and verification (MRV), and carbon uptake - systematically identifying the environmental, technical, and social considerations for mineral-based OAE across coastal, river, and open-ocean settings. Our synthesis shows that mineral dissolution, environmental interactions, and CO_2 uptake potential have received much attention through experiment and model studies. In contrast, critical gaps remain in techno-economic analyses covering mineral acquisition, transport and deployment logistics, and resource capacities, as well as in social and legal dimensions, including societal acceptance, regulatory frameworks, and MRV accountability. We conclude that OAE deployment as currently envisioned is likely unfeasible, founded on unrealistic assumptions regarding scale and logistics, though the fundamental CDR potential of OAE remains compelling. Large-scale application will require approaches that integrate into existing infrastructure as well as product-, service- and revenue-streams, diverging substantially from current paradigms. We identify the absence of field-scale experiments as the most critical bottleneck limiting the development and maturation of OAE.

1. Introduction

1.1. Negative emissions are a necessity to reach international climate goals

The repercussions of climate change and measures for its mitigation are among the most pressing challenges of modern society. In 2015, UNFCCC member states adopted the Paris Agreement, aiming to limit global average temperature rise to well below 2 ºC and to balance anthropogenic greenhouse gas sources and sinks. To reach this goal, greenhouse gas emissions, particularly CO_2, must be reduced to net-zero by 2050 (Shukla et al. 2022). While the bulk of reductions must come from sustainable land use and carbon-neutral economies, scenarios indicate that additional negative emissions, i.e. "carbon dioxide removal" (CDR), are necessary to balance hard-to-abate emissions from sectors such as steel, cement, and agriculture (Shukla et al. 2022). Negative emissions therefore

need to be scaled to ≥ 7–9 Gt CO_2/year by 2050, equivalent to approx. 15–20% of current annual greenhouse gas emissions (Smith et al. 2024).

To achieve this, atmospheric CO_2 must be removed and stored on timescales of centuries to millennia (Swoboda & Oschlies 2025). Numerous ecosystem- and technology-based CDR approaches in terrestrial and marine environments have been proposed and are under development or deployment (NASEM, Smith et al. 2024). However, current global CDR deployment covers only ~20% of the required amount, with novel technologies accounting for a mere ~0.1% of balanced emissions (Smith et al. 2024). It is therefore crucial that novel CDR approaches are developed and deployed safely and sustainably at the necessary speed and scale.

1.2. The carbonate system as a sink for anthropogenic CO_2

The ocean acts as one of the largest natural carbon sinks on Earth, taking up approximately 30% of annual CO_2 emissions (Friedlingstein et al. 2026). Net CO_2 influx is driven by higher atmospheric p CO_2 relative to surface waters. Upon dissolution, ~99% of CO_2 is converted to other dissolved inorganic carbon (DIC) species via the carbonate system (Zeebe & Wolf-Gladrow 2001), enabling seawater to store ~100 times more CO_2 than via gas dissolution alone. Therefore, the carbonate system of the global oceans is one of the most significant carbon sinks on earth.

The capacity of seawater to store CO_2 within the carbonate system is tied to seawater alkalinity (Dickson 1981, Wolf-Gladrow et al. 2007). Ocean alkalinity enhancement (OAE) aims to increase this storage capacity by adding alkaline minerals to seawater (Renforth & Henderson 2017). When CO_2 dissolves, it dissociates and releases H^+ ions:

(1) $$CO_2 + H_2O \leftrightarrow H_2CO_3 \leftrightarrow H^+ + HCO_3^- \leftrightarrow 2H^+ + CO_3^{2-}$$

The relative molar ratio of the dissociated CO_2 species within the carbonate system is approximately 1 (H_2CO_3) : 100 (HCO_3^-) : 10 (CO_3^{2-}) in natural seawater. Alkalinity represents the total proton acceptor capacity of seawater (Dickson 1981). Dissolving alkaline materials releases proton acceptors that neutralize H^+, creating a charge offset that drives further CO_2 uptake and dissociation. OAE is therefore considered one of the most promising CDR approaches.

In contrast, rivers contain far less alkalinity and variable DIC inputs (Da et al. 2025), making most river waters net CO_2 sources to the atmosphere or surface ocean (Friedlingstein et al. 2026; Middelburg et al. 2020). Nevertheless, rivers are the primary conduit for weathering-derived alkalinity to the ocean, delivering an estimated 26–36 Tmol/year (Middelburg et al. 2020; Da et al. 2025), and are therefore also being considered for OAE deployment, either by buffering excess DIC or transporting minerals to the coastal ocean.

Timely CDR implementation requires OAE to be tested at larger scales to verify its safety and sustainability. Currently, OAE research remains focused on underlying mechanisms and model-based efficiency projections (Lück et al. 2025), with only a few pilot projects underway (e.g. Project VESTA, Ocean Alk-align, Planetary), despite ambitions for Gt-scale removal within the coming decade.

As part of the "CDRatlas" research project, we developed a comprehensive, interdisciplinary process chain assessment for mineral-based OAE, i.e. a detailed flow diagram covering relevant steps and considerations for OAE implementation from start to finish. The process chain was developed through six consecutive expert workshops covering environmental, technical, logistical, societal, and legislative considerations. It encompasses coastal, open-ocean, and river-based mineral deployment, as well as reactor-based dissolution approaches, structured around the overarching themes of "initial planning and considerations", "preceding steps", "deployment", "monitoring, reporting and verification", and "carbon uptake and storage". An overview of the underlying steps is provided in Fig. 1. The detailed process chain can be viewed in supplementary figure S1.

In the following, we will outline in detail the necessary steps and considerations that were concluded during the process chain assessments for the safe and sustainable implementation of OAE and review the current knowledge on effects and processes induced by OAE. With this work, we aim to support initiatives and actors with a hands-on summary of the current scientific knowledge on OAE implementation on the basis of the developed OAE process chain and highlight key knowledge gaps that need to be addressed by the scientific community for the

safe and sustainable application of OAE.

2. Initial planning and considerations

The primary action for the implementation of OAE (and any given CDR-approach) is the assessment of the feasibility and desirability of the considered measure. For this, overarching questions that need to be addressed include whether a considered OAE approach is technologically and economically viable, environmentally safe, and socially acceptable. The outcome of each of these questions may vary considerably, depending on the chosen approach, the aimed scale of implementation and the considered location. These determine governing laws and regulation, conflicting assets, societal perception, environmental suitability, sensitivity and risks and available resources for implementation that need to be addressed. Further, following the initial confirmation that an OAE intervention is feasible and desirable, ongoing reevaluation is crucial to maintain its feasibility and desirability.

2.1. Locality as a key parameter for feasibility and desirability assessments

Apart from the chosen approach, arguably the most relevant consideration for the assessment of the feasibility and desirability of an OAE approach is the locality where it is implemented (fig 2). The choice of location will determine (1) the environmental suitability of the measure, including CO_2 uptake- and storage capacity, which in turn affects the environmental risk of the intervention, (2) governing law and regulations for the OAE approach depending on the jurisdiction of the location, which, to a degree, reflects the societal perception of the OAE measure and, (3) available resources, including required feedstock, infrastructure, power supply and skilled labor force, which may be in collaboration or competition with existing and future assets and revenue streams from industries and local communities. Based on the outcome of these three overarching dimensions, one can address the question whether an OAE measure at a specific location can be technologically viable, environmentally safe, economically viable and socially acceptable.

2.1.1. Environmental suitability

The environmental suitability of OAE at a given location is crucial, as the chemical and physical reactions and mechanisms of OAE as well as the prevailing physical and biogeochemical processes and ecosystem of the selected environment are non-negotiable and fundamentally determine whether OAE is technologically viable and environmentally safe in the region. The processes that impact the environmental suitability are covered in detail in this work and include the feedstock acquisition (section 3.1.); the capacity and means for alkalinity addition to sea- or freshwater (section 4.1. - 4.4., 4.7., 4.8.); the dissolution capacity of the feedstock (section 4.5. & 4.6.); the CO_2 uptake and storage potential (section 4.8.); as well as the impact on additionality of the intervention (4.9.). Depending on the chosen measure and scale of implementation, the physical and biogeochemical properties of the region will constrain the CDR potential and risks during and after implementation, which may impact the suitability of the OAE measure.

2.1.2. Societal perception, governing law & regulations

Appropriate regulatory frameworks and positive societal perception are essential for the safe, sustainable, and broadly accepted implementation of OAE. Although societal dimensions of OAE remain understudied (Lück et al. 2025), large-scale deployment will require alignment with public priorities. Historically, public acceptance has proven decisive for the success of emerging technologies, independent of their demonstrated safety or efficacy (Satterfield et al. 2023). The case of genetically modified crops illustrates this: despite scientific consensus on safety, environmental compatibility, and production efficiency, public resistance has been substantial. As

democracies are mandated to reflect public will, OAE legislation and its acceptance will ultimately be shaped by perceived risks and benefits (Satterfield et al. 2023).

Opinion formation on emerging technologies commonly reflects three dimensions: (1) attributes of the technology itself, (2) the values and ethical positioning of those evaluating it, and (3) the state of knowledge, management, and regulation (Satterfield et al. 2023). Key concerns specific to OAE include environmental impacts; the perception of the ocean as a pristine and fragile system; coastal industrialization; feedstock sourcing through mining; material dispersal and its association with ocean dumping; resource burdens; preferred integration with existing infrastructure and community-inclusive business models; and considerations of justice (including public oversight, benefit-burden distribution, and actor ethics) (Satterfield et al. 2023; Nawaz & Belotti 2025).

Currently, no OAE-specific legislation exists, and regulatory approaches rely on interpretations of existing environmental law. Key determinants of applicability include deployment location (inshore vs. offshore), materials and technology, environmental thresholds, and seafloor utilization (Webb et al. 2021; NASEM). At the international level, the most relevant frameworks are the 1972 London Convention, the 1996 London Protocol, and the 1982 UNCLOS, which have formed the basis of recent ocean-based CDR regulatory efforts (Webb et al. 2021; Webb 2024). However, these agreements permit conflicting interpretations with respect to OAE (personal correspondence A. Proelß), and there is no international consensus on whether OAE should currently be deployed, as evidenced by the London Convention and Protocol being invoked to restrict OAE, despite the Paris Agreement encouraging its use (Webb 2024).

Alongside deployment regulation, carbon crediting is an equally unresolved regulatory domain. In principle, removals may be credited through compliance carbon markets tied to agreements such as the Kyoto Protocol or the Paris Agreement, or through voluntary carbon markets (VCMs), which are privately organized and not currently eligible for fulfilling Nationally Determined Contributions (NDCs). Up until now, carbon credits sold through OAE projects are sold via the VCM, and no official international crediting framework currently acknowledges carbon removal via OAE.

Recent UNFCCC progress on Article 6.2 of the Paris Agreement outlines how carbon removals may be accounted for and traded as Internationally Transferred Mitigation Outcomes (ITMOs), potentially enabling VCM credits to contribute to NDCs. Under this framework, participating countries must designate a national authority to authorize ITMOs and register removals in the UNFCCC or an approved national or third-party registry (Florence School of Regulation, 2025). Nevertheless, the specific rules, modalities, and procedures under Article 6 remain under negotiation. Additionally, national implementation, including authority designation, registry establishment, and MRV protocol development, will require substantial time (personal correspondence A. Proelß). Until these mechanisms are in place, OAE-based CDR will likely remain within the VCM and thus outside NDC accounting.

Finally, the rule of law constitutes a critical cross-cutting consideration. Weak governance has been shown to enable injustices that prioritize economic interests over local livelihoods, particularly when projects are executed in the Global South under investment and management from the Global North (Lyons & Westoby 2014). Notwithstanding UNFCCC efforts to address such inequities, this dimension warrants careful attention in OAE implementation, both for ethical reasons and for its implications for societal acceptance.

2.1.3. Available resources

The viability of OAE depends on available resources. For mineral-based OAE, prerequisites include suitable alkaline feedstock from quarries or industrial by-products (section 3.1.); processing, transport, and storage infrastructure (section 3.2. & 3.4.); and sufficient energy supply (section 3.5.).

Large-scale OAE will require a trained workforce to implement OAE, conduct MRV, and address challenges arising during commercial deployment. Currently, OAE remains largely rooted in academic research, with emerging industry closely tied to academic institutions. Consequently, relevant expertise is geographically concentrated in countries with active OAE research programs. When following this narrative, published OAE literature may thus serve as an indicator for workforce availability and thereby, regional capacity to implement OAE: according to an overview study on published literature on CDR by Lück et al. (2025), CDR research is dominated by China and OECD countries, with South American and African contributions amounting to only ~6% of publications. Moreover, only 152 OAE-specific studies (0.52% of all CDR literature) have been published up to 2022. Combined with the need of suitable environmental conditions and resources for implementation, the

relatively small amount of publications on OAE and the geographic on China and the OECD may indicate a regional lack of skilled labor force that may further constrain options to perform OAE across nations and continents, despite otherwise favorable conditions. This underscores the importance for collaboration and knowledge exchange towards nations with less capacity to engage in OAE research to assure future implementation remains just.

Competition for limited space may represent an additional implementation constraint, particularly for coastal and riverine approaches. In particular, this may include other activities along shorelines and shallow waters, including recreational or touristic activities; shipping; infrastructure such as harbours or coastal protection; areas with restricted access such as military territories; protected areas and nature reserves; and commercial use, including sand, gravel and phosphate extraction sites or aquaculture sites. In contrast, the limited competition for spatial use in open ocean environments has been identified as a potential advantage for offshore OAE approaches (NASEM). While the implementation of OAE and the presence of other activities or access restrictions do not have to be mutually exclusive, identifying suitable large-scale sites that satisfy both regulatory and environmental criteria, particularly in coastal zones, may prove a significant constraint.

Resource and space requirements for OAE may also create tensions with existing industries and local communities.

For illustration, as outlined in detail in section 4.4, an example of reactor-based OAE site with a capacity of 0.1 Mt CO_2 removal per year would require a minimum of 38 truckloads (25 t per load) of CaCO3 and 472 million liters of water per day which over the course of a year would amount to 3.3% of the annual water usage by the German manufacturing sector (Statistisches Bundesamt, 2025). Such demands for OAE implementation must be balanced against existing local resource use. Conversely, the resource intensity of OAE may create opportunities for mutually beneficial collaboration with other industries, for example, through shared infrastructure or guaranteed supply demand, potentially improving operational efficiency and generating broader local economic co-benefits.

3. Preceding steps

3.1. Mineral mining and processing

3.1.1. Considered mineral types and deposits

Mineral types commonly considered for OAE include the silicate-based mineral olivine; carbonate-based minerals; oxides; hydroxides and alkaline industrial by-products such as steel slag; among a broader range of candidates (table 1; Renforth et al. 2017). Compilations of mineral deposits have been gathered for olivine and limestone (Caserini et al. 2022; Geerts et al. 2025).

A key factor in mineral selection is contamination by toxic heavy metals (e.g., nickel, lead, cadmium) and hazardous fibers (asbestos), which pose environmental and health risks during and after deployment (Bach et al. 2019; Flipkens et al. 2023; NASEM). Deployment must not exceed environmental thresholds for such components, which may restrict the use of highly contaminated sources or application in areas with elevated baseline levels (Eisaman et al. 2023; see section 4.7.).

3.1.2. Mining facilities

For extensive OAE implementation, mining facility capacity must be evaluated in terms of resources and reserves. ‘Resources’ encompasses the overall extraction capability, including availability of power and water supply for waste treatment and recycling, and become relevant at the intended scale of application. ‘Reserves’ refers to the accessible and permitted mineral stocks exploitable under profitable conditions, determining the timescale over which a facility can supply OAE.

Mining operations carry potential for widespread social and environmental repercussions, requiring attention to sustainability; occupational health and safety standards; and environmental risks including pollution, sedimentation and erosion, land subsidence, dust contamination, and ground vibrations from blasting. Interlinked social considerations include protection of recreational land and human habitats; food security; and health of employees and proximate communities. Transparent communication about mining operations and their implications is furthermore crucial to prevent public backlash that could hinder OAE deployment.

3.2. Processing facilities

3.2.1. Crushing and grinding

After mining, minerals must be processed depending on the mineral type and OAE approach chosen. A key step for all mineral-based OAE involves comminution of rocks to small grains via crushing or grinding, with required particle sizes ranging from centimeter-scale rocks to ~10 µm particles depending on the approach and deployment site conditions (e.g. Skousen et al. 2017; Köhler et al. 2013). Crushing can commonly achieve particle sizes down to ~25 mm, with finer sizes achievable through an additional grinding step (Wills & Napir-Munn 2006; see section 4.6.). Since comminution produces a broad spectrum of grain sizes (Wills & Napir-Munn 2006), alternative uses for unsuitable fractions should be considered to reduce waste and balance costs and emissions. In contrast, existing product streams that produce suitable grain size fractions as side-products may be a cost efficient feedstock source. Comminution efficiency is largely determined by mineral hardness, reflected in the grindability, i.e. abrasion or work index. A higher abrasion index increases energy demand and mill wear, affecting cost efficiency and resource use. The abrasion index may also vary with grain size, influencing the efficiency of comminution to finer fractions (Wills & Napir-Munn 2006).

3.2.2. Calcination and hydration

In cases where the dissolution kinetics of limestone are not fast enough for a selected OAE-approach, limestone (calcium carbonate; $CaCO_3$) may be broken down during a calcination process to form quicklime (calcium oxide; CaO) under the release of CO_2 (i.e. calcination). Hereby, approx. 785 g of CO_2 of chemical process emissions are released for the production of 1 kg quicklime. As a result, due to the production process, quicklime has a lower efficiency for CO_2 removal than limestone:

$$CaCO_3 \rightarrow CaO + CO_2$$

For the calcination process, the limestone is heated to approx.. 900 – 1050 C, thus marking an additional major source for CO_2 if not performed with CO_2 neutral energy sources (Eisaman et al. 2023). To reduce emissions from the limestone conversion and energy supply, the CO_2 may be captured and stored in long-lasting carbon-based products or storage sites, thus holding potential to improve the efficiency of quicklime-based approaches. After the production of quicklime, an alternative further hydration step may be conducted to form calcium hydroxide ($Ca(OH)_2$):

$$CaO + H_2O \rightarrow Ca(OH)_2$$

This may be advantageous for some OAE applications, as calcium oxide can irritate eyes and skin upon contact or cause lung irritations when inhaled. Further, calcium oxide hydration is an exothermic reaction with a low-grade heat release which may be recovered for industrial processes (Foteinis et al. 2022). However, hydration decreases the density from 3.34 g/cm^3 (quicklime) to 2.21 g/cm^3 (calcium hydroxide) which requires a larger volume for transport and thus may lower its efficacy for OAE applications (Foteinis et al. 2022).

3.3. Alkaline minerals from industrial byproducts

Industrial alkaline byproducts represent a potential alternative feedstock for OAE. Steel slag, a solid mineral-like byproduct of steel purification, has received particular attention in this context (Fisher & Barron 2019; Moras et al. 2024; Guo et al. 2024; Nakamura et al. 2023). Its composition varies between batches but primarily includes silicates (e.g. aluminum silicates, calcium aluminum silicates), iron oxides, and trace metals (Fisher & Barron 2019). As with mined minerals, the presence of toxic trace and heavy metals poses potential environmental and health risks during OAE application (see section 4.7). A practical advantage of steel slag is the existence of well-established handling and processing procedures, which can be tailored to specific OAE approaches. For instance, rapid cooling via water quenching or air granulation yields highly porous granules with large surface areas favorable for dissolution (Tossavainen et al. 2007; Lindvall et al. 2019).

A further alternative to raw minerals may be cement kiln dust or lime kiln dust which form as a waste product during the calcination processes within clinker production in the cement industry and quicklime production in the limestone industry (Flipkens et al. 2025, see details on the calcination process in section 3.2.2.). Both materials contain high amounts of quicklime (CaO) and hydrated quicklime (Ca(OH)2), while being already comminuted to particle sizes < 100 µm and being readily available as part of existing industry streams, thus holding large potential as feedstock for OAE. Experiments show that both materials introduce alkalinity when added to water. However, cement kiln dust in particular contained trace metals and impurities that may cause environmental harm if concentrations exceed suitable limits. Further, upon dissolution, a large fraction of ˜70-75 wt% of the added kiln dust remained as unreacted phases. While the major proportion of this unreacted phase consists of Ca(OH)2, which would add additional alkalinity when dissolved, further research is required to assess the fate of such unreacted phases from kiln dust as OAE feedstock (Flipkens et al. 2025).

3.4. Transport and logistics

Given the high density of the minerals considered and the necessity for fine comminution, transport and logistics require careful consideration to minimize emissions and maximize net carbon sequestration. Key considerations are the transport and storage of minerals, both of which vary depending on the state of processing, i.e. rocks and boulders versus fine-grained material. Insights on transport and logistics were by employees at Sibelco.

Within the mining facility, internal transport involves excavators and haul trucks for flexible haulage, and conveyor belts, bucket elevators, or pneumatic systems for processing-related transport. CO_2-neutral internal infrastructure is advantageous for overall OAE efficiency. Transport from mining to comminution facilities is most efficient when both are co-located, enabling belt-drive systems, whereas separate facilities require additional infrastructure (roads, rail) and greater maintenance.

For long-distance transport to distribution centers or deployment sites, rail and ship-based transport are preferred options due to their large carrying capacities and low CO_2 emission per ton cargo. However, storage and loading infrastructure at railways or harbours may pose limiting factors: packaged material requires cranes and storage areas, while bulk material necessitates silos and pneumatic loading facilities. Bulk carriers designed for comminuted material are required for ship transport, as finely ground material behaves near-fluidly under wave motion. Road transport via truck bulk carriers offers flexibility but is unsuitable for long-distance or large-scale applications due to limited carrying capacity (~25 t/truck in Europe, ≤10 m^3/load; Žnidarič 2015) and high costs and emissions per ton. Deployment sites must likewise provide infrastructure for handling and storage of alkaline minerals.

Overall, the combination of logistical requirements may pose considerable limitations regarding mineral sourcing and site selection for OAE applications.

3.5. CO_2-neutral power supply

A key consideration for the application of OAE, as for any CDR approach, is the power source used for the implementation. Emissions that are generated during power generation have to be accounted for when budgeting the net carbon removal of the OAE measure. Therefore, power sources with low or no emissions will enhance the

net CO_2 removal and thereby increase the efficiency of the OAE measure. While from a climate perspective any net CO_2 reduction may be considered a success, from an economic perspective, emissions-intensive power sources reduce both net removal and the associated carbon credit revenue. Renewable energy sources are therefore preferable, irrespective of whether net removal can still be achieved with non-renewable alternatives.
When it comes to the availability of power sources, particular attention should be given towards the location of the power sources; the capacity and intermittency at which power can be supplied; as well as the expected runtime over which the power source will be operating.

4. Deployment

For OAE deployment, the alkalinity feedstock must be introduced into seawater. Mineral introduction options include the addition of minerals on beaches, coastal areas, the open ocean or river systems. Minerals may be introduced directly into the environment or dissolved beforehand within reactor-based systems and introduced as alkalized liquid.

4.1. Coastal mineral introduction

A proposed OAE method involves spreading crushed minerals in high-energy coastal environments, including the intertidal zone and shallow coastal areas influenced by waves, tides, and currents (Meysman & Montserrat 2017, Eisaman et al. 2023). Site selection and deployment for this approach require balancing several competing factors related to grain size, geomorphology, and hydrodynamics. Regarding grain size, a key advantage of this approach is that minerals can be applied at relatively large sizes (millimeter range), as water movement and wave action enhance their reactivity and dissolution (Meysman & Montserrat 2017, Flipkens et al. 2023, see section 4.5.3. & 4.5.4.), thereby reducing comminution costs and energy requirements (see section 3.2.1.). However, the grain size composition of added sediments should reflect local conditions to maintain the physical integrity of the nourished environment (Elko et al. 2021). While larger grains are generally preferred, blending with finer material may thus be necessary, potentially increasing comminution costs. Moreover, high proportions of fine grains can restrict pore water flow and limit mineral dissolution rates, reducing OAE efficacy (Geerts et al. 2025, see section 4.5.3.). It should also be noted that introduced minerals the higher density compared to prevailing coastal minerals (e.g. quartz sand: ~2.6 g/cm^3 compared to Olivine: ~3.2 g/cm^3) may be buried preferentially due to density-specific gravitational sorting or distributed differently within the hydrodynamic setting at a site, potentially lowering dissolution rates and thereby OAE potential (Garzanti & Andò 2019, Fuhr et al. 2022; see section 4.5.). Regarding geomorphology, coastal conditions must allow mineral introduction without causing significant erosion or sediment displacement, ensuring dissolution occurs in shallow waters conducive to atmospheric CO_2 uptake (see section 4.8.4.). Since wave action promotes dissolution but excessive hydrodynamic energy causes rapid mineral dispersion, sites with moderate energy regimes are preferable, providing an efficient trade-off between sediment retention and dissolution enhancement (Elko et al. 2021).
Deployment strategies differ fundamentally between intertidal and subtidal environments, and are therefore discussed separately below. For intertidal deployment, established beach nourishment techniques could be adapted, such as pumping mineral slurries via temporary pipelines and redistributing them with bulldozers (Hannides et al. 2019). Beyond this, distribution methods remain largely conceptual, including the use of construction machinery such as bulldozers or wheel loaders (Hangx & Spiers 2009). The large-scale intertidal deployment of OAE demands flexible logistics and infrastructure capable of moving large mineral quantities across extensive coastlines. Considering a density of ~ 3.2 tons m^{-3} in the case of the olivin endmember Forsterit (Mg_2SiO_4), a 5 cm layer of olivine already amounts to 16 tonnes per 100 m^2 and 160000 tonnes per 1 km^2. Assuming a usable intertidal width of 50 m, spreading across 1 km^2 would require a coastal stretch of 20 km. Large-scale intertidal spreading thus demands flexible logistics and infrastructure capable of moving large mineral quantities across extensive coastlines.
For shallow subtidal environments, ship-based distribution is considered the most viable approach (Dale et al. 2024, Fuhr et al. 2025), for which vessels such as split-hull scows, open hopper barges or flat-deck barges could

be repurposed for mineral spreading. Given the large quantities involved, cost efficiency is a central concern. Mineral spreading may therefore be integrated with existing coastal management practices to reduce costs and generate additional co-benefits. Relevant examples include beach nourishment, where sandy beaches are restored or maintained to compensate for natural erosion (Foteinis et al. 2023, Elko et al. 2021), and harbour or waterway maintenance dredging (Fred Lee & Jones-Lee 2000).

Across all deployment contexts, intertidal and subtidal alike, careful assessment of local benthic habitats is essential to prevent adverse impacts on benthic and infaunal communities through burial and increased turbidity (Elko et al. 2021). Best management practices from beach nourishment can serve as a framework for the sustainable deployment of alkaline minerals in coastal environments (Rosov et al. 2016).

4.2. Deep ocean mineral introduction

Open ocean dispersal of alkaline minerals requires particles to remain in the surface layer until fully dissolved, ensuring that the resulting alkalinity increase occurs within waters that exchange with the atmosphere (see section 4.8.4.). Consequently, very small particle sizes are necessary to minimize sinking velocities (Köhler et al. 2012, Moras et al. 2022, Moras et al. 2024b). A further complication arises from drag and entrainment effects between particles settling in close proximity, which can significantly increase their collective settling velocity (Guo et al. 2024b). Minerals must therefore be evenly dispersed in surface waters, as bulk dumping will result in rapid settling.

To date, no field studies have assessed the settling behavior and dissolution of fine alkaline mineral particles in the open ocean. Model studies on olivine suggest that particles of ~1 µm would achieve ~80% dissolution in the surface layer over one year, whereas increasing particle size to 10 µm reduces this fraction to ~5%, indicating olivine to be unsuitable for open ocean dispersal (Köhler et al. 2012). In contrast, magnesium hydroxide appears more promising, since experimental studies show that particles smaller than 63 µm dissolve within ~1 hour, and those larger than 180 µm within ~7 hours (Moras et al. 2022, Moras et al. 2024b). As dissolution reduces particle size and thus settling velocity over time, the rapid dissolution of magnesium hydroxide may potentially allow even larger particles to dissolve within the surface layer despite initially too high settling velocities.

Alkaline mineral particles can interact with dissolved and particulate organic matter, particularly marine snow, in the surface ocean. Attachment to marine snow can lead to ballasting and enhanced export of organic carbon to deeper water layers (Iversen & Robert 2015, van der Jagt et al. 2019, Swoboda et al. 2024), potentially enhancing carbon export via the biological carbon pump (see section 4.9.1.). Conversely, the lower density of marine snow relative to mineral particles may reduce particle settling velocities upon attachment, thereby extending the time available for surface dissolution and benefiting OAE efficiency. However, studies on gypsum ballasting demonstrate that transparent exopolymeric particles (TEP), polysaccharide exudates produced by phytoplankton, can coat mineral particles and inhibit dissolution, leading to their persistence throughout the water column and eventual deposition on the seafloor (Wollenburg et al. 2018, Wollenburg et al. 2020, Swoboda et al. 2024). Interactions between alkaline minerals and marine snow and TEP may therefore either enhance or impede open ocean OAE efficiency, underscoring the need for dedicated research into such processes.

4.3. River-based OAE

River-based OAE refers to measures that utilize the water of rivers as a medium to add alkalinity and thereby drive net atmospheric CO_2 reduction through estuaries or the coastal ocean, while riverine alkalinity enhancement (RAE) refers to measures that reduce net atmospheric CO_2 via the river itself. The addition of alkaline minerals to rivers, in particular limestone, is a common method that has been used for decades to buffer pH increases or counterbalance contamination from industry, such as acid rain or acid mine drainage (Skousen et al. 2017, Svenson et al. 1995). To date however, the introduction of alkaline minerals to rivers for the purpose of CDR has received little attention, yet recent studies indicate considerable potential of the approach (e.g. Ou et al. 2025, Tian et al. 2025). By combining observations of liming practices to balance river acidification, observations on carbon dynamics within rivers and estuaries as well as observations of alkalinity input from rivers in coastal oceans, implications on the needs and suitability of RAE and river-based OAE can be outlined.

4.3.1. Methods and infrastructure requirements for river-based OAE

Methods for alkalinity introduction into rivers may stem from reactor-based alkalinity dissolution techniques (see section 4.4. on details of reactor-based OAE) or through passive treatment techniques which may be adopted from geochemical river alkalinization techniques for acid buffering (Skousen et al. 2017). A selection of established passive treatment techniques for acid buffering aims at channeling river water through alkaline rocks, thereby enhancing the dissolution rates through persistent water flow and/or extended exposure time (see section 4.5.3. & 4.5.4. on details of alkaline mineral dissolution). This may comprise the bulk addition of limestone sand into streams; coarsely ground rocks into created drains or leach beds; or adding larger rocks and boulders into streams and rivers (Skousen et al. 2017). Alternative approaches for the addition of limestone, in particular for the buffering of acidified rivers and lakes via acid rain, is the use of lime dosers or diversion wells which offer a more controlled addition of alkalinity. Lime dosers are typically placed along rivers or lakes and consist of a vertical cylinder for the storage of finely ground limestone, and an outlet that allows the continuous addition of a defined dosage of limestone to the water (e.g. Svenson et al. 1995). In the case of diversion wells, crushed limestone is added to a vertical storage cylinder, in which river water is fed to the bottom of the cylinder and is forced to rise through the crushed limestone and exit as alkalized water from an outlet on top of the cylinder back into the river (Skousen et al. 2017, Arnold 1991). While the addition of alkaline minerals for acid treatment is well established, a potential constraint for its use for RAE and river-based OAE is that treated lakes, streams and rivers are acidic, thus enhancing the dissolution rate of the added alkaline minerals (see section 4.5.2. on mineral dissolution and the effect of pH on dissolution kinetics). Therefore, pH neutral freshwater sources may limit the potential of established acid treatment approaches for RAE and river-based OAE, by limited dissolution.

With regards to infrastructural requirements on the implementation of RAE and river-based OAE, as for any mineral-based OAE approach, the transport, handling and storage of alkaline minerals are crucial factors to consider (see section 3.4. on details regarding OAE logistics). For details on requirements of reactor-based mineral deployment, see section 4.4.. With regards to passive methods where alkaline minerals are added into the natural environment, the initial transport and construction is required, with little maintenance requirements afterwards, making them particularly cost effective (Svenson et al. 1994, Skousen et al. 2017). On the other hand, diversion wells and lime dosers require fast running waters and close location to infrastructure for refilling, e.g.. roads, supervision and service (Svenson et al. 1994). Overall, RAE and river-based OAE may hold large potential with regards to efficient logistics of mineral-based OAE methods, due to the abundance of rivers and the density of available infrastructure. Therefore, the likelihood that a suitable river for the deployment of OAE is in close proximity to mining and processing facilities is presumably high, though it should be noted that some rivers naturally contain high alkalinity, making them less suitable for mineral addition due to the risk of secondary precipitation (Tian et al. 2025). Further, shipping networks along rivers are commonly well established, holding large potential for efficient and dedicated logistics and deployment solutions that may lower costs and emissions of RAE and river-based OAE beyond traditional liming approaches.

4.3.2. CO_2 removal in rivers and estuaries via riverine alkalinity enhancement and river-based OAE

While acidified freshwater sources are abundant, it is important to differentiate the source of acidification, and whether it is caused by natural processes or anthropogenic actions. In the context of anthropogenic freshwater acidification, RAE and river-based OAE qualifies as CDR only if it delivers a net reduction in atmospheric CO_2, which requires accounting for both the emissions from the alkalinity addition itself and those from the original pollution source, as the two are directly linked. If this condition is not met, such measures constitute emission avoidance rather than removal (Swoboda & Oschlies 2025). Given the low likelihood of a negative net CO_2 balance, and the risk of incentivizing continued freshwater pollution, applying river-based OAE to counteract anthropogenic acidification is ethically and environmentally questionable while offering low CDR potential and is therefore not considered further here.

Natural freshwater acidification commonly occurs when organic matter is released into freshwater and respired by microbes, thus enhancing pCO_2 within the freshwater and lowering its pH (e.g. Korsman 1999, Petrin et al.

2008, Richey et al. 2002, Abril et al. 2014). In cases where waters naturally contain high amounts of organic matter, e.g. peatlands, wetlands or areas with periodic heavy rainfalls (e.g. monsoon impacted rivers) or floodings, the organic respiration may lead to elevated levels of pCO_2 and substantial CO_2 outgassing (Halsey et al. 1996, Yao et al. 2007, Richey et al 2002, Abril et al. 2014, Butman & Raymond 2011). For instance, measurements of pCO_2 at the Xijiang River basin during wet season showed average values of up to 6300 µatm, and estimates for CO_2 outgassing at the lower ranges of the Xijiang River ranged from 31-58 tons CO_2 ha^{-1} $year^{-1}$ (Yao et al. 2007). In the amazon basin, average pCO_2 values of the main stream where 4350 +- 1900 µatm $year^{-1}$ and it was estimated that CO_2 outgassing of the entire amazon basin comprised 1.8 Gt CO_2 $year^{-1}$ (Richey et al. 2002). In such cases of largely enhanced pCO_2 values within freshwater bodies, approaches for RAE may hold potential to buffer acidification and thereby reduce riverine CO_2 outgassing.

Despite such alleged potential for reduced CO_2 emissions via the buffering of naturally acidified waters, it has to be considered that within these areas, acidified waters are an integral part of the local ecosystems (Petrin et al. 2008, Renberg et al. 1993). The introduction of alkalinity, in particular liming practices due to acid rain, have been observed to negatively impact local ecosystems by altering species composition and abundance and inducing harmful algal blooms (Angeler et al. 2017, Kaushal et al. 2013, Mant et al. 2013). Thus, the application of RAE in naturally acidic waters for the reduction of riverine CO_2 outgassing should consider potential environmental repercussions, for which observations on the effects of liming may serve as a guideline to identify limitations.

In addition to riverine CO_2 outgassing, naturally acidified rivers drive considerable CO_2 outgassing within estuaries upon discharge, with global estimates of ~0.27 Pg C yr^{-1}, corresponding to ~19% of carbon removed by the global open ocean (Borges & Abril 2012; Cai et al. 2021; Shen et al. 2019; Bertin et al. 2023). The primary driver is a mismatch between high riverine DIC input and insufficient buffering capacity during estuarine mixing, leading to CO_2 outgassing until surface waters equilibrate with the atmosphere and/or mix sufficiently with seawater (Borges & Abril 2012; Bertin et al. 2023; Shen et al. 2019). A model study of the Amazon river plume found that an addition of 20 µmol kg^{-1} TA could achieve 3.6 Mt CO_2 yr^{-1} of carbon removal, with 66% (2.4 Mt CO_2 yr^{-1}) attributable to reduced outgassing in the low-salinity mixing zone rather than additional CO_2 uptake, indicating substantial potential for estuary-based CO_2 outgassing reduction via river-based OAE (Mu et al. 2023). The addition of alkaline minerals to acidified rivers at downstream reaches or directly within the river plume may enable CDR with minimal ecological disturbance, as this approach avoids exposing freshwater ecosystems to significant alkalization, while mimicking the natural transition toward seawater mixing. This way, river-based OAE would effectively accelerate the natural buffering process that occurs as river water mixes with seawater within estuaries, resulting in a reduction of CO_2 outgassing while limiting environmental impacts to those already associated with the freshwater to seawater transition. It should be noted that approaches which reduce naturally occurring emissions instead of enhancing CO_2 uptake, requires comprehensive monitoring of impacted ecosystems and carbon pools to assure the measure does not indirectly induce additional outgassing of CO_2 (Swoboda & Oschlies 2025; see section 4.8.3. on air-sea gas exchange for details).

4.3.3. Coastal CO_2 removal via river-based OAE

Rivers may also be considered as a transport medium for the addition of alkalinity to coastal oceans for the enhancement of CO_2 uptake. Here, studies on the effect of rivers with high alkalinity on the coastal ocean may serve as natural analogues to assess the CO_2 potential and environmental impacts that could be expected. For instance, alkalinity introduction from the Mississippi river (a river with naturally elevated alkalinity) into the Gulf of Mexico has shown to decrease coastal pCO_2 values and enhances CO_2 uptake (Gomez et al. 2021). Building on this principle, the study by Ou et al. 2025 showed that a 10% increase of alkalinity concentration in the mississippi river would yield an additional removal of 1.8 Mt CO_2 yr^{-1} within the Gulf of Mexico. Further, estimates for CO_2 removal via a 10% alkalinity increase in the 25 largest rivers by discharge amounted to 23.2 Mt CO_2 yr^{-1}, indicating considerable potential of river-based OAE. However, studies also indicate constraints on the dissolution capacity of added alkalinity sources within rivers, which may lead to secondary precipitation (Tian et al. 2025). Therefore, rivers with naturally high levels of alkalinity are likely less suitable as transport mediums of additional alkalinity to coastal oceans.

4.4. Reactor based alkalinity approaches

Reactor-based OAE involves dissolving alkaline minerals under controlled conditions, enabling optimization of dissolution rates and efficiency while minimizing unwanted side effects such as secondary mineral precipitation from oversaturation (see section 4.6.2.). This flexibility allows the use of mineral types and conditions that would otherwise be unsuitable for direct environmental application. Depending on the approach, dissolution may produce either a CO_2-unequilibrated alkaline solution, which facilitates CO_2 uptake upon release into rivers or the ocean, or a CO_2-equilibrated solution, where CO_2 is injected prior to release to avoid uncertainties in natural CO_2 equilibration and to limit pH shifts and alkalinity oversaturation upon discharge (Moras et al. 2022, Hartmann et al. 2023, Oelkers et al. 2018, Batchelor-McAuley et al. 2022, see section 4.6.). CO_2 injection into the reactor may additionally catalyze mineral dissolution by lowering pH (Eisaman et al. 2023). Reactor-based OAE therefore holds particular promise where robust carbon accounting and environmental risk minimization are prioritized.

However, reactor-based OAE requires active management of conditions that are otherwise regulated by the natural environment, including water supply, dissolution conditions, and CO_2 equilibration. This imposes additional infrastructure requirements relative to environmental spreading approaches. Key prerequisites include logistical capacity for large-scale mineral handling, as well as reliable access to power, water, and potential reagents (e.g. CO_2, HCl). Co-location with existing infrastructure, such as industrial water inflows or outfalls, may ease permitting and reduce costs (NASEM report), but available water volumes may limit the scale of mineral dissolution, particularly where strong dilution is needed to prevent secondary precipitation and environmental impacts.

An important constraint for CO_2-equilibrated reactor approaches concerns the source of CO_2, since only CO_2 captured from the atmosphere via approaches such as DACCS, BECCS, or biochar qualifies as CDR. Other CO_2 sources than the atmosphere for equilibration, such as from industrial emissions, would constitute emission avoidance rather than net atmospheric CO_2 removal (Swoboda & Oschlies 2025).

Water density strongly controls how the discharge spreads and becomes diluted (Li et al. 2020). Less dense solutions (e.g., from freshwater sources or warmer temperatures) rise and may create a stratified surface layer. This can enhance CO_2 uptake for CO_2-lean solutions but may trap alkalinity near the surface. Denser solutions (e.g., saltier or colder) sink and may form stratified bottom layers, which may suit CO_2-equilibrated solutions but prevent CO_2-lean brines from absorbing atmospheric CO_2. Stratification of the outflow limits mixing with ambient water, slowing dilution of contrasting properties such as pH, salinity, or temperature. Reduced dilution can increase environmental risks to marine organisms, as these differences persist longer. Thus, discharge depth and local current regimes largely determine mixing and dilution rates (Pritchard et al. 2013). Overall, limited mixing may drive environmental risk more than the concentration of specific constituents in the discharge.

The strength of reactor-based OAE approaches may also be their weakness, since all processes are executed in an artificial setup that needs to be maintained. Thus, the resource demands of reactor-based OAE could be substantial. Based on correspondence with startup founders during the RETAKE Synergy Summit (March 2025), approximately 1 L of water is required to dissolve 2 g $CaCO_3$. Assuming an idealized stoichiometric CO_2 removal efficiency of 0.29 t CO_2 per tonne $CaCO_3$ (Eisaman et al. 2023), removing 100,000 t (0.1 Mt) CO_2 yr^{-1} would require the dissolution of ~344,827 t $CaCO_3$ yr^{-1}, equivalent to ~60 truckloads per working day (25 t per truck; 230 working days yr^{-1} ; Žnidarič 2015) and a water throughput of ~172 billion litres yr^{-1}, which is approximately 3.3% of the total annual German water consumption by the manufacturing sector (5.15 billion m^3 yr^{-1}; Statistisches Bundesamt, 2025). The required $CaCO_3$ mass corresponds to ~2.7 times the concrete used in the 200 m high Main Tower skyscraper in Frankfurt, while the 0.1Mt CO_2 removed represents approx. 0.6% of the annual CO_2 emissions (or approx. two days) of on-site operations of the BASF in 2024 (BASF, 2025). Considering the scale of required resources and additional technological challenges during implementation (e.g. assuring controlled conditions within the reactors despite such high water throughput), operations for reactor-based OAE will require highly efficient processes and are likely limited in their site selection and scale due to limitations of input variables for the adjustment of conditions within the reactor and co-location of suitably scaled infrastructure.

4.5. Mineral dissolution

Upon introduction into seawater, alkaline minerals begin to dissolve at rates governed by seawater temperature, pH, water movement, mineral-fluid interfacial area, and the intrinsic dissolution kinetics of the mineral (Oelkers et al. 2018, Batchelor-McAuley et al. 2022). Depending on the OAE approach, these rates may impose specific site selection criteria or require modification of seawater properties. It should be noted that the biogenic, chemical, and physical diversity of seawater can substantially affect dissolution rates in ways that are difficult to predict from observations alone, underscoring the importance of in-situ assessments for any given application.

In general, dissolution rates vary markedly across mineral types considered for OAE. Olivine dissolves considerably more slowly than oxide or hydroxide minerals, with full dissolution of 1–100 µm grains taking approximately 3–250 years, respectively (Köhler et al. 2013, Ramasamy et al. 2024). In contrast, quicklime, calcium hydroxide, and brucite at ≤63 µm dissolve within hours in seawater (Moras et al. 2022, Moras et al. 2024). Calcium carbonate cannot dissolve in surface seawater under typical conditions due to global aragonite and calcite supersaturation, except in low-pH, low-O_2 regions such as the Baltic sea or anoxic basins with low pH (Dale et al. 2024). Its use for OAE therefore usually requires the modification of seawater chemistry, for example via reactor-based approaches (Eisaman et al. 2023, see section 4.4.).

4.5.1. Temperature

Overall, the dissolution rate of minerals considered for OAE increases with seawater temperatures, thus accelerating the release of alkalinity (e.g. Hangx & Spiers 2009, Bharadwaj et al. 2013, Adkins et al. 2020, Ramasamy et al. 2024). While not as relevant for other alkaline minerals, model observations on olivine dissolution rates show a 60% increase in 29 °C water (Banda Sea), compared to 5 °C seawater (Black Sea, Ramasamy et al. 2024), indicating warmer surface waters may be favorable for coastal OAE with olivine.

4.5.2. pH

Natural surface seawater pH ranges between ~7.5–8.5 (Ramasamy et al. 2024), while reactor-based approaches may significantly alter pH to enhance dissolution rates. For oxides, hydroxides, and carbonates, dissolution rates increase with decreasing pH (Bharadwaj et al. 2013). In the case of calcium carbonate, even small pH reductions such as those from ocean acidification may enable solubility in otherwise supersaturated surface waters (Eyre et al. 2014). Olivine dissolution shows a saddle point with minimum rates at ~pH 7 and increased rates at both lower and higher pH. However, CO_2 partial pressures near atmospheric levels neutralize this enhancement between pH 7–12 (Wogelius & Walther 1990), potentially reducing olivine dissolution rates in surface seawater with effective air-sea gas exchange.

4.5.3. Ambient water velocity

Mineral dissolution rates are significantly influenced by hydrodynamic conditions. Water flow across a surface creates a drag force that reduces flow velocity near the surface, such that molecular diffusion becomes the dominant transport process within a thin layer above the surface, termed the diffusive boundary layer (DBL) (e.g. Liu & Dreybrodt 1997; Ploug et al. 1999; Secchi et al. 2022). DBL thickness is inversely proportional to flow velocity, decreasing with increasing flow until turbulence disrupts it (Moradi et al. 2018).

For mineral dissolution, a thick DBL restricts transport of dissolved material away from the mineral surface, potentially saturating the DBL and constraining further dissolution (Liu & Dreybrodt 1997). This has been demonstrated for olivine, where rotation-enhanced flow increased dissolution rates up to 19-fold compared to stagnant conditions (Flipkens et al. 2023). Consequently, dissolution rates of minerals spread on the seafloor depend strongly on pore water exchange to maintain low DBL saturation, and grain sizes must be sufficiently large to permit flow within pore space (Flipkens et al. 2023). Dissolution in coastal environments may therefore be significantly reduced under low ambient water movement (see 4.1. for details on coastal mineral spreading).

4.5.4. Mineral surface area

Two principal surface area factors affect mineral dissolution: surface roughness and coatings that limit exchange with ambient water. While larger surface areas are generally assumed to enhance dissolution (Oelkers et al. 2018), uneven surfaces locally increase DBL thickness, further shielding the mineral from water exchange (Zetsche et al. 2020; Secchi et al. 2022). Surface coatings may similarly restrict exchange. A five-year lasting observation of olivine showed bacterial colonization forming biofilms on reactive surfaces, nearly stagnating dissolution rates after ~30 days (Oelkers et al. 2015). Additionally, DBL oversaturation can trigger secondary mineral precipitation, coating the mineral surface and impeding dissolution, as observed for calcium oxide and hydroxide, which may become coated by aragonite when alkalinity levels are too high (Fuhr et al. 2022, Moras et al. 2022; see section 4.6). Hydrodynamic conditions affecting the DBL therefore significantly influence both the rate and stability of mineral dissolution.

4.6. Alkalinity oversaturation and mineral precipitation

The addition of alkaline minerals to seawater may result in secondary mineral formation, either through incomplete dissolution of the added mineral or precipitation driven by oversaturation of dissolution products, which reduces the net alkalinity available for CO_2 removal (Geerts et al., 2025). The efficiency of this process is described by the alkalinity production factor, where a value of 1 indicates complete dissolution without secondary mineral formation, and a value of 0 indicates that secondary minerals have reincorporated an amount of alkalinity equivalent to that initially introduced. In cases of so-called runaway precipitation, secondary mineral formation may remove more alkalinity than was originally added, resulting in a negative alkalinity production factor and a net reduction in the CO_2 uptake capacity of seawater, potentially driving CO_2 outgassing. Secondary mineral formation thus represents a critical constraint on OAE efficiency. The type, composition, and quantity of secondary minerals formed depend on the introduced mineral, the degree of oversaturation of dissolution products, and the ambient chemical and physical conditions of the receiving seawater. Consequently, accurate assessment of the alkalinity production factor for any OAE application must account for the individual contributions of all secondary precipitates formed (Geerts et al., 2025).

4.6.1. Authigenic secondary precipitation

4.6.1.1. Phylosilicate mineral precipitation

The direct alteration of olivine or precipitation of its dissolution products, also known as reverse weathering, may lead to phyllosilicate formation. Clay minerals, representing a subset formed under early diagenetic conditions (< 50°C, low to moderate pressures), are most relevant for coastal OAE.

Serpentine oversaturation has been observed in controlled OAE experiments (Fuhr et al. 2022; Flipkens et al. 2023). However, broader applicability in coastal systems is considered limited, as serpentine typically forms at 50–600°C associated with hydrothermal conditions, with similar constraints applying to derivative minerals such as talc (Geerts et al. 2025), suggesting restricted precipitation despite oversaturation during OAE experiments.

Among clay minerals, sepiolite may precipitate directly from olivine dissolution products at pH > 8 in the presence of sufficient Mg^{2+} and silicate concentrations (Wollast et al. 1968; Griffioen 2017; Geerts et al. 2025), though precipitation kinetics under coastal seawater conditions are slow and are further impeded by Na^+ (Baldermann et al. 2018; Tosca & Masterson 2014; Geerts et al. 2025). Within OAE experiments, mixes of amorphous silicate containing phases that may have included sepiolite have been reported (Fuhr et al. 2022), as well as direct sepiolite precipitation, albeit in small amounts (Rigopoulos et al. 2018).

4.6.1.2. Redox precipitation

Under oxic conditions, Fe^{2+} released during mineral dissolution may react to form iron (hydr)oxides, consuming 2 mol alkalinity per mol Fe^{2+} (Geerts et al. 2025). Iron (hydr)oxide formation has been confirmed during olivine

dissolution experiments (Fuhr et al. 2022; Flipkens et al. 2023). As Fe content in olivine varies, the resulting reduction in the alkalinity production factor is composition-dependent. For an olivine with 20 mol% Fe content, complete Fe^{2+} conversion to iron (hydr)oxides would reduce the alkalinity production factor from 1 to 0.8, making this a relevant consideration for net CO_2 removal efficiency, particularly for the Fe-rich olivine endmember fayalite (Geerts et al. 2025).

Under anoxic conditions, sulfate reduction yields +2 mol alkalinity per mol H_2S generated ($SO_4^{2-} + 2CH_2O \rightarrow H_2S + 2HCO_3^-$). If released Fe^{2+} subsequently reacts with this H_2S to form iron sulfides (FeS_x), 2 mol alkalinity are consumed, giving a net cycle effect of zero. However, where FeS formation prevents H_2S from reaching oxic conditions and undergoing sulfide oxidation ($H_2S + 2O_2 \rightarrow SO_4^{2-} + 2H^+$, −2 mol alkalinity), an additional alkalinity loss is avoided (Geerts et al. 2025).

As a result, iron sulfide formation improves the net alkalinity production factor when H_2S would otherwise have escaped to oxic waters, but not where it would have remained in permanently anoxic porewater, implying the need for baseline assessments on natural alkalinity cycling when budgeting the alkalinity production factor under the release of Fe^{2+} via OAE.

4.6.1.3. Carbonate precipitation

Calcium carbonate ($CaCO_3$) precipitation is the most recognised form of secondary mineral formation associated with OAE and represents a key constraint on alkalinity addition efficiency. Surface seawater is generally supersaturated with respect to $CaCO_3$, with aragonite saturation states (Ω_a) currently ranging from 1.5 to 4.5, increasing with water temperature (Feely et al., 2023). Alkalinity addition shifts the carbonate system toward higher carbonate ion concentrations, further elevating Ω_a and potentially triggering precipitation either homogeneously, heterogeneously onto mineral particles, or pseudo-homogeneously onto organic colloids (Moras et al., 2022; Fuhr et al., 2022). In natural seawater, homogeneous precipitation is unlikely given the ubiquitous presence of particles favouring heterogeneous or pseudo-heterogeneous nucleation at lower Ω (Schulz et al., 2023).

The Ω_a threshold for precipitation depends on the precipitation pathway and mineral type. For pseudo-heterogeneous precipitation, the threshold has been determined at $\Omega_a \approx 12.3$ at 20°C and a salinity of 35 (Marion et al., 2009), though this value is expected to vary with temperature and salinity. For heterogeneous precipitation, threshold Ω depends on lattice compatibility between $CaCO_3$ and the substrate mineral. Calcium oxide and calcium hydroxide additions showed no precipitation for ≥ 30 days up to $\Omega_a \approx 5$, while precipitation occurring within hours to days at higher saturation states reduced seawater alkalinity (Moras et al., 2022; Hartmann et al., 2023). For olivine, heterogeneous $CaCO_3$ precipitation was observed at Ω_a of 3–3.8 within 0.5–4 days using ultramafic sand (~70% olivine content; Fuhr et al., 2022), whereas no precipitation occurred at $\Omega_a \leq 1$ with ~90% olivine sand (Flipkens et al., 2023).

Three strategies can mitigate $CaCO_3$ precipitation and thereby increase effective alkalinity addition: rapid dilution, prior CO_2 equilibration, and introduction of alkalinity as an aqueous solution. Rapid 1:1 dilution to $\Omega_a \approx 5$ within 10 min to 1 day doubled the stable alkalinity addition from 250 to 500 µmol kg^{-1} over 30 days, and a 1:7 dilution following a 2000 µmol kg^{-1} addition similarly preserved alkalinity when applied within 10 min to 1 hour; delayed dilution resulted in prior mineral precipitation (Moras et al., 2022). These results indicate that the faster the precipitation threshold is exceeded, the more rapidly dilution must follow. CO_2 equilibration concurrent with alkalinity addition reduced Ω_a from 5 to 3.3 for a 250 µmol kg^{-1} addition, suggesting that pre-equilibration with CO_2 extends the window for alkalinity addition before precipitation onset (Moras et al., 2022; Hartmann et al., 2023; Schulz et al., 2023). Introducing alkalinity as an aqueous solution eliminates heterogeneous nucleation sites, allowing substantially higher addition levels: alkalinity increases of up to 4800 µmol kg^{-1} ($\Omega_a \approx 13$) remained stable over four days (Hartmann et al., 2023), and additions of 2100 µmol kg^{-1} ($\Omega_a \approx 12$) were stable over 90 days, while 2400 µmol kg^{-1} ($\Omega_a \approx 13.5$) led to precipitation after 10 days, consistent with the pseudo-heterogeneous threshold of $\Omega_a \approx 12.3$ (Marion et al., 2009). These findings indicate that aqueous alkalinity delivery can approach but not safely exceed the pseudo-heterogeneous precipitation threshold.

4.6.2. Biogenic secondary precipitation

4.6.2.1. Biological calcification

Biological calcification, by which organisms such as coccolithophores, molluscs, and corals incorporate $CaCO_3$ into shells or skeletons, reduces free alkalinity by 2 equivalents per mole of $CaCO_3$ removed. Therefore, if OAE were to enhance calcification above baseline levels, net added alkalinity and thus OAE efficiency would be reduced.

Large-scale introduction of limestone-based minerals has been suggested to promote calcifiers through the release of Ca^{2+} (Renforth et al. 2017; Bach et al. 2019). Analogies with ocean acidification studies, where very low CO_2 levels (~120 ppm) favor calcification in laboratory settings (Riebesell et al. 2000), have been referred to, to support this concern (Renforth et al. 2017; Lehman & Bach 2025). However, changes in calcification driven by shifts in HCO_3/CO_3 ratios cannot be directly transferred to effects of alkalinity or Ca2+ saturation (Renforth et al. 2017). Current laboratory and model studies find no indication that limestone-based OAE promotes calcification (Gately et al. 2023; Seifert et al. 2025), and Cretaceous records show calcifiers flourished when CO3 concentrations were only ~25% of modern levels (Fabry et al. 2008; Horita et al. 2002). There is therefore no evidence that OAE generally enhances calcification, and strong indication that conclusions about biological calcification cannot be drawn from shifts in carbonate chemistry alone (Fabry et al. 2008; Horita et al. 2002).

Nevertheless, alkalinity addition counteracts ocean acidification, restoring calcification in negatively impacted organisms (e.g. Albright et al. 2016). In regions where ocean acidification has already suppressed baseline calcification, OAE may temporarily restore it within locally constrained areas, albeit at a cost to OAE efficiency. In high-value ecosystems impacted by ocean acidification, such as coral reefs, this tradeoff may be acceptable, as ecosystem protection could provide additional revenue streams (e.g. tourism) that offset the costs of otherwise unprofitable OAE measures.

4.6.2.2. Mineral precipitation within biogenic microenvironments

Microbial communities actively modulate their microenvironment, creating highly localised conditions that can induce secondary mineral precipitation due to local supersaturation, even when bulk water conditions favour mineral solubility (Krause et al. 2018; del Buey et al. 2023). Under OAE, elevated alkalinity levels may lower the threshold for such supersaturation within microbial microenvironments, leading to alkalinity loss through microbial processes that induce precipitation of secondary minerals, such as by calcium carbonate and clay minerals (Krause et al. 2018; del Buey et al. 2023, see table. 2).

While microbial-driven precipitation of alkalinity-consuming secondary minerals is well established, its relevance under realistic OAE conditions remains unclear and warrants dedicated investigation. Given that shelf ecosystems commonly host microbial mats performing ammonification, denitrification, and sulfate reduction at scale and which are known to elevate alkalinity (Fan et al. 2015; Hochroth & Pfister 2024; Thomas et al. 2009), microbial communities can be expected to play a role in coastal OAE practices.

4.7. Environmental effects of mineral contamination and trace metal introduction

Alkaline minerals may contain contaminants such as trace metals, heavy metals, or other hazardous constituents, necessitating an assessment of natural baseline levels and subsequent control measurements to ensure that the minerals do not exceed safe threshold values when added. As a result, environmental threshold values would limit the total amount of minerals that can be applied (Schuiling & Krijgsman, 2006). Therefore, the composition of the minerals must be confirmed from potential mine operators when selecting deployment sites, to avoid use of unsuitable sources (see details in section 3.1.1.).

4.7.1. Biological impacts of OAE on marine ecosystems

The introduction of alkalinity into seawater alters carbonate chemistry and seawater pH, and depending on the alkalinity source, may introduce bioactive nutrients, micronutrients, or toxic trace metals with the potential to impact marine ecosystems. Experimental studies on coastal phytoplankton communities indicate that gross primary production is largely unaffected by alkalinity addition via NaOH, olivine, or calcium oxide, but that species composition shifts significantly between control and treatment groups (Ferderer et al., 2022; Guo et al., 2024), suggesting nutrient cycling across trophic levels may be altered. At the species level, olivine addition elicited neutral to beneficial physiological and biogeochemical responses across six major phytoplankton species, with no observed toxic effects (Hutchins et al., 2023). However, species-level performance gains may confer competitive advantages that will nonetheless drive community composition shifts. Notably, addition of non-CO_2-equilibrated NaOH reduced picoeukaryote abundance, likely because this group relies predominantly on dissolved CO_2 rather than HCO_3^- as a carbon source (Ferderer et al., 2022).

In contrast to the limited responses observed experimentally, an Earth System Model study found that OAE indirectly reduced net primary production by up to 15%, with consequential reductions in phytoplankton-driven carbon uptake, while biological feedbacks simultaneously enhanced OAE-driven CO_2 uptake such that net oceanic CO_2 uptake remained higher in OAE scenarios (Seifert et al., 2025; see section 4.9.1.). These contrasting results highlight the complexity of carbonate system-phytoplankton interactions and the limitations of controlled experimental settings for predicting ecosystem-scale responses.

Toxic trace metal release has been identified as a potential threat to marine biota (Bach et al., 2019; Guo et al., 2022; Guo et al., 2024; Eisaman et al., 2023), with nickel shown to suppress growth in certain phytoplankton species (Guo et al., 2022). However, toxic effects can be largely avoided through careful selection of mineral sources with appropriately low contaminant levels.

Overall, knowledge of OAE impacts on marine biota remains limited, particularly at the community level. Current evidence suggests that large-scale OAE will primarily affect phytoplankton community composition, with likely cascading consequences for higher trophic levels and biogeochemical fluxes, including the biological carbon pump (see section 4.9.1.).

4.8. Atmospheric carbon uptake and storage

The primary step which makes an OAE practice also a CDR practice is achieved when the amount of atmospheric CO_2 is reduced. Thus, the mechanisms behind air-sea gas exchange in regards to CO_2 are of particular relevance for the conceptualisation and execution of OAE practices. At the core of the exchange process is gas diffusion, which is described by Fick's First Law. The net CO_2 flux across the ocean–atmosphere interface is typically expressed as:

$$F = k \cdot \alpha \cdot (pCO2\ atm - pCO2\ ocean)$$

Here, k represents the gas transfer velocity, a parameter that quantifies how efficiently gas molecules traverse the air–sea boundary layer; α is the solubility coefficient of CO_2 in seawater, which is a function of temperature and salinity; and the difference between the atmospheric and oceanic partial pressures of CO_2 ($pCO_2\ atm - pCO_2\ ocean$) acts as the driving force for diffusion. In essence, CO_2 will move from regions of higher partial pressure to lower partial pressure until equilibrium is achieved. As OAE's impact on CO_2 gas-transfer is a crucial step for a reduction of atmospheric CO_2, the parameterisation of the above expression and processes impacting them are outlined below.

4.8.1. Surface processes impact the CO_2 gas transfer velocity, k

Air-sea CO_2 transfer occurs via diffusion across the surface boundary layer, and is enhanced by forces which disrupt the thin surface boundary layer and replenish it with water or air unsaturated in CO_2. Therefore, the gas

transfer velocity k is primarily governed by wind speed, which enhances surface roughness, promotes wave breaking, and generates bubbles that continuously renew the boundary layer (Prytherch et al., 2010). Processes that inhibit boundary layer renewal reduce k and thereby suppress CO_2 exchange. Most notably, sea-ice formation creates a near-impermeable barrier to gas exchange (Prytherch et al., 2017); given that full CO_2 equilibration following OAE is expected to require approximately one year, the advection of treated water into colder regions or seasons is a relevant consideration for predicting equilibration timescales. Biologically derived surfactants similarly reduce k by suppressing diffusion and dampening capillary-gravity waves, limiting boundary layer renewal (Schmidt & Schneider, 2011; Pereira et al., 2018). Observational estimates indicate surfactant-driven reductions in gas transfer velocity of 2–32% in the Atlantic and ~20% in the Pacific (Pereira et al., 2018; Mustaffa et al., 2020). Further, surfactant concentrations covary with biological production and peak during bloom periods (Schmidt & Schneider, 2011). The concentration of surfactants and the transition of water bodies across seasons may thus be relevant considerations when predicting the additional uptake of CO_2 via OAE.

4.8.2. The solubility coefficient of CO_2

The CO_2 gas solubility determines the amount of CO_2 that can dissolve in seawater and, following Henry's law, is directly proportional to the partial pressure of CO_2. The primary controlling factors are water temperature and salinity (Teng et al., 1996; Zhao et al., 2015). Temperature exerts the strongest influence, with rising sea surface temperatures reducing CO_2 solubility, elevating surface pCO_2 and driving outgassing. This underlies the general seasonal pattern in which the ocean acts as a CO_2 source during warmer seasons and a sink during colder seasons (Rustogi et al., 2023; Honkanen et al., 2021; Honkanen et al., 2024). Salinity also decreases CO_2 solubility, but within the typical open-ocean range of 32–38 PSU the effect is negligible (~2% at 20°C; Boutin et al., 2021; Unisense Unit Converter), making salinity a relevant consideration only in low-salinity regions such as the northern Baltic Sea, where near-zero PSU values increase CO_2 solubility by up to ~15% relative to 38 PSU at 20°C. While it should be noted that pressure increases enhance CO_2 solubility (Teng et al., 1996) variations in atmospheric pressure are inconsequential for air-sea exchange and may only become relevant when OAE-treated water is at depth, for example through discharge via outfalls (see section 3.4.).

4.8.3. Controls on the direction of CO_2 gas exchange

The direction of air-sea CO_2 exchange is determined by the pCO_2 gradient between atmosphere and ocean ($pCO2\ atm - pCO2\ ocean$): a positive difference drives oceanic CO_2 uptake, a negative difference drives outgassing. Although the ocean is broadly considered a net CO_2 sink, this characterisation is based on averaging across extended temporal and spatial scales (Rustogi et al., 2023). At finer time scales, large portions of the ocean act as net CO_2 sources seasonally, monthly, and daily (Rustogi et al., 2023; Honkanen et al., 2021; Honkanen et al., 2024), with certain regions such as the Baltic Sea functioning as annual net sources (Parard et al., 2017; Wesslander et al., 2010). Consequently, while most of the ocean surface exhibits annual net CO_2 uptake, the majority also experiences net outgassing over sub-annual periods throughout the year (Rustogi et al., 2023).

The distinction between OAE driving additional CO_2 uptake versus reducing CO_2 outgassing is rarely addressed (Swoboda & Oschlies, 2025) but practically significant, as the underlying processes operate on fundamentally different timescales. Additional CO_2 uptake via OAE is generally expected to require up to one year to complete (see section 4.8.1.), whereas the incorporation of dissolved CO_2 into the carbonate system occurs near-instantaneously (personal correspondence with K. Schulz). This has direct implications for MRV: quantifying additional CO_2 uptake requires tracking the treated water mass over approximately one year, while a reduction in outgassing would be detectable on much shorter timescales, potentially simplifying MRV considerably. Furthermore, seasonal and sub-seasonal variability in air-sea CO_2 flux at a deployment site must be considered when attributing the mechanism of carbon increase in seawater and designing appropriate MRV frameworks.

Since field studies on OAE remain scarce due to logistical, financial and, in some cases, legal constraints, CO_2 uptake estimates rely predominantly on modelling. Where models do not resolve sub-annual variability in air-sea CO_2 flux direction, OAE efficacy may be underestimated or its timing misrepresented, causing issues with model

parameterisation and validation against field data. Where field measurements are conducted, it is essential that both CO_2 influx and outgassing are monitored to ensure the full effect of OAE is captured.

4.8.4. Ocean circulation and regional variability

Beyond surface processes, vertical mixing and redistribution of water masses within the water column can significantly influence regional CO_2 exchange and are therefore critical to OAE efficacy. The ocean water column is broadly stratified into layers of uniform density, within which varying mixing processes redistribute water masses vertically, both within and between layers. These processes span local to global scales and short to millennial timescales, and are driven by density differences and wind-induced shear forces. The most relevant for OAE include wind-driven mixing of the surface layer; the seasonal mixed-layer pump (Dall'Olmo et al., 2016); eddy- and frontal-driven subduction (Mahadevan & Tandon, 2006; Omand et al., 2015); and density-driven up- and downwelling.

Since air-sea gas exchange is strictly confined to the ocean surface, OAE-treated water must remain at the surface until CO_2 equilibration is complete. Horizontal advection may carry treated water into subduction or downwelling zones, while seasonal thermally-driven stratification in spring can subduct surface water via the mixed-layer pump before equilibration is achieved (Dall'Olmo et al., 2016). Even within the surface mixed layer, variations in mixed-layer depth and recirculation timescales will modulate the rate of air-sea CO_2 exchange. Despite mixing processes and advective trajectories are difficult to predict and subject to seasonal and interannual variability, it is crucial that in deployment planning possible efforts are undertaken to minimize the risk of premature subduction of OAE-treated water.

4.8.5. Concluding remarks and considerations on air-sea gas exchange

The ocean's three-dimensional circulation means that OAE-treated surface water may be subducted or mixed into the interior before full CO_2 equilibration is achieved. Among the factors governing air-sea CO_2 exchange rates, horizontal advection of treated water masses deserves particular attention, as these masses will traverse regions of varying environmental and seasonal conditions prior to equilibration. Although ocean mixing is often unpredictable, many areas exhibit consistent or recurrent subduction events, including downwelling zones, frontal subduction, and seasonal thermally-driven mixing, and OAE deployment should therefore be located and timed to avoid these before equilibration is complete. Additionally, seasonal variability in temperature, sea-ice coverage, and surfactant films from primary production can substantially modulate gas exchange rates and should be accounted for both during application and along the trajectory of the treated water mass.

A further critical consideration is whether OAE drives net atmospheric CO_2 uptake or, alternatively, a reduction in CO_2 outgassing. Since the conversion of dissolved CO_2 into carbonate species is near-instantaneous, a reduction in outgassing would constitute equally immediate carbon storage. This distinction is practically significant: OAE-driven suppression of outgassing would circumvent many of the uncertainties associated with quantifying new CO_2 uptake, substantially simplifying MRV requirements (see section 5.1.2.).

Current understanding of OAE-induced air-sea gas exchange relies predominantly on models, as laboratory experiments cannot replicate open-ocean dilution and field signals are likely undetectable. Models commonly treat the ocean as an annual net CO_2 sink, implicitly assuming OAE drives CO_2 uptake, yet direct flux measurements reveal episodic outgassing on daily to seasonal timescales. Given that equilibration following OAE is estimated to require approximately one year (ref), treated water will inevitably encounter periods of CO_2 outgassing before equilibration is complete. Since pCO_2 transfers rapidly into the carbonate system, any surplus CO_2 present during such a shift would be incorporated directly into the carbonate pool rather than exchanged with the atmosphere. Consequently, the predominant dominant climatic effect of OAE may be a suppression of CO_2 outgassing rather than additional atmospheric uptake, which would be an outcome that current model parameterisations, by assuming net sink conditions, may systematically overlook.

4.9. Additionality

OAE is an environmental intervention to which natural biogeochemical processes will respond and adjust. With respect to the carbon cycle, OAE may reduce atmospheric carbon removal that would otherwise occur through naturally occurring processes, thereby partially counteracting its intended climate benefit. It is therefore essential that CDR via OAE exceeds any concurrent reduction in natural carbon removal, i.e., OAE-driven CDR must be additional (Bach et al., 2024; Swoboda & Oschlies, 2025). The two primary pathways by which OAE may compromise natural carbon removal are interference with the biological carbon pump and suppression of natural alkaline mineral weathering.

4.9.1. Impacts of OAE on the biological carbon pump

The biological carbon pump (BCP) is an umbrella term for the biological processes that transfer carbon from the surface ocean to the interior, thereby maintaining low surface pCO_2 and driving atmospheric CO_2 uptake. Phytoplankton fix dissolved inorganic carbon (DIC) into particulate organic carbon (POC) via photosynthesis, which is subsequently modulated by (1) consumption and trophic transfer, including excretion of fecal pellets, (2) gravitational settling of phytoplankton, fecal pellets, and marine aggregates, and (3) active vertical transport via diel migration and physical mixing (Boyd et al., 2019). Although surface remineralisation returns a large fraction of fixed carbon to the DIC pool, approximately 1–10% is exported below 2000 m, where it is stored on centennial to geological timescales (Passow & Carlson, 2012; Boyd et al., 2019). Estimates suggest the BCP sequesters 6–17 Gt C yr^{-1} (Boyd et al., 2019), equivalent to 60–170% of annual oceanic anthropogenic CO_2 uptake (Friedlingstein et al., 2026) and ~6–18% of total atmospheric carbon uptake by the global ocean (Lal, 2008), establishing it as one of the principal natural mechanisms moderating anthropogenic climate forcing.

Because OAE reduces surface pCO_2 through inorganic means, it operates in direct competition with the BCP for atmospheric CO_2. An earth system model study found that OAE locally reduced biologically driven CO_2 uptake by 20–60%, resulting in a 3% global decrease in phytoplankton-driven uptake (Seifert et al., 2025). Prior suggestions that alkalinity-supplying minerals could enhance primary production through nutrient fertilisation, such as diatom stimulation via olivine or coccolithophore stimulation via $CaCO_3$ introduction (Bach et al., 2019), could not be confirmed. Instead, NPP declined across all investigated phytoplankton groups irrespective of species (Seifert et al., 2025). Although OAE still yielded a net increase in atmospheric CO_2 removal, the concurrent NPP reduction implies a shift in the fate of sequestered carbon, where relatively more carbon is retained in the inorganic carbonate system and less is converted to POC, restricting its availability to BCP processes. Furthermore, low-pCO_2 conditions have been shown to favour small picoplankton species, and such a community shift at the base of the food web would propagate cascading effects through higher trophic levels. Since aggregate settling velocities and remineralisation depths are strongly dependent on community composition, a shift toward smaller phytoplankton would alter the nutritional quality and carbon flux efficiency of sinking material throughout the water column (Ploug et al., 2008; Iversen et al., 2010; Romero et al., 2020).

In summary, while OAE increases net atmospheric CO_2 removal, it does so in competition with the BCP and redirects sequestered carbon away from biological pathways, with likely consequences for marine ecosystem structure and the natural carbon storage efficiency of the BCP.

4.9.2. Impacts of OAE on natural alkalinity release

Global ocean alkalinity is primarily governed by the input of weathering-derived alkalinity and its removal through $CaCO_3$ formation (Middelburg et al., 2020). Although the upper ocean is generally supersaturated with respect to calcite and aragonite ($\Omega > 1$), local saturation states can fall below 1 due to biological processes such as respiration and organic matter degradation, often compounded by restricted water exchange, upwelling, or reduced light availability. Such undersaturation is naturally buffered by dissolution of $CaCO_3$ in sediments and biogenic carbonates from primary producers and corals. Anthropogenic CO_2 uptake progressively erodes this supersaturation margin, increasing the frequency of localised undersaturation and $CaCO_3$ dissolution. The addition of highly soluble OAE-derived alkalinity raises ambient Ω, thereby suppressing the dissolution of less soluble

natural sources such as benthic $CaCO_3$ and calcifying organisms (Renforth & Henderson, 2017), potentially weakening natural weathering feedbacks and reducing natural alkalinity-driven carbon storage, particularly within coastal sediment microenvironments (Bach, 2024).

A laboratory study by Bach (2024) quantified this effect by measuring natural $CaCO_3$ dissolution from beach sand across varying DIC concentrations and in the presence of added alkalinity sources (olivine, steel slag, and NaOH). Natural alkalinity release declined across all treatments, though the magnitude varied considerably: CO_2-equilibrated NaOH had the smallest suppressive effect, followed by steel slag, while CO_2-unequilibrated NaOH and olivine caused the largest reductions, even leading to a net negative additionality of the added alkalinity at higher DIC concentrations. These findings suggest that interactions between added and natural alkalinity sources in coastal environments are likely to reduce OAE additionality and constrain the feasibility of coastal mineral-based OAE (see section 4.1.).

To date, there are no field observations on the effect of OAE on the dissolution of natural alkalinity sources in the open ocean. In the open ocean, surface $CaCO_3$ dissolution is predominantly attributed to dissolution of highly soluble magnesium calcites and metabolically driven undersaturation via respiration within microenvironments such as marine particles (Sulpis et al., 2021; see section 4.9.1.). Applying the same reasoning as in Bach (2024), OAE-induced increases in Ω could displace $CaCO_3$ dissolution to deeper water layers, reducing the likelihood that released alkalinity contributes to atmospheric CO_2 buffering. Given that approximately 30% (900 ± 300 Mt) of annual open-ocean $CaCO_3$ dissolution occurs within the surface ocean, suppression of this natural alkalinity flux by large-scale OAE could meaningfully limit additionality and complicate open-ocean implementation strategies.

In summary, large-scale OAE may substantially suppress natural alkalinity cycling in both coastal and open-ocean environments, reducing its net additionality. The results of Bach (2024) suggest that prior CO_2 equilibration of added alkalinity could mitigate this effect, indicating that equilibration protocols may be essential for OAE feasibility. Critical knowledge gaps remain regarding the magnitude and timescales of OAE impacts on natural alkalinity cycling and the extent to which application strategies such as pre-equilibration can minimize these effects, all of which must be addressed before large-scale deployment.

5. Monitoring, reporting and verification and carbon uptake

5.1. Carbon storage via OAE and caveats regarding set targets, assessment limitations and MRV

A fundamental challenge in determining CDR viability lies in the tension between the constraints of the natural world with predetermined rules and concepts that are non-negotiable and the anthropocentric frameworks used to evaluate it, which vary with knowledge, culture, and societal context. Consequently, what constitutes net atmospheric carbon removal, an environmental side effect, or a socio-economic implication is determined by the methodologies used to assess them, as well as the legally defined targets and regulatory boundaries applied to interpret results.

The primary objective of OAE is net reduction of the atmospheric carbon pool, yet accurately quantifying this removal is challenging due to complex biogeochemical interactions and the difficulty of establishing additionality relative to a counterfactual scenario (see section 4.9.). Moreover, evaluating OAE solely on net carbon removal risks overlooking critical environmental side effects, socio-economic implications, and governance considerations. Assessing OAE success therefore requires integrated, transdisciplinary assessments encompassing environmental processes (including carbon fluxes, mineral precipitation, and additionality), socio-economic outcomes, and compliance with regulatory boundaries (see Fig. 3.).

5.1.1. Mechanisms of carbon storage via OAE and how regulatory frameworks set targets

CDR measures affect natural processes that are dynamic and change over time, while whether a CDR application is deemed successful is defined by governing law and regulation. Carbon storage success is primarily determined by two timepoints: (1) when additional carbon must be stored, and (2) the duration over which it is stored. A CDR practice should ideally reduce atmospheric carbon promptly and store it over timescales relevant for societal climate mitigation, e.g. centennial timescales (Swoboda & Oschlies 2025). However, both the required timeliness of removal and relevant storage timescales remain undefined in law and will ultimately be determined by political, societal, and feasibility perspectives.

It is furthermore crucial to recognise that both the OAE application and the stored carbon partake in the carbon cycle and are subject to time-dependent environmental processes. The total carbon storage is therefore determined not only by the quantity of additionally stored carbon, but also by ongoing processes during uptake and storage that may cause carbon to increase or decrease over time (Swoboda & Oschlies 2025). Carbon storage is thus a continuous process, and defining when and for how long storage must be accomplished determines CDR efficiency (fig. 4.).
The effect of CDR actions on atmospheric carbon can be grouped into three categories: (1) carbon removed by the CDR measure, (2) carbon emitted in performing it, and (3) carbon that would have been removed in the absence of the measure, i.e. additionality (see section 4.9.). Net atmospheric carbon reduction is derived from these three variables.

5.1.2. Monitoring, reporting and verification (MRV) for OAE and considerations on assessment scales and responsibilities

Monitoring, Reporting, and Verification (MRV) is a structured, multistep framework to ensure that CDR measures are safe, sustainable, and effective. It encompasses (1) an overarching protocol defining system boundaries, targets, and thresholds; (2) acquisition of measurements and model simulations; (3) assimilation, interpretation, and reporting of results under applicable regulations; and (4) independent third-party verification, ensuring separation of responsibilities. Taken together, MRV ensures that both the feasibility and the risks of the OAE measure are evaluated comprehensively, consistently and proactively, while ensuring credibility of the measure and the avoidance of conflicts of interest or fraud (e.g. within the Clean Development Mechanism and Article 6 of the Paris Agreement; UNFCCC, 2022).
When defining targets and thresholds for OAE, impacts on the environment can create special cases where the resulting impact can not be evaluated as (un)favorable or (un)acceptable. For instance, if the introduction of alkalinity leads to a shift in the microbial community composition without any further detectable impact on the ecosystem or biogeochemical cycles, how can it be judged if this shift in microbial composition is (un)favorable or (un)acceptable? While one could argue that any living organism has the right to exist on its own terms, with the question of microbial rights marking a particularly interesting case (Cockell et al. 2011), the fundamental idea behind OAE is to change the environment in favor of society's needs, i.e. modulate the environment towards additional carbon uptake. Thus, a shift in the environment towards a desirable state from a human centric perspective is implied through the very concept. While it is important to consider the environmental impacts that are not clearly evaluable, they will have to be weighted against the proposed benefits of OAE and in the context of existing rule of law and the potential application of double standards (e.g. industrial use of antibiotics in the context of the microbial rights argument). Therefore, if an application of OAE goes ahead, defining environmental targets and thresholds for MRV will have to consider the dilemma that an impact on the environment is inevitable.
As outlined above, the success of OAE is not determined by just the net carbon removal, but by this in conjunction with environmental repercussions and socioeconomic impacts (fig. 3.). Thus, the responsibility of a MRV process is to track a core set of indicators that reflect OAE success on net carbon removal (composed of total carbon removal, generated emissions, additionality and permanence of storage); environmental and ecosystem repercussions (e.g. release of toxic impurities, loss or shifts in biodiversity); and socioeconomic impacts (e.g. loss

of equity, social acceptability or evasion of regulations and law; Gattuso et al., 2018; Fawzy et al., 2020, Swoboda & Oschlies 2025).

A further consideration is that the effects and impacts of OAE, as well as any other CDR measure, may overlap and interact, thus leading to cumulative effects. This may include interactions between other OAE projects (e.g. overlapping alkalinity plumes), cumulative impacts such as interactions between concurrent applied marine and terrestrial CDR approaches (Keller et al. 2018) or shifts in the global alkalinity cycle, and the potential for large-scale feedbacks on the air–sea CO_2 flux (Williamson et al., 2012; Boyd et al., 2023). Therefore, requirements for MRV processes must reflect and distinguish between local to global scales at different governance levels. At the project level, MRV should focus on local carbon removal and associated environmental and social impacts to ensure compliance with permitting and regulatory frameworks. At national and international levels, the scope must expand to include the interactions between other OAE projects; cumulative impacts such as interactions between concurrent applied marine and terrestrial CDR approaches or shifts in the global alkalinity cycle; and the potential for large-scale feedbacks on the air–sea CO_2 flux (Williamson et al., 2012; Boyd et al., 2023). As a result, set system boundaries as well as modelling and measurement approaches need to address the varying needs and responsibilities of MRV from the project level to the national and international coordination of OAE and overall CDR applications.

Considering overlapping effects between multiple CDR projects on the national and international level, responsibilities of the verifying bodies will merge with those bodies responsible for permitting, to ensure that the application of CDR projects is coordinated within the realm of set targets and thresholds. This will require communication pathways that are tailored accordingly. Local projects primarily exchange information with local and national regulators, while national and international bodies are responsible for coordination and harmonisation of MRV results across jurisdictions (including frameworks such as the London Protocol and the UNFCCC), and permitting of new projects under the consideration of potential cumulative impacts through existing international CDR measures (Smith et al. 2024; Swoboda & Oschlies 2025).

As a result, a key prerequisite for effective MRV is transparent communication and open data sharing. Methodologies should be disclosed, data made interoperable, and reporting structured to enable comparability between projects and across governance scales (e.g. by following FAIR data principles). This transparency is essential not only for building trust among stakeholders but also for facilitating cumulative global assessments and ensuring that large-scale OAE deployment delivers genuine, durable atmospheric CO_2 removal without unintended environmental trade-offs (Gattuso et al., 2021; IPCC, 2022).

5.1.3. System boundaries as a determining factor for the assessment of carbon storage and overall viability of OAE

System boundaries define the scope of processes, effects, area and timescales considered in an MRV assessment, covering impacts on the carbon cycle (emissions, uptake, and additionality), resource use, environmental impacts, and socio-economic effects (Eisaman et al. 2023; Satterfield et al. 2023; Fennel et al. 2025; Chlela & Selosse 2025). Their selection therefore fundamentally determines the viability of assessment outcomes. Boundaries that are too narrow may misrepresent OAE responses and skew perceived viability (Fennel et al. 2025), while overly extensive assessments may reduce cost-effectiveness.

The required level of detail scales with application scope. Individual applications should address their direct impacts, but OAE effects may overlap with other CDR measures and extend across borders and long timescales (Keller et al. 2018). Full system assessments integrating multiple CDR measures will therefore be necessary as part of permitting, requiring national and/or international authorities to develop the expertise and capacity for such analyses. In practice, MRV will operate across multiple levels of responsibility, each requiring appropriate model and measurement detail. Model- and measurement-based evaluations will be essential to define appropriate system boundaries and determine when simplified approaches suffice versus when system-wide assessments are required.

5.1.4. Limitations within measuring and modelling of OAE applications

A key constraint for assessing OAE carbon removal and environmental repercussions is that its effects unfold over large spatial and temporal scales, making them difficult to trace. Detection of added alkalinity requires that

the added alkalinity signal exceeds both natural background variability and measurement uncertainty. This signal in the water column dilutes over time through mixing and advection (see section 4.8.4.). Direct measurements are therefore only feasible near the site of alkalinity addition, while model simulations are required to assess OAE performance and effects at larger scales where direct measurements become unsuitable (Ho et al. 2023; Fennel et al. 2025).

Models can only reflect OAE effects to the degree their parametrization allows, thus, processes not captured by model parameters will be absent from simulation outcomes. This is a relevant consideration, given considerable gaps in existing knowledge regarding OAE ecosystem feedbacks (e.g. trophic interactions and their effect on the biological pump; see section 4.9.1.) and biochemical processes (e.g. surfactants in the sea-surface microlayer and their impact on air-sea gas exchange; see section 4.8.1.). For guidance on model uncertainty and limitations, we refer to Ho et al. 2023.

6. Conclusion

The process chain assessment which served as the basis of this work enables key steps for OAE implementation to be mapped, and constraints across the entire process chain that currently limit the development and implementation of OAE as a meaningful measure for climate mitigation to be identified. Of all the steps in the OAE process chain, those related to mineral dissolution and model-based assessments appear to be currently the best understood. A key underexplored limiting factor is societal perception of OAE implementation. The assumption that CDR, and by extension OAE, will inevitably be deployed because it is deemed necessary, while neglecting the requirement for broad societal acceptance, reflects an insufficient consideration of how modern democratic decision-making functions. Field trials should therefore incorporate research into societal perception and impacts, alongside dedicated measures for meaningful public outreach and stakeholder involvement in decisions about OAE implementation, to identify and address concerns on the environment and equity as well as potential benefits and options for involvement.

Further, the techno-economic requirements of large scale OAE appear underestimated within the cited work. In particular, the fact that alkaline minerals are heavy cargo and removal of 1 t CO_2 requires at minimum 0.6 to over 2 t of alkaline mineral, its deployment relies on highly efficient logistics during acquisition, processing and deployment of alkaline minerals. In addition, associated resource requirements, in particular with regards to suitable transport infrastructure, spatial needs and water use may limit OAE potential considerably.

Lastly, MRV challenges with regards to the larger scale implementation of OAE are currently underrepresented. Because OAE actively intervenes in the global carbon cycle, the carbon removed through OAE forms a new carbon stock whose permanence and long-term effects must be considered, including overlapping effects between CDR projects and potential reductions in additionality through the accumulation of multiple deployments. Such large-scale feedback mechanisms require extensive model analyses that include data inputs from past and present OAE projects which individual project initiatives will not be able to fulfill. Thus, under the authority of higher jurisdictions, close coordination across projects and nations with the accompanied monitoring of large-scale feedbacks will be necessary. MRV must therefore operate not only at the project level for direct oversight, but also at a systemic level that accounts for cross-project interactions and large-scale feedbacks, with outcomes informing the permitting of future projects. As larger-scale field trials are conducted, the requirements, options, and repercussions of large-scale OAE will become increasingly clear, enabling the formulation of regulations, legislation, and MRV protocols that reflect the needs of a safe and sustainable OAE implementation.

With regards to the underlying mechanisms of OAE and resulting effects on the environment, there are a large number of processes that are being considered for implementation, yet the relative importance of these processes cannot be assessed. For instance, CO_2 release due to various forms of secondary mineral precipitation is rightfully a key concern about OAE that has been frequently investigated. However, there are no observations on whether the dilution of added alkalinity in the natural environment would allow a potential oversaturation to persist long enough to induce secondary mineral precipitation. A further example is the assumption that CO_2 gas-exchange via OAE requires approximately 1 year to fully unfold, while the process of OAE potentially reducing CO_2 outgassing from the surface ocean without temporal delay is underappreciated. If suppressed outgassing represents

the primary mechanism by which OAE lowers atmospheric CO_2 concentrations, this would have widespread implications on conceptualizing OAE measures, e.g. favoring waters and seasons that tend to exhibit CO_2 outgassing, decreased risk of secondary mineral precipitation by the direct transition of alkalinity into the carbonate system or simplification of MRV processes since treated water bodies do not have to be tracked until CO_2 gas-exchange is completed. However, field observations on CO_2 gas-exchange by OAE are still missing. The lack of such basic observations means that the actual relevance of proposed environmental effects cannot be assessed under real-world conditions, and that estimates of OAE potential rest on assumptions that may not accurately reflect its true capacity. Meaningful field trials are therefore necessary to better understand the fundamental processes and requirements of OAE in situ, which will allow the creation of robust estimates of how and to what extent OAE could contribute to CDR.

Drawing from this synthesis, it can be expected that successful large-scale application will likely be achieved very differently than currently anticipated. As outlined, fundamental processes are not well understood in the context of large-scale field application and underexplored aspects, in particular with regards to logistics, infrastructural requirements, technical application, societal perception and regulation, thus constraining the view of how OAE implementation is currently envisioned. As a result, even basic "back-of-the-envelope" calculations reveal that implementation scenarios currently discussed (see section 4.4. for an example on reactor-based OAE and 4.1. on coastal alkalinity distribution) are unlikely to succeed as presently considered. This however, does not preclude OAE from playing a meaningful role in climate mitigation.

In order for OAE to be successfully implemented at larger scale, we expect that projects will have to be tightly integrated into existing infrastructure as well as product-, service- and revenue-streams to allow a cost efficient implementation without disruptive resource or spatial requirements. This may reduce project cost, ease permitting and improve net CO_2 uptake efficiency. Further, such integration may diversify revenue sources of projects, making OAE less dependent on covering costs exclusively through carbon crediting. By structuring OAE implementation into discrete steps, the outlined process chain and gathered knowledge allows for a systematic assessment of where integration into existing processes may be feasible.

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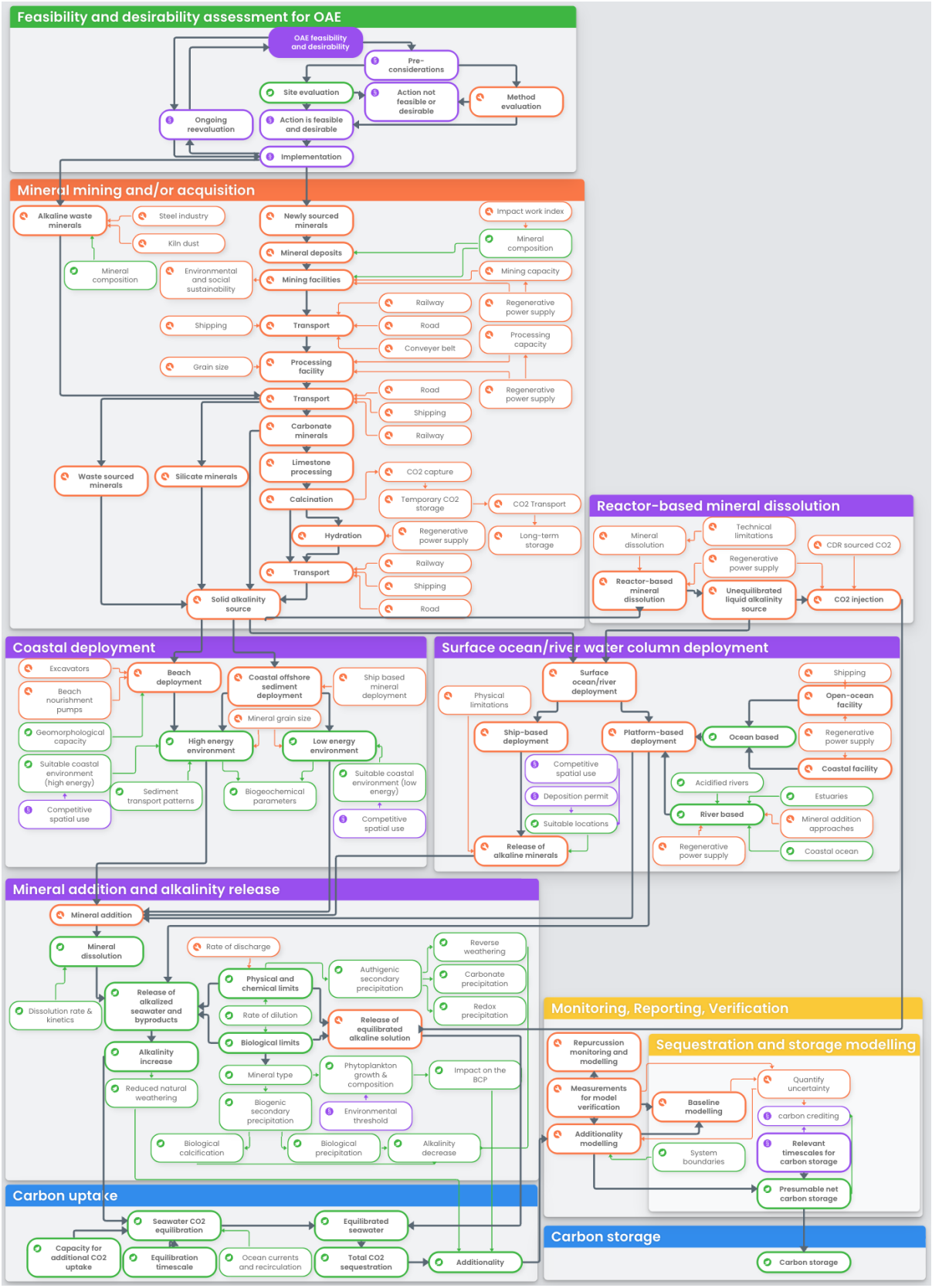
Competing interests

The authors declare no competing interest.

Supplementary material



Supplementary figure S1. **Detailed listing of relevant steps and considerations for OAE implementation defined within the process chain.** Individual steps are grouped according to their main area of relevance, including technical- (orange), environmental- (green) and societal- (purple) steps. Steps have been further grouped

in boxes under overarching themes for OAE implementation, covering "initial planning and considerations" (green), "project preparations" (orange), "deployment" (purple), "monitoring, reporting and verification" (pink), and "carbon uptake and storage" (blue).

Data accessibility statement

No data were generated in this work.

AI use disclosure

The authors used ChatGPT (OpenAI) and Claude (Anthropic) as support for language editing and literature search. All scientific content, interpretations, and conclusions were developed by the authors.

Contributions

- Contributed to conception and design: SS, SA, MF, MW, AD, AO
- Drafted and/or revised the article: SS, SA, MF, MW, AD, AO
- Approved the submitted version for publication: SS, SA, MF, MW, AD, AO

Data accessibility statement

No data were generated in this work.

Figures

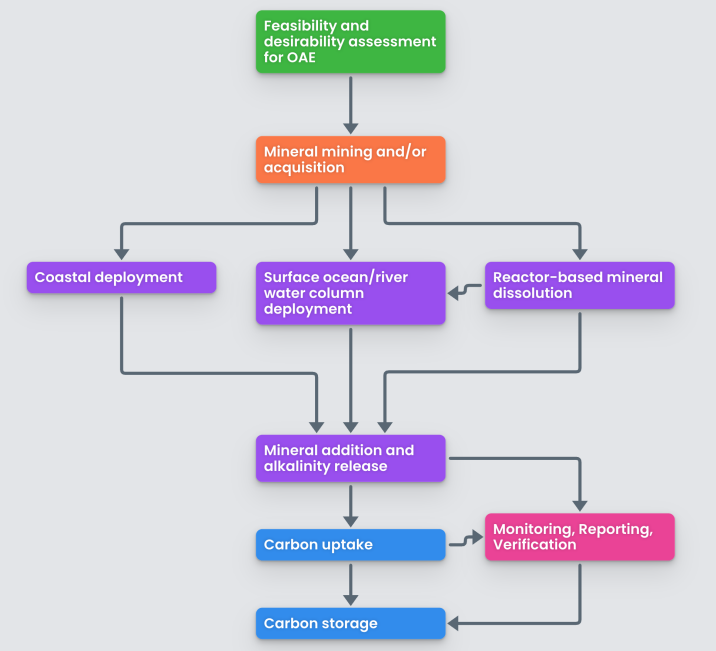


Figure 1. **Main steps for OAE implementation defined within the process chain.** The main steps have been grouped under overarching themes for OAE implementation, covering "initial planning and considerations" (green), "project preparations" (orange), "deployment" (purple), "monitoring, reporting and verification" (pink), and "carbon uptake and storage" (blue).

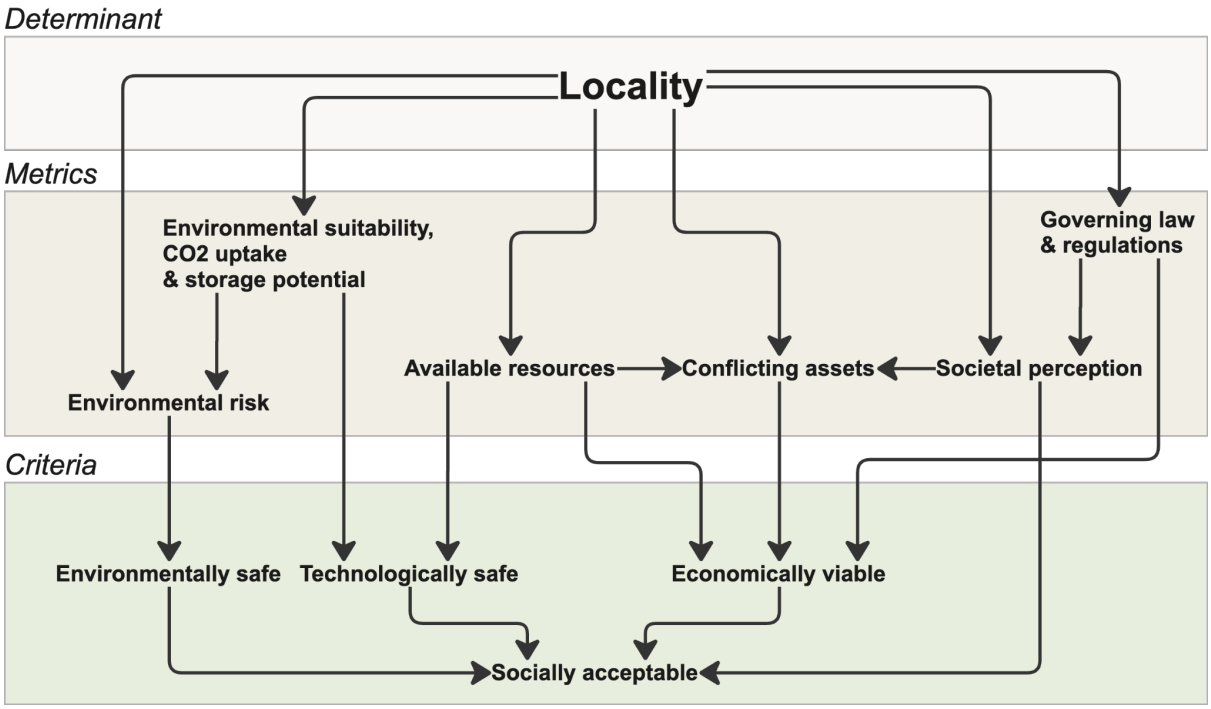


Figure 2. **Dependencies of relevant metrics and associated criteria on locality as the determining factor for viable CDR.**

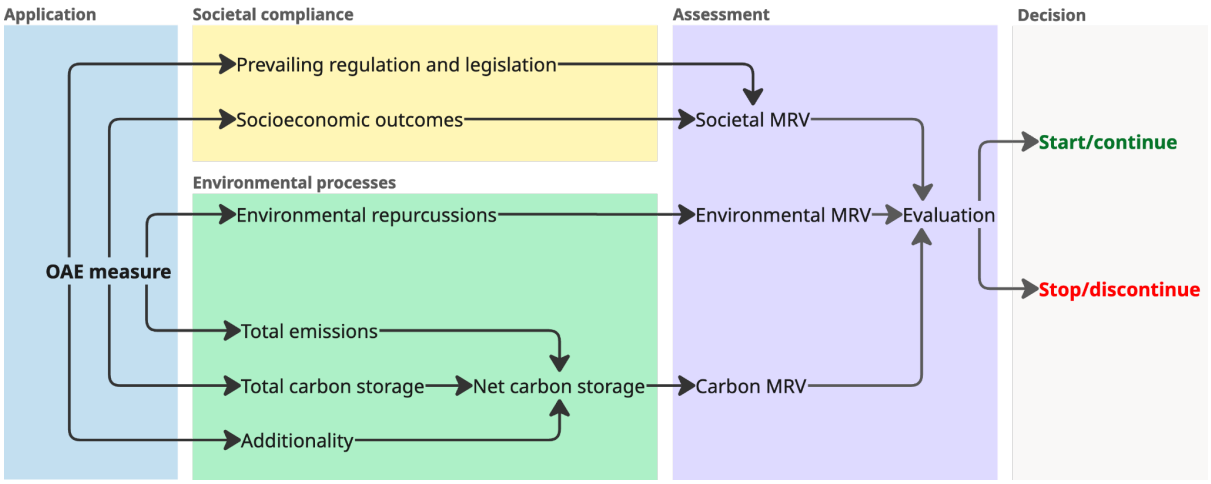


Figure 3. **Overview of societal and environmental metrics and assessment schemes that may lead to the determination of viable OAE measures.**

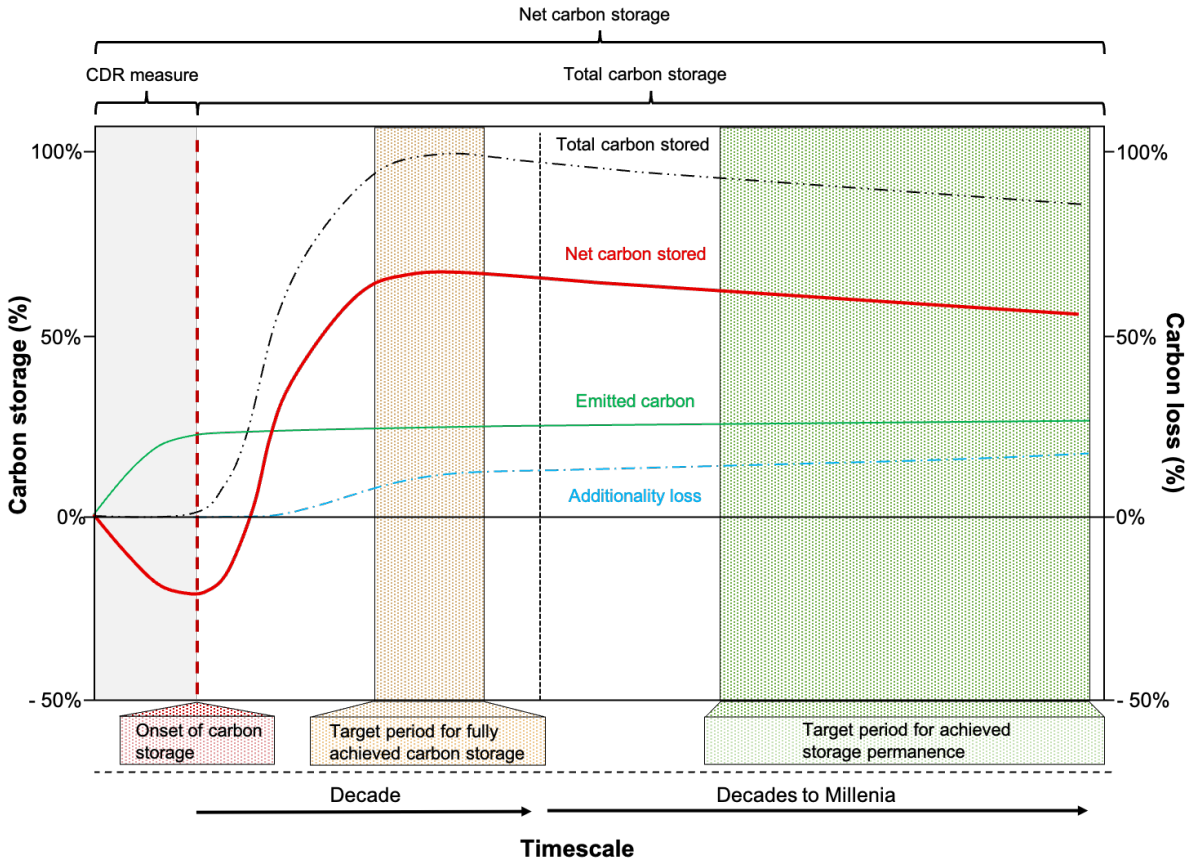


Figure 4. **Conceptual overview of the development of net carbon storage via an OAE measure in relation to the achieved total carbon storage, losses through additionality and carbon emitted during the process.** The overview implies the onset of carbon storage is achieved with the reduction of atmospheric carbon and not the claim of a future removal. The outcome of achieved carbon storage and achieved storage permanence varies with the chosen timepoint for its completion.

Tables

Table 1. **List of minerals commonly considered for OAE.** Values adapted from Renforth & Henderson 2017. Note that the theoretical CO_2 removed per tonne mineral represents the stoichiometric maximum, while experimental results show that actual CO_2 uptake is lower (e.g. 84% of the theoretical value in an experimental setup for olivine; Montserrat et al. 2017).

Mineral typ	**Molecular formula**	**Main alkalinity reaction (simplified)**	**Theoretical CO_2 removed per tonne mineral**
Calcium carbonate (limestone)	$CaCO_3$	$CaCO_3 + CO_2 + H_2O \rightarrow Ca^{2+} + 2HCO_3^-$	≈0.44 t CO_2
Calcium oxide (quicklime)	CaO	$CaO + 2CO_2 + H_2O \rightarrow Ca^{2+} + 2HCO_3^-$	≈1.57 t CO_2
Calcium hydroxide (portlandite)	Ca(OH)2	$Ca(OH)_2 + 2\ CO_2 + H_2O$ -> $Ca^{2+} + 2HCO_3^-$	≈1.19 t CO_2
Magnesium orthosilicate (olivine endmember forsterite)	Mg_2SiO_4	$Mg_2SiO_4 + 4CO_2 + 4H_2O \rightarrow 2Mg^{2+} + 4HCO_3^- + H_4SiO_4$	≈1.25 t CO_2
Magnesium hydroxide (Brucite)	$Mg(OH)_2$	$Mg(OH)_2 + 2CO_2 \rightarrow Mg^{2+} + 2HCO_3^-$	≈1.51 t CO_2

Table. 2. **Overview of microbial processes leading to alkalinity production and resulting secondary mineral precipitates.** *While both FeS and FeS_2 formation consume alkalinity in the mineral formation step (−2 mol per mol Fe^{2+}), their net outcomes when fully coupled with sulfate reduction differ. FeS yields a net of 0 mol alkalinity (+2 mol from sulfate reduction, −2 mol from FeS formation). FeS_2 additionally incorporates a second mole of H_2S ($FeS + H_2S \rightarrow FeS_2 + H_2$), requiring a second round of sulfate reduction (+2 mol alkalinity) while releasing H_2 rather than H^+, incurring no additional alkalinity cost. The net outcome for FeS_2 is therefore +2 mol alkalinity per mol formed. However, these net outcomes represent a benefit only where Fe^{2+} would otherwise have undergone oxidation in oxic conditions ($Fe^{2+} \rightarrow Fe(OH)_3$, −2 mol alkalinity per mol Fe^{2+}); where Fe^{2+} remains permanently under anoxic conditions without contact with oxygenated water, no alkalinity is gained relative to the anoxic baseline (see section 4.6.1.2.).

Microbial process	Net microbial effect on alkalinity	Mineral precipitates	Does Mineral Formation Consume Alkalinity?	Study
Ammonification	↑ Increases	CaCO3	Yes	Krause et al. 2018

Denitrification	↑ Slight Increase	$CaCO_3$ (calcite, aragonite)	Yes	Thomas et al. 2009
Sulfate Reduction	↑ Increases	FeS, FeS_2, $CaCO_3$	Yes*	Thomas et al. 2009, Geerts et al. 2025
Unknown	↑ Increases	Clay minerals	Yes	Buey et al. 2023

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