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Uncertainties in the application of enhanced rock weathering for climate-change mitigation

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Abstract

Enhanced rock weathering (ERW) on agricultural soils is under consideration as a long-term carbon dioxide removal (CDR) strategy. In this Perspective, we evaluate uncertainties related to ERW around feedstock availability, plant-soil system impacts, CDR efficiency along the land-ocean continuum and socio-economic considerations. The composition of (ultra)mafic rocks places constraints on the availability of suitable feedstock when considering their potential for CDR and toxic element contents. For ERW application at scale, dedicated mining for suitable feedstock appears unavoidable. ERW can positively and negatively impact soil structure, hydrology, and overall carbon and nutrient cycles, and so optimal ERW will require site-specific assessment of effective CDR and mitigation of potential negative impacts. Additionally, the fate of weathering products along the land-ocean continuum in rivers remains poorly constrained, which is a challenge for verifying successful CDR. The socio-economic effects and constraints of ERW regarding financing and risk responsibility are also uncertain. Ultimately, large-scale ERW deployment appears limited by substantial challenges throughout its application, from its initial setup to final CDR. Future research prioritising site-specific assessments, long-term monitoring along the land-ocean continuum, and system modelling to constrain uncertainties and address socio-economic factors is needed to ensure that ERW deployment is effective, equitable, and sustainable.

Introduction

Combating ongoing and future climate change will require a drastic reduction in the emission of anthropogenic greenhouse gases such as carbon dioxide (CO_2)[1]. Beyond this, it is increasingly recognised that additional measures involving the durable capture and storage of existing atmospheric CO_2 — so-called atmospheric carbon dioxide removal (CDR) strategies — will be necessary to meet internationally agreed goals for carbon neutrality[2–4]. In addition to technological solutions such as direct air capture, there exist several pathways that achieve CDR by manipulating natural systems, for example increasing soil organic carbon (SOC, total organic carbon present in soils derived from plants, soil fauna and microbial residues) and actively managing forests and peatlands. Such nature-based CDR pathways have entered an economic carbon market that enables companies or individuals to buy carbon credits to compensate for emissions[5].

One nature-based CDR pathway that has been proposed to exhibit large CDR potentials is terrestrial enhanced rock weathering (ERW). ERW involves the application of finely ground silicate-rich rock (mafic and ultramafic) on soils to facilitate weathering of the minerals by atmospheric derived carbonic acids (**Box 1**). At its core, ERW aims to remove the CO_2 produced by anthropogenic activities by converting it into dissolved inorganic carbon (DIC, water-present inorganic carbon species such as carbon dioxide, bicarbonate and carbonates) and, ultimately, into stable secondary carbonates in marine environments [6–10]. The newly produced weathering products act to increase ocean alkalinity, which is the capacity of the ocean to buffer pH changes and regulate interactions between CO_2 and the oceans. Co-benefits thus include combating observed ocean acidification[11], as well as soil improvement and inorganic fertilization in agricultural settings owing to the release of rock-derived nutrients[12–16].

The potential to accelerate the geological carbon cycle and increase CDR by ERW was identified in the 1990s[17] and the first experimental[18–20] and modelling[8] efforts emerged in the early 2020s. Progress has been made towards widespread ERW use and application. For example, overviews of CDR measurement methods in soils[21]; relevant accounting techniques[22–24]; new industry standards[25]; and a holistic community-led framework for CDR quantification by non-profit organisations, such as Cascade Climate[26], have been developed. Despite these advances, several potential limitations of ERW have been raised[27,28], including critical assessments of its CDR potential[29,30], silicate rock powder impacts on soil functions (for example, soil fertility and water holding capacity)[31,32], biogeochemical effects (for example, toxic element release and nutrient cycling)[15,30,33], and unintended consequences for aquatic systems (for example, alkalinity changes)[34]. Multi-year field-scale ERW trails require considerable research investment[28,35] and the quantification of net CDR remains challenging[28]. Overall, the intergovernmental panel on climate changes (IPCC) currently lists the readiness of ERW as “low to medium” (technology readiness level 3-4) due to a missing understanding of long-term impacts, co-benefits, spillover effects, costs and removed emissions assessments[1].

In this Perspective, we discuss the limitations and impacts that arise at multiple stages of the ERW process. After introducing ERW as a weathering process, we highlight resource limitations on suitable feedstocks, in terms of weathering potential and their possible toxicity, based on available global rock chemistry data[36]. We then outline several potential physical and chemical impacts on plant-soil systems, and discuss possible CDR efficiency reduction along the land-ocean continuum. Finally, we discuss socio-economic considerations and provide recommendations for future research if the current ERW movement is to be considered a large-scale CDR technology in agricultural systems.

Rock weathering processes

ERW builds on the principle of silicate rock weathering that occurs naturally and is part of the inorganic carbon cycle. This section provides an overview of the geochemical principle of silicate rock weathering, how CDR is reached and the confounding effects of carbonate weathering.

Silicate weathering processes

By releasing base cations such as calcium (Ca^{2+}), magnesium (Mg^{2+}), sodium (Na^{+}) and potassium (K^{+}), the weathering of silicate rocks on land generates alkalinity and promotes the conversion of atmospheric CO_2 into DIC[37–39] (mainly as bicarbonate, HCO_3^{-}; **Box 1**). Once formed in soils, weathering products are eventually transported to the oceans, where they contribute to the marine alkalinity reservoir. There, DIC can either remain stable as dissolved species or be involved in the precipitation of carbonate minerals such as calcite ($CaCO_3$). By transforming carbon from CO_2 to DIC and carbonates, weathering regulates Earth's climate over geologic timescales[37–39]. However, natural weathering fluxes are small (~0.4 Gt CO_2 yr^{-1})[38,40,41] compared to annual anthropogenic CO_2 emissions (~40 Gt CO_2 yr^{-1})[42] as these processes are restricted by kinetic limitations (for example, temperature, CO_2 partial pressure) and supply limitations (for example, water and mineral surface area)[43].

ERW practices aim to increase CO_2 fluxes associated with weathering through addressing the above supply limitations, which in practical terms can be achieved by distributing finely ground silicate rock powder to soils over large areas (**Box 1**). Because DIC and — especially — carbonates are long-lived, CDR achieved by ERW should be durable over several millennia[6] , substantially longer than conventional CDR methods such as afforestation, SOC sequestration and peatland management[44]. It was proposed that ERW deployment with an annual application of 40 t ha^{-1} of basaltic rock on agricultural soils could result in CO_2 uptake of 0.16–0.49 Gt CO_2 yr^{-1} by 2070 in the continental United States alone[45]. Global ERW-based CDR potentials ranging from 2–95 Gt CO_2 yr^{-1} have been discussed in the literature, with this variation largely depending on the silicate mineral considered as feedstock[8,9,46].

Box 1: Chemical principle of enhanced rock weathering

This Box outlines the general steps of in-situ carbon dioxide removal (CDR) from silicate weathering, both natural and enhanced (see figure).

Bicarbonate formation

Bicarbonate (HCO_3^-) is generated when atmospheric CO_2 naturally partitions into water to form carbonic acid, which can then deprotonate (equation 1).

Silicate mineral weathering

Generated protons cause dissolution of silicate minerals, which releases base cations (Ca^{2+}, Mg^{2+}, Na^+, and K^+) that are then replaced by protons. For example, wollastonite dissolution at circumneutral pH produces Ca^{2+} and orthosilicic acid[6,15] (equation 2). By consuming protons and driving equation 1 towards the right, weathering effectively consumes CO_2 and produces alkalinity (the capacity of a fluid to buffer acidification) in the form of HCO_3^-.

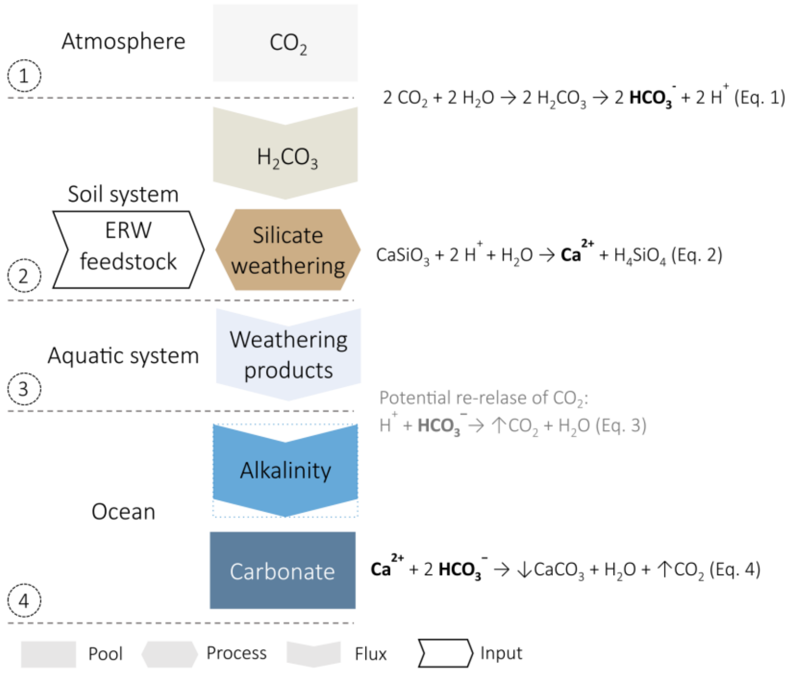
Transport of weathering products

A fraction of HCO_3^- and released base cations are transported from soils to groundwater and inland waterways, finally reaching the oceans, where they increase marine alkalinity. However, if additional proton equivalents are derived from non-carbonic acid sources (such as sulphuric or nitric acid), CO_2 can be re-released and CDR is reduced (equation 3).

Carbonate formation

Seawater exhibits naturally high alkalinity and by providing additional alkalinity, ERW further increases the carbonate saturation state, promoting precipitation and resulting in long-term CDR (equation 4).

Considering the stoichiometry of equations 1,2 and 4 ERW followed by carbonate precipitation would remove one mole of atmospheric CO_2 per mol of silicate mineral weathered[6]. Thus, carbonate precipitation reduces CDR potential by 50% relative to the scenario where all HCO_3^- remains in solution.



Confounding carbonate weathering

In addition to silicates, many feedstocks being considered contain trace amounts of the carbonate minerals (secondary carbonates)[47,48]. Carbonates exhibit much faster weathering rates that most silicates[6], to the point that pure carbonate rocks (such as limestone and dolostone) have been proposed as feedstocks within an ERW framework[49]. However, unlike silicate weathering, carbonate dissolution does not result in CDR if secondary carbonates re-precipitate in rivers and oceans[6,49–53]. This difference is because carbonate dissolution consumes only one mole of CO_2 per mole of divalent base cation released — compared to two during silicate weathering (**Box 1**) — and therefore CO_2 will be re-released by carbonate precipitation over the residence time of base cations and DIC in the ocean (~10,000 yr)[34]. Partially for this reason, in 2006 the IPCC restricted the CDR potential by agricultural liming with limestone and dolomite and defined default emission factors to be 0.12–0.13 ton CO_2-C equivalents per ton liming material[54], which is a metric measure to compare to other greenhouse gasses using gas specific global-warming potentials. These factors have been debated for over two decades and the potential CDR of liming materials remains uncertain[55–57]; regardless of the exact factor, however, strict differentiation between carbonate and silicate weathering is paramount when accounting for CDR by ERW[47,48].

Feedstock constraints

Deployment of ERW for CDR requires the supply of suitable material, with silicate-rich mafic and ultramafic rocks with high weathering rates commonly proposed as ERW feedstocks. This section briefly highlights different feedstocks and discusses feedstock chemical variability as it relates to CDR potential and possible toxicity, focusing on natural silicate-rich mafic and ultramafic rocks.

Types of feedstocks

Global assessments of the availability of silicate-rich feedstocks have considered broad lithological categories[8,46]. These assessments have identified mafic and ultramafic rocks as scalable sources[47,58,59]. However, individual lithologies reveal considerable heterogeneity regarding weathering kinetics and CDR potential[60,61]. Alternative sources, such as artificial alkaline materials (for example, steel slag and demolition waste), also hold some CDR potential, but many contain carbonates, which can limit CDR potentials[50,62]. Besides heterogeneity in CDR potentials, toxic element contents vary greatly among mafic and ultramafic feedstocks[28,63]. Further, alternative waste materials can contain high pollutant contents[57,64,65] and their use in agriculture is restricted in many countries[66,67]. Overall, regulatory limits for potential toxic elements in soil amendments exist for many countries.

ERW is often discussed in an agricultural context, where existing infrastructure can be leveraged to spread feedstock material[15]. Additionally, silicate-rich mafic and ultramafic rocks are considered for large-scale ERW deployments[6,58].

Constraints on suitable feedstock material

Suitable ERW feedstock must exhibit sufficiently high reactivity. Average CO_2 uptakes of 0.30 t CO_2 t^{-1} rock and 0.80 t CO_2 t^{-1} rock for mafic and ultramafic rocks, respectively, are often used when estimating overall gross CDR potentials[9,45,46,58,68,69]. Based on life-cycle assessments that account for ERW-associated emissions from transport and powder preparation, a minimum value of 0.30 t CO_2 t^{-1} rock has been proposed as a threshold for efficient CDR with silicate-rich rocks[70]; rocks with CDR_{max} below this threshold (where CDR_{max} represents the maximum CO_2 that can be captured per ton of rock material, with adjustments for downstream CDR reductions[68]) are likely of limited suitability for large-scale implementation as a net CDR would only be possible in close geographic proximity to the source mine.

In this Perspective, the limitations of ERW feedstock suitability for CDR are identified using subaerial rock data from a whole-rock geochemical database (compiled from the Geochemistry of Rocks of the Oceans and Continents repository GEOROC)[36] to constrain the wide compositional range of mafic and ultramafic silicate rocks (**Supplementary Fig. 1 and**

2). The theoretical CDR_{max} are calculated for 30 mafic and ultramafic rock types based on the stoichiometry of base cations (Ca^{2+}, Mg^{2+}, Na^{+}, and K^{+}), assuming complete silicate mineral dissolution (**Supplementary Note 1**).

In addition to CDR potentials, the presence of toxic elements in feedstocks is a limitation on suitable rock material. Nickel (Ni) is a common toxic element present in the GEOROC observations, and is used to assess potential toxicity. Ni is essential for plants and organisms in low concentrations, potentially having fertilizing effects, but becomes toxic at higher levels, harming plant health and productivity[71,72]. Consequently, as for other potentially toxic elements, Ni contents in amendments applied to agricultural soils are strictly regulated in some countries. For example, liming and inorganic amendments in the European Union are not allowed to exceed 90–100 ppm Ni (European regulation 2019/1009)[73]. Exceeding this threshold would likely induce risks, possible adverse ecological effects and negative economic consequences for farmers; therefore, a 100 ppm threshold can be applied as a benchmark

Ni contents can also serve as a proxy for chromium (Cr; **Fig. 1a**). Hexavalent Cr — the bioavailable species — is highly regulated with European Union regulatory limits at 2 ppm for inorganic amendments in agricultural soils[73]. However, speciation data is not available in the GEOROC database so it is can only be used to consider total Cr content. Although the release of Cr species will depend on the feedstock and local conditions, silicate rock weathering is the main source of hexavalent Cr in soils and groundwater[74]. The potential formation of hexavalent Cr after ERW feedstock application, as well as bioavailable speciation of Ni and other toxic elements[75], should be further explored to better estimate risks of large-scale CDR deployments.

Overall, 13% of all reported subaerial mafic and ultramafic rock samples in the GEOROC database exhibit a CDR_{max} greater than 0.30 t CO_2 t^{-1} rock and a Ni content less than 100 ppm (29% of subaqueous samples; **Fig. 1b** and **Supplementary Figure 3**). On average, mafic rocks exhibit a lower concentration of toxic Ni contents than ultramafic rocks, but have lower CDR efficiencies due to slower dissolution rates[7,15,76].

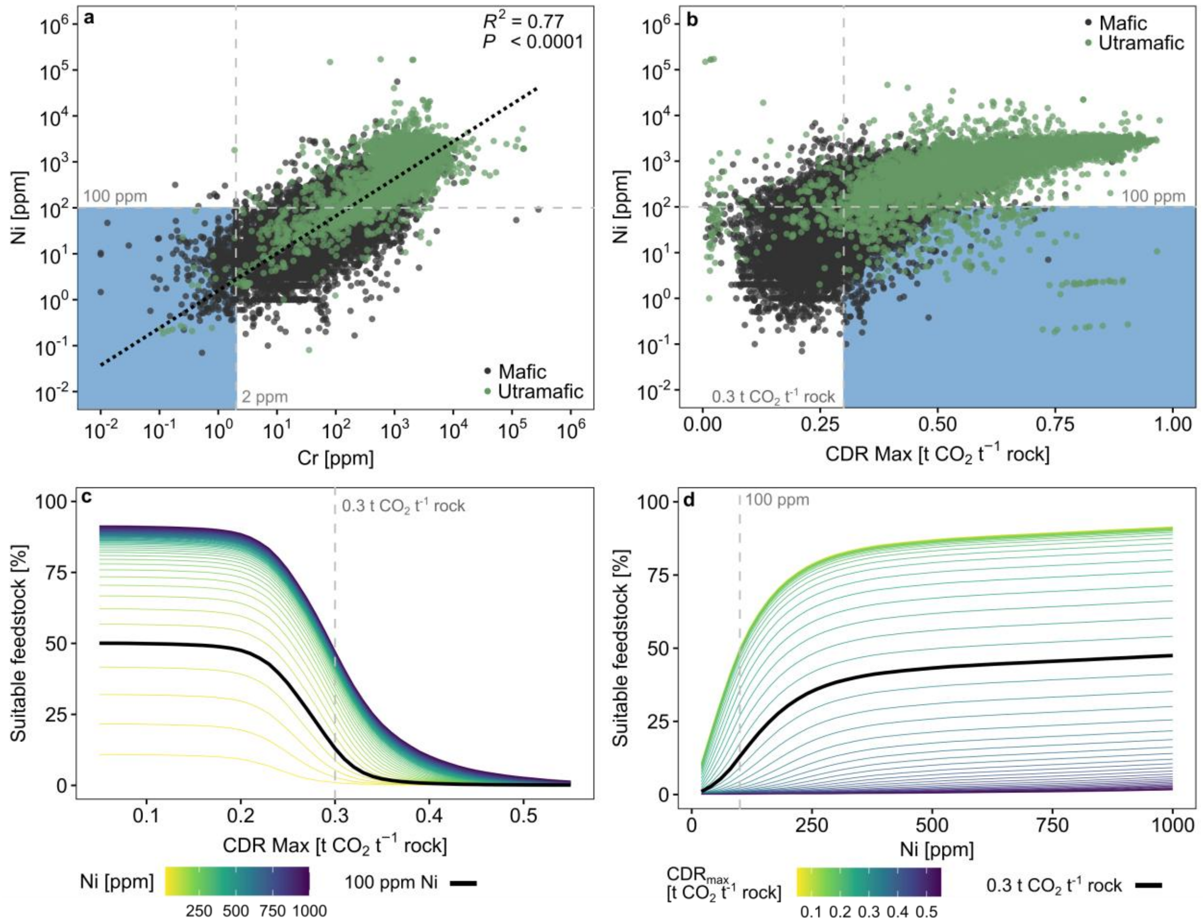


Figure 1 | Global mafic and ultramafic feedstock composition. a, Total Ni and chromium (Cr) content (both log scale) of mafic and ultramafic rocks, including an ordinary least squares linear regression. Dashed grey lines indicate the European Union Ni threshold of 100 ppm and hexavalent Cr threshold of 2 ppm (here only total Cr is shown) for inorganic amendments applied to agricultural soils[46,69,73]. **b**, Maximum carbon dioxide removal potential (CDR_{max}) and total Ni content (log scale) for subaerial mafic and ultramafic rocks. Dashed grey lines indicate a CDR_{max} benchmark of 0.30 t CO_2 t^{-1} rock and the European Union Ni threshold of 100 ppm[46,69,73]. Shaded areas in **a** and **b** represent potentially suitable enhanced rock weathering (ERW) feedstocks. **c,** The fraction of suitable ERW feedstock as a function of CDR_{max} for a range of Ni thresholds (coloured contours; 20–1000 ppm). **d**, as in **c** but as a function of Ni thresholds for a range of CDR_{max} values (coloured contours; 0.05–0.55 t CO_2 t^{-1} rock). All data are derived from the GEOROC database[36] (109,389 observations including Ni and CDR_{max}; and 103,889 observations including Ni and Cr). All considered rock types are listed in Supplementary Note 1; global distributions of included observations as well as additional element cross-plots are shown in **Supporting Information Figures 1-3**.

A sensitivity analysis further reveals that small changes in CDR_{max} and Ni thresholds have a large impact on the fraction of suitable rock observations (**Fig. 1c**). For example, lowering CDR_{max} to 0.25 t CO_2 t^{-1} rock increases the suitable rock fraction to 36%, while raising it to 0.35 t CO_2 t^{-1} rock reduces suitable material to only 2% of all observations. Higher CDR_{max} might be required when higher production and transport emissions need to be compensated. Similarly, removing the Ni limitation while holding $CDR_{max} \geq 0.30$ t CO_2 t^{-1} rock increases the suitable rock fraction to 48% (**Fig. 1d, Supplementary Table 1**). The GEOROC database only includes single observations for each quarry or natural outcrop and is biased to overrepresent the Northern Hemisphere and generally underrepresents tropical regions. Although the assessment of GEOROC data begins to reveal limitations on suitable feedstocks, extrapolating to global quantities is challenging due to large uncertainties given the lack of volumetric data.

Feedstock supply

The possibility of using existing by-products from mining operations as feedstock material for ERW has been widely considered[60,77,78]. However, assessments of whether by-product quantities are sufficient, accounting for estimated CDR potentials by ERW, remain limited. The United Kingdom, for example, produces approximately 3.7 Mt of potentially suitable rock powder annually[79]. However, to meet its national CDR goal of 6–30 Mt CO_2 yr^{-1} using ERW alone[10], it would require over five times this amount, assuming an average realised CDR of 0.30 t CO_2 t^{-1} rock. Although some national-level data exist, global rock by-product data are scarce, hindering an accurate estimate of the ERW potential of quarry fines. The potential of ERW using mining by-product can therefore only be roughly estimated.

The potential of ERW using mining by-product can only be partly estimated using available data. The European Aggregates Association[80] reports that ~45 Gt yr^{-1} of aggregate, rock grains or fragments of all sizes used as resource material of all rock types are produced globally. Around 47% of this total comes from crushed rock (with this fraction derived from that reported for EU, UK, and EFTA countries), with the rest from sand and gravel deposits. This value corresponds to ~21 Gt yr^{-1} of crushed rock produced globally, of which ~20% is assumed to be fine-grained material[79,81], resulting in ~4.2 Gt yr^{-1} of rock powder. By considering that mafic and ultramafic igneous rocks constitute ~9% and 1% of solid rocks exposed at Earth's surface[75,82], a total quarry fines by-product production of ~0.42 Gt yr^{-1} can be estimated, which would correspond to a CDR of ~0.15 Gt CO_2 yr^{-1} (assuming a CDR_{max} of 0.30 and 0.80 t CO_2 t^{-1} rock for mafic and ultramafic rocks, respectively[68]).

Conventional CDR measures such as forest and SOC management achieved ~2 Gt yr^{-1} CO_2 globally between 2013 and 2022, with technologies such as ERW contributing 0.0013 Gt CO_2 yr^{-1} (ref.[44]). Therefore, the estimate of ~0.15 Gt CO_2 yr^{-1} indicates a sizable CDR potential for mining by-products relative to current ERW deployments, albeit representing an upper bound based in CDR_{max} that does not account for potential limitations due to toxicity.

However, the estimated potential represents only ~0.4% of annual anthropogenic CO_2 emissions (40.7 Gt CO_2 yr^{-1}), ~7.5% of the lower-bound global potential proposed for CDR by ERW (2 Gt CO_2 yr^{-1}), and is substantially lower than the multi-gigaton removal potential modelled for global ERW deployment[8,46]. Scaling ERW to the gigaton-level will therefore require dedicated feedstock production in addition to increased mining and reopening of closed mines[79]. Therefore, there are urgent questions around the practical feasibility, potential environmental and social impacts, and sustainability of expanded mining operations[83]. The necessity for additional mining often goes unmentioned or is relativised as a minor expansion of an already existing supply chain[62]. ERW also competes for mafic rock materials with the construction industry, which also plans to use mining by-products for new applications[84–87], such as basalt-fibre production[88–90]. Any competition for mining by-products is likely to reduce the overall amount of feedstock available for ERW and future work including economic developments of the building industry and related fields in models for feedstock availability is required.

Toxic element accumulation

Even where no soil amendment toxicity regulations exist, the repeated application of ERW feedstock will likely be restricted due to soil environmental regulations on toxic element accumulation in topsoil, which are in place to avoid bioavailability risks[63]. For example, the commonly proposed dose of 40 t ha^{-1} of mafic rock powder annually could result in Ni and Cu accumulation at rates of at least +0.5 and +1 ppm yr^{-1}, respectively[63]. At this rate, copper accumulation would likely exceed limits on the scale of years to decades, depending on national legislation tolerances[63]. This concern is supported by some field evidence: a grassland trial detailed in a pre-print article reported substantial rises in Ni (+790%), Cr (+187%), and V (+18%) after a single application of 50 t ha^{-1} of basaltic rock[91].

The accumulation of toxic elements in soil does not always lead to toxic element accumulation in crops. For example, in a four-year field trial with 50 t ha^{-1} of annual basalt application, a large decrease in water-soluble Ni content (-8 ppm, -25%) was observed[35]. Additionally, no toxic trace element accumulation in maize was observed, which is in line with similar findings for maize[53], potato[20] and oat[92] after basalt application.

The apparent contradiction between an increase in toxic elements in soil and decreased levels of crop toxic elements can be explained by ERW's liming effect, which raises soil pH and reduces toxic element mobility[93]. As long as silicate weathering buffers pH, trace elements will accumulate in soil but remain less available for plants. Once the buffering effect is removed over time due to the weathering, trace element concentrations in soil and crops could increase, potentially exceeding pre-ERW levels, and require additional ERW applications or pH-buffering agents, such as agricultural lime, to maintain low bio-availability of toxic elements. Ultimately, the long-term fate of ERW feedstock-derived toxic elements in soils, plants, and aquatic ecosystems remains largely unknown[15,20].

Effects of ERW on plant-soil systems

The application and subsequent gradual weathering of silicate rock powder on and in soils can cause a cascade of physical, chemical, and biological reactions, each potentially impacting soil fertility[15,28]. This section addresses potential effects on soil structure and hydrology, biogeochemical cycles, SOC cycling and the alteration of weathering rates in soils.

Soil structure and hydrology

The effects of the large-scale application of rock-powder on soil should be investigated further. For example, necessary doses suggested for ERW are generally between 10–40 t ha^{-1} of rock[21,35,45], with >100 t ha^{-1} of rock proposed in some cases[94]. However, most deployments outside of research have applied lower doses of ~10 t ha^{-1}. The large-scale application at higher rates could cause soil compaction[15,95,96], which has been identified as a major threat of soil fertility and yields and is already observed as a result of other soil management strategies[97–99].

Vertical transport of ERW powder in soils might induce temporary clogging of soil macropores, as is sometimes observed during liming operations[100,101]. Such clogging could become especially important when considering that suggested ERW application rates can be one-to-two orders of magnitude higher than those currently used for liming. Clogging limits fluid and gas exchange between soil and the atmosphere, which would be detrimental both to soil fertility (as this exchange is vital to soil organisms and crops[102]) and to the capability of ERW to maintain optimal weathering rates (**Box 1**)[103]. ERW feedstock grain size will likely be a crucial parameter affecting clogging[61]. Further, applied ERW powder can be either ploughed into conventional tilled fields or left on the surface in the case of no-till fields, grasslands, and forests; the magnitude of clogging will likely depend on initial soil properties and the application mode.

ERW might have beneficial effects for soil health. Concomitant Ca^{2+} release from ERW feedstock can favour formation of soil aggregates due to flocculation[104], as is commonly observed after liming operations in acidic soils[105,106]. Further, weathering of most rock powders leads to the formation of secondary minerals such as clays, known for their water-holding capacity[107,108], which can be substantial in sandy soils.

Overall research into the impact of ERW on soil physical properties remains scarce and providing generalised application guidelines for rock powder thus remains challenging. In an early pot experiment conducted in 1986, the immediate effects of different application rates for silt-sized and sand-sized basalt powder (ranging from 150 to 600 t ha^{-1} of rock, or 5–20 wt. %) on water holding capacity, soil porosity, and shear strength were evaluated[109]. Although results were inconsistent across doses, silt-sized basalt reduced sandy soil water holding capacity by ~5% and sand-sized basalt decreased the shear strength of clay soils by ~10%, facilitating root penetration and water infiltration[109]. By contrast, a one-year field trial with no tillage conducted in Switzerland applying 20 t ha^{-1} of basaltic rock powder led to no substantial changes in soil texture, structural quality, bulk density, aggregate stability, or water infiltration rates[31].

Soil pH, biogeochemistry, and nutrients

A soil pH between 5 and 7 is commonly maintained in agricultural soils through liming operations[110] to optimise plant growth, and crop yields, and minimize nitrous oxide and methane greenhouse gas emissions[111,112]. By producing alkalinity (**Box 1**), ERW of silicate rocks might raise soil pH and is thus starting to be considered as an alternative to lime[113,114]. This increase has been observed with mafic and ultramafic feedstocks on acidic soils[14,23,35]. However, dissolution rates of (ultra)mafic silicates are orders of magnitudes lower than those of carbonate-rich materials[6], which are commonly used for pH maintenance. The potential for silicate rock applications to manage and maintain soil pH at site-specific and crop-specific optimum conditions is largely unknown.

For acidic soils, increasing pH will shift the variable charge of clays, oxides, and organic compounds towards greater electronegativity[115], which can be beneficial to mobilise phosphorus, immobilise heavy metals, and replenish rock-derived base cations for plant uptake[15]. However, research into ERW for low-pH soils, particularly in the tropics[15,33,116,117], is scarce. In temperate regions, it is evident that some soils with pH below 5.2 cannot be suitable for efficient CDR due to the weaker weathering efficiency of H_2CO_3 compared to other acids present in soils[118]. By contrast, soils that are already alkaline ($pH > 7$) might be minimally impacted by the additional alkalinity provided by ERW[31], and CDR potential in such soils is generally low as mineral dissolution rates decrease with increasing pH[119]. Furthermore, silicate dissolution will only occur after the dissolution of carbonates that are already present in higher pH soils (following the “paedogenic cascade” of pH-buffering capacity)[120]. Therefore, soil pH will be a crucial criterion for selecting optimal soils for ERW applications.

Rock powder itself can act as a source of micronutrients (such as zinc, molybdenum, iron, manganese, and copper) and macronutrients (Ca, Mg, K, silicon, and phosphorus)[12,35,92,121]. Such inputs might be especially important in acidic or highly weathered soils that are depleted in cations and micronutrients[122,123]. However, tropical ecosystems, where CDR potentials are likely highest[9], are described by an efficient nutrient recycling loop between soil and aboveground vegetation[124,125]; any pH shifts might cause disturbances that greatly affect overall ecosystems functionality. ERW applications in tropical, nutrient depleted environments should therefore focus on agricultural systems where nutrient replenishments are most likely to exhibit a positive impact owing to the general lack of fertilizer available to subsistence farmers. In addition, field trials that introduced rock-powder have reported positive effects on nutrient contents and soil fertility, including improved yields[13,16,94,95] and plant resilience[15] in different climatic systems.

It has been proposed that phosphorus addition to soils from rock powder can increase plant productivity and enhance SOC sequestration — a potential pathway for additional CDR by ERW[9,33]. Although phosphate-containing rocks such as apatite are suitable P fertilisers, phosphorus availability and potential synergistic effects during silicate-rock weathering have not been widely investigated under field conditions[15,60].

Soil organic matter cycling

Along with dissolved weathering products, silicate rock powder introduces new mineral surfaces into soils. Mineral surfaces are known to preserve SOC in the form of mineral-associated organic carbon (MAOC). Although some components of this pool can be more dynamic[126–129], MAOC is generally considered as a stable and slowly cycling SOC reservoir[130,131]. Despite this importance, reported impacts of ERW on MAOC are mixed. For example, a 2-year field trial found that the application of basalt powder to cropland soils reduced the formation of stabilised MAOC compared to controls where no basalt powder was applied [132], whereas an increase in MAOC formation in forest soils following wollastonite application led to the conclusion that rock powder application increases SOC stabilisation[133].

MAOC formation is governed by physicochemical adsorption and desorption processes that depend on soil mineral and organic matter composition, as well as microbe-mediated organo-mineral interactions[134–136]. All of these factors can be impacted by ERW, depending on soil and feedstock properties. For example, interactions between cations such as Ca^{2+} and organic matter are known to enhance SOC stabilisation[137,138], which in turn increases the retention of ERW-derived base cations within soils and thus reduces the fraction of total base cations that are transported into the aquatic system.

Finally, shifts in pH caused by ERW and subsequent nutrient release can accelerate SOC decomposition and increase soil CO_2 emissions[139,140]. Soil pH controls the adsorption and desorption of SOC on soil mineral surfaces[134], microbial traits such as carbon use efficiency[141] and overall microbial activity and composition[142]. The precise effect of ERW on SOC shifts and overall soil organic matter dynamics will vary depending on soil type, land use, climate and rock powder composition. For example, tropical peatlands could be unsuitable for ERW application as the pH increase can destabilise SOC, potentially re-releasing CO_2 and reducing the CDR potential of ERW by 18–60%[143]. Therefore, calculations of net CDR potential must carefully consider how soil organic matter and nutrient cycles are influenced by the application of ERW feedstocks and the changes in inorganic and organic carbon to account for the complex shifts in the biogeochemical responses of soil, which depend on its initial properties.

Soil and landscape constraints

Estimating the effective or in situ CDR potential of ERW is difficult as soils are complex systems that integrate physical, chemical, and biological processes. Small-scale variations in soil properties, grain size of applied rock powder, and measurement type and location can cause large variability in observed weathering rates[69]. The degree to which organic acids from plants[144], soil microfauna and mesofauna (for example, nematodes[145]) contribute protons that dissolve silicate minerals is uncertain. Further, in agricultural systems, ammonium-base nitrogen fertilization results in acidification by nitrification[146].

Weathering can be governed by stronger acids than carbonic acid, such as mineral-surface-bound protons and organic acids[116], particularly in tropical regions where soils are generally acidic. Similar effects have been found in temperate soils with pH below 5.2 after 30-month field trials[118]. This phenomenon will require a correction for non-carbonic acid derived weathering when estimating in-situ CDR, although silicate dissolution by organic or nitric acids in agricultural systems still generates alkalinity as base cations that can be transported to the ocean continuum. The associated downstream shifts of carbonate equilibration could thus still drive CO_2 uptake by gas exchange and HCO_3^- generation at the landscape scale, even if these processes are difficult to quantify at the plot scale.

Long-term CDR relies on the effective transport of base cations and HCO_3^- to oceans, where they can precipitate as carbonates (**Box 1**)[6]. Retention of weathering-derived base cations in soils through soil carbonate precipitation, incorporation into exchangeable pools through absorption, or incorporation into secondary clay minerals such as smectite can influence the carbonate system equilibrium and result in the re-release of CO_2. Additionally, harvesting in agricultural systems will remove base cations that have accumulated in harvest biomass, thus limiting the fraction of weathered cations available for transit in the land-ocean continuum.

Weathering might be impacted by passivation effects. Weathering of applied rock material generates authigenic weathering products that can dynamically impact weathering rates and CDR potentials. For example, the formation of secondary iron (oxyhydr)oxides such as ferrihydrite and goethite can coat applied rock powders and reduce their dissolution rates[62]. Further, weathering rates decrease over time and with increasing solute concentrations and rock-fluid ratios, owing to a continuous loss of reactive surface area[147]. The long-term dynamics of such passivation effects on CDR potential remain unknown and require more site-specific research[69].

Finally, dynamic shifts in soil moisture affect weathering rates. Estimates from the United Kingdom indicate that in-situ basalt dissolution is lowered by a factor of 5–25 compared to optimal rates due to slow water-limited dissolution rates[19]. Current empirical models agree with this observation, suggesting that weathering rates under realistic conditions can be up to two orders-of-magnitude lower than those predicted under optimal conditions[148]. This finding highlights that the achievable CDR potential in soils over relevant timescales, such as years to decades, can be orders of magnitude below the theoretical maximum[116,118,149].

ERW along the land-ocean continuum

The efficiency of long-term CDR by ERW relies on the transit of weathering products to the ocean to feed ocean alkalinity and ultimately form carbonate. This section discusses potential uncertainties and how CDR effects can be reduced during the transport of base cations and HCO_3^- from the site of application to oceans.

Impacts on river and ocean alkalinity

As the planet warms, Earth's oceans and water bodies are expected to become more alkaline due to an increase in natural weathering processes, potentially disrupting the ocean alkalinity thermostat[150]. Any ERW will further increase alkalinity compared to background conditions (**Box 1**). Because natural weathering by silicate and carbonate rocks currently removes 0.3-3 Gt CO_2 yr^{-1} (ref. [41]), successfully achieving the lower-bound global CDR target for ERW (such as 2 Gt CO_2 yr^{-1})[46] would increase total silicate weathering fluxes 1.6-to-6-fold. Meeting proposed CDR targets would thus greatly increase alkalinity transport[151], far exceeding current natural levels, and amplify the rise expected from natural weathering on a warmer planet.

As a comparison, paleoenvironment reconstructions indicate that natural silicate weathering flux increased by ~34–42% during the Paleocene-Eocene Thermal Maximum hyperthermal event[152] — often considered an ideal geologic analogue for anthropogenic warming — relative to baseline levels. This proportional increase in weathering flux is nearly a full order of magnitude smaller than that required to achieve proposed CDR scenarios using ERW. Such large-scale changes in alkalinity and base cation concentrations might affect entire fluvial and marine ecosystems, potentially with negative effects[153] for which the feedbacks are unclear. Although increased alkalinity might benefit some aspects of river systems — for example, liming-induced pH increases can improve conditions for certain fish species — abrupt or large increases might be detrimental to other aspects such as invertebrate abundance[154].

Alkalinity loss during transport

Even if proposed ERW deployments are reached, a critical uncertainty is whether additional alkalinity can be effectively transported to oceans, where it remains stable on a millennial timescale[34].

There are two main mechanisms by which alkalinity can be lost during transport. First, contribution of proton equivalent from acids other than carbonic acid shift the carbonate system toward H_2CO_3 and promote the re-release of CO_2 from solution (**Box 1**). For example, weathering of the iron-sulphide mineral pyrite in both natural and in anthropogenic acid-mine drainage systems leads to the formation of sulphuric acid and thus alkalinity consumption[155,156]. This effect might be particularly important in mine tailings and large river basins, where shallow weathering fronts and “open system” conditions can elevate pyrite weathering rates[157].

Second, increasing alkalinity levels also increase the calcite saturation index, SI_{cacl}, a value that indicates if calcite remains dissolved or precipitates in water depending on pH, alkalinity and temperature (**Box 2**), with higher values indicating a higher likelihood of carbonate precipitating. Carbonate precipitation could potentially halve CDR potential compared to the case where all HCO_3^- remains in solution (**Box 1**). Models of intermediate carbonate precipitation and DIC equilibrium changes in rivers in the United Kingdom suggest a reduction in CDR potentials by 16–27% considering an application rate of 10–50 t ha^{-1} basalt on agricultural soils[151].

Accurately modelling carbonate precipitation during transport is complicated by uncertainty in the exact SI_{calc} value at which precipitation initiates. Although $SI_{calc} > 0$ indicates calcite precipitation is thermodynamically favourable, a higher SI_{calc} is generally necessary to initiate precipitation in natural river systems due to the presence of Fe, Mg, phosphate, and sulphate, which can inhibit carbonate precipitation[158,159]. Based on global values and observations showing precipitation occurring at $SI_{calc} \sim 1$ (Ref.[160–162]), initial system models have adopted $SI_{calc} = 1$ as a threshold for carbonate precipitation in rivers[49,151]. However, this threshold is likely to vary between rivers depending on temperature and solution chemistry, particularly the concentration of precipitation inhibitors. Waters with low ionic strength and low total dissolved solids, for example, might exhibit a threshold for calcite precipitation at $SI_{calc} < 1$ (Ref. [49]). Conversely, laboratory experiments have shown that calcite precipitation is fully inhibited at $SI_{calc} = 1$ when Mg^{2+} and sulphate concentrations are elevated[159].

To explore how ERW scenarios might impact carbonate precipitation globally, the Global River Chemistry (GLORICH)[163] database can be used to calculate SI_{calc} all rivers for which sufficient data exist (n = 7,236, **Supplementary Figure 4**). ~3 % of river stations exhibited $SI_{calc} > 1$ and are thus considered supersaturated with respect to calcite, following standard thresholds (**Fig. 2a**). The sensitivity of SI_{calc} was estimated if alkalinity increased up to 10-fold compared to background weathering conditions, a possible range expected if lower-bound[46] global estimates of 2 Gt CO_2 yr^{-1} CDR by ERW are reached[38,41]. Importantly, these calculations hold pH and Ca^{2+} concentrations constant relative to modern observations (for example assuming all ERW-derived base cations are Mg^{2+}) and therefore represent a lower-bound for SI_{calc} increases (**Box 2**). Actual SI_{calc} changes will depend on additional factors such as exact feedstock chemistry.

Box 2: The calcite saturation index, SI_{calc}

SI_{calc} is a measure of a solution's tendency to dissolve or precipitate carbonates. It is a function of solution Ca^{2+} and CO_3^{2-} concentrations, temperature, and ionic strength and is defined as:

$$SI_{calc} = \log\left(\frac{\gamma Ca^{2+}\gamma CO_3^{2-}}{K_{sp}}\right) \qquad (\mathbf{5})$$

where γCa^{2+} and γCO_3^{2-} are the activities of Ca^{2+} and CO_3^{2-}, respectively, and K_{sp} is the (temperature-dependent and ionic-strength-dependent) thermodynamic constant of carbonate dissolution. In an ideal solution, $SI_{calc} = 0$ indicates equilibrium with calcite, $SI_{calc} > 0$ indicates oversaturation or supersaturation (precipitation is thermodynamically favourable), and $SI < 0$ indicates undersaturation (dissolution is thermodynamically favourable)[164]. Most natural inland waters are undersaturated with respect to calcite, as seen in the distribution of SI_{calc} values for 7,326 globally distributed river catchments[163] (**Fig. 2a**).

Despite these simplifications, these calculations reveal that the fraction of rivers supersaturated with respect to calcite would increase drastically under many ERW scenarios. For example, a 3-fold and 10-fold increase in alkalinity would lead to ~25 % and ~40 % of all measured rivers, respectively, displaying $SI_{calc} > 1$ (**Fig. 2b**). If a lower SI_{calc} threshold of 0.5 is adopted, as has been suggested for some systems[49], then these fractions increase to ~40 % and ~50 %, respectively, for the same alkalinity increases. The fraction of supersaturated rivers would remain at near-modern levels only in the case that SI_{calc} thresholds are high, near 1.5, and alkalinity increases are moderate (up to ~3-fold).

Conversely, some global-scale and regional-scale models suggest that river chemistry might not substantially impact CDR potential[165]. Simulations of basalt application in the USA suggest that dissolution of basalt (application of 10 t ha^{-1} $year^{-1}$ with dissolution of 1 t ha^{-1} $year^{-1}$) across 100 catchments (mostly <10,000 ha) results only in small reduction in CDR (<5%) over two years[166]. However, system models that incorporate higher application rates matching USA national-scale CDR potentials (160–300 Mt CO_2 yr^{-1})[45] are required to fully evaluate the potential effects on river chemistry. Overall, the poorly defined SI_{calc} threshold[165] mean that DIC longevity and downstream environmental impacts remain an uncertainty.

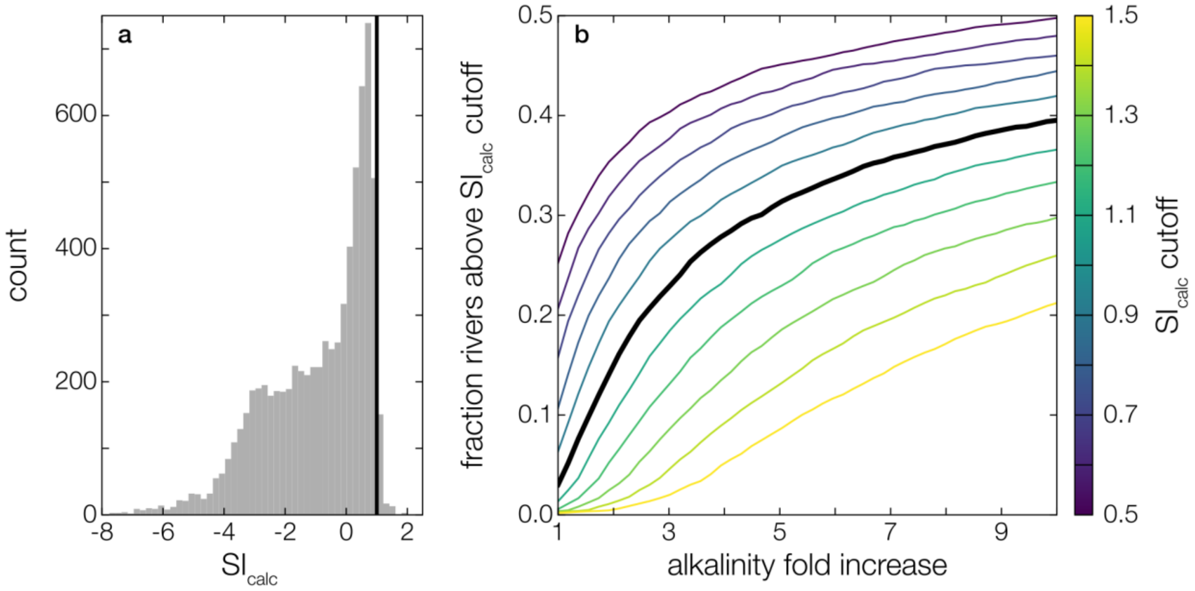


Figure 2 | Calcite saturation index (SI_{calc}) for global rivers. **a**, Observed SI_{calc} following $SI_{calc} = \log\left(\frac{\gamma Ca^{2+} \gamma CO_3^{2-}}{K_{sp}}\right)$ using compiled river-water pH, temperature, alkalinity, and ion concentration data for all available samples of the GLORICH database (173,286 individual datapoints)[163,167]. The ion activity coefficients (γCa^{2+} and γCO_3^{2-})[168] and constants of K_{sp} and DIC [169] were calculated for all available catchments (7,326) by averaging multiple observation at each station. **b**, Effects of alkalinity increase (alkalinity and Mg^{2+} charge increase up to 10-fold) on global river SI_{calc}, showing the proportion of rivers that would reach supersaturation above a given SI_{calc} cutoff between 0.5–1.5. This plot represents a minimum estimate for SI_{calc} increase because pH were held constant and all enhanced rock weathering (ERW) yields Mg^{2+} rather than Ca^{2+}; generation of Ca^{2+} would further increase calcite saturation state. The black line in **a** and **b** indicates SI_{calc} = 1. The global distribution of the rivers in shown in **Supplementary Figure 4**.

CO_2 outgassing dynamics

The fate of CO_2 released by carbonate precipitation remains highly uncertain. In rivers with a typical pH of 7.7 (**Supplementary Figure 5**)[163,167], some dissolved CO_2 can be converted to HCO_3^- depending on the availability of cations for charge balance[164,170]. In rivers with lower pH, however, this carbon will remain as CO_2, thus increasing its partial pressure in solution and promoting CO_2 outgassing. This effect is likely to be particularly important at confluence zones, where the mixing of water chemistries from multiple rivers can drive abrupt shifts in pH and SI_{calc}. Additionally, high flow rates and turbulence can increase gas transfer velocities and further enhance outgassing[171,172]. To effectively monitor CO_2 re-release, a thorough understanding of river chemistry and flow mechanisms, including temporal dynamics along the entire flow path, is crucial[173]. However, comprehensive data is lacking for much of the world, with only some exceptions (for example, continental USA[165]),.

Although calcite precipitation leads to CO_2 re-release, newly formed river carbonates might become thermodynamically unstable and re-dissolve as river chemistry changes under dynamic conditions. Such re-dissolution would re-sequester CO_2 and drive CDR back toward its initial potential. However, the timescales of precipitation and re-dissolution, and how these processes would be impacted by river flow dynamics, water residence times, and chemistries, are uncertain and rarely considered[26,49,170]. Such uncertainty largely arises because CO_2 outgassing prompted by ERW-derived changes in river chemistry will be highly variable and catchment specific. Importantly, pH and CO_2 partial pressure can both vary over time owing to changes in photosynthesis and respiration in response to temperature, and spatially owing to tributary and groundwater inputs[174].

Finally, the ocean-air CO_2 equilibrium, which ultimately defines the efficiency of CO_2 removal, varies spatiotemporally[175] and renders quantification of stored CO_2 challenging and dynamic. The full transport process — from site of application to the ocean — cannot be accurately captured by any monitoring or verification at the plot, stream, or river scale. The magnitude and temporal dynamics of this CDR reduction thus remain unclear[176]. More generally, the residence times of ERW-derived HCO_3^- and base cations during their transit from terrestrial ecosystems to the oceans remain highly unconstrained.

Socio-economic considerations

The Sixth Assessment report of the IPCC provided an assessment of enablers and barriers for ERW deployment[1]. In summary, environmental-ecological, socio-cultural, economic and institutional knowledge on enablers and barriers is limited or absent. Potential large-scale ERW deployments must address such uncertainties. Although existing life-cycle assessments often consider ERW-related emissions and net costs of CDR for a given region[8,35,45,70], socio-cultural and institutional assessments are less well developed, as discussed herein[83,177].

Public and political acceptance

Public acceptance of ERW is most likely when societal stakeholder expectations on its outcomes converge (for example, climate mitigation outcomes, agronomic benefits and income opportunities for farmers), promoting necessary and compatible relationships for implementations [178,179]. However, involved stakeholders, such as farmers, companies and administration, might have opposing priorities regarding the implementation and outcome of ERW. For example, the agronomic benefits of ERW are highly dependent on the fate of cations as nutrient sources and on long-lasting liming effects[15]. Agricultural stakeholders would likely prefer that cations are taken up by crops to achieve fertilisation and improve crop production. Therefore, the goals and targets of ERW as an agronomic co-benefit tool might not be in full alignment with those for CDR, for which the effective transit of base cations and HCO_3^- along the land-ocean continuum is the ultimate desire (**Box 1**).

Further, the variable content of toxic elements in potential ERW feedstock (**Fig. 1**) illustrates that careful feedstock selection will be required when considering health, food and environmental safety[27]; however, it is unclear whether risks to health and wellbeing imposed by ERW would become the responsibility of project managers, scientists or governments to manage[28]. There might be an underestimation of how important food safety is to societal stakeholders, who are likely to require a high burden of proof that there is no toxic element accumulation in crops and agricultural products[180]. There might also be an overestimation of the capacity of local institutions to meet the demands for long-term monitoring of toxic element accumulation and bioavailability over time[28]. Finally, the financial incentive for carbon market actors using ERW as a CDR technology is to maximise climate mitigation outcomes rather than maximising agronomic co-benefits or minimise the risk for toxic element accumulation[28].

The required additional mining for the expansion of ERW operations will not necessarily lead to employment and growth opportunities and might lead to labour displacement owing to decreased demand for mineral fertilizers and limestone[45]. At the same time, previously noted risks of mining expansion on local communities such as decreased local air quality, habitat destruction and decreased water quality would still occur depending on the local regulations regarding environment and employment[1,83].

Distribution of risks

Farmers will take on a disproportionate amount of risk from ERW strategies compared to other stakeholders. Society expects them to provide sufficient, safe and healthy food, while avoiding environmental pollution, applying climate-smart management and contributing to biodiversity outcomes[181,182]. The uncertainty regarding long-term effects of ERW comes with substantial hidden risks to meeting these societal expectations.

The National Fertilizer Plan of Brazil offers an example to show that political acceptance of ERW in a given region is potentially more aligned to agronomic outcomes than climate mitigation outcomes. The Brazilian government encourages and supports the use of powdered silicate rocks as a way to reduce reliance on imported fertilizers and provide affordable plant nutrients to farmers of smallholdings [183]. These policies were made possible through efforts to build local knowledge on silicate rock application and to regulate governance and coordination of silicate rock materials as fertilizers; contaminant levels in feedstock material was deemed a high priority for the farmers [183–185]. In addition, the farmers and government considered CDR by ERW as a co-benefit [183,184].

Economic viability and realisation

The economic viability of ERW for CDR depends on which stakeholder assumes the cost of procurement, transportation, grinding, and field application[186]. Four primary financing models might emerge for ERW[28,45,186]. The first involves individual farmers purchasing silicate rock material, similar to buying liming agents, fertilizers or soil amendment products. The second involves market-based systems, where silicate rock powder is applied through project implementation within the voluntary carbon market. The third is agro-food value chains, where stakeholders such as retailers and food processors invest in silicate rock powder as a way to improve the sustainability of their supplying farms, and potentially claim climate mitigation outcomes in the context of corporate sustainability targets. Finally, governments and third parties such as financial and philanthropic institutions might initiate incentive programs promoting the adoption of CDR by ERW.

Summary and future perspectives

ERW faces important challenges before it can be implemented as a large-scale CDR strategy. These challenges underscore the need for more comprehensive, holistic research and improved methodologies to accurately assess the efficacy, environmental, and socio-economic consequences of ERW (**Fig. 3**). This section outlines specific recommendations.

Regarding the availability of ERW feedstock, the priority of future research should be quantifying available feedstock by explicitly considering regional and local distributions. Life-cycle analyses are needed to define site-specific CDR_{max} benchmarks that consider feedstock composition, its distribution, and restrictions around using mining by-products owing to potential conflicting demands with other resource chains. The use of generalized CDR_{max} to estimate CDR potentials (for example, the commonly used value of 0.3 t CO_2 t^{-1} rock) should be avoided to prevent overestimations and should be assessed in conjunction with realistic dissolution rates for a given site. Rocks with lower CDR_{max} might also be suitable for ERW, but will require more locally focused life-cycle assessments[70,187], and substantially larger feedstock applications to reach net CDR. Site specific decisions are required to determine the acceptable toxic element contents of ERW feedstocks. Experiments that assess and monitor trace element dynamics to quantify accumulation and bio-availability after multiple applications need to be established under diverse pedoclimatic conditions to provide local advice prior to upscaling of ERW.

At the plant-soil system scale, clearly identifying site-specific and soil-specific restrictions for ERW should be prioritised prior to large-scale deployments. The impacts of ERW ultimately depend on soil pH, soil type, rock powder grain size, and application types and methods. Despite short-term findings (weeks to years), long-term and multifactorial scientific evidence regarding the impact of ERW on soil physical properties is sparse. Long-term, multi-year studies to fully assess and monitor ERW safety and efficacy are required. Such research will help to identify soils that offer optimum conditions for ERW and restrict the total area deemed available for up-scaling. Additionally, the complex biogeochemical interactions of applied ERW feedstock with soil carbon and nutrient cycles will require targeted experiments under realistic and specific field conditions to explore the balance of SOC stabilisation against destabilisation. Finally, soil-specific and site-specific conditions can impact the type of acid that dissolves silicate minerals (carbonic, nitric, sulphuric, or organic acids), leading to uncertainties on in-situ CO_2 uptake. The fate of weathering products in the agricultural plant-soil system should therefore be specified for site and management practices.

Along the land-ocean continuum, a better understanding of alkalinity shifts in surface waters is needed, particularly focusing on local and catchment-scale assessments. The impact of increased alkalinity on aquatic ecosystems should be thoroughly evaluated prior to large-scale ERW deployment. Additionally, understanding the spatiotemporal down-stream dynamics of this alkalinity, such as carbonate precipitation and CO_2 re-release during transit, should be prioritised. More catchment-scale investigations and empirical and system modelling efforts are thus required to constrain SI_{calc} and determine the overall efficiency of ERW. Ultimately, definitions and adoptions of a net CDR that accounts for all possible sources of CO_2 re-release are needed.

The socio-economic effects of ERW require more research to explore and define conflicting stakeholder interests. The co-benefits of ERW for soil fertility and food security might contradict the motivation to maximise CDR. Individual farmers are likely to prefer to optimise agronomic benefits, whereas market-based systems suitable for ERW are incentivised to maximise climate mitigation outcomes. ERW clearly holds the potential to strengthen agro-food value chains if agronomic benefits and climate mitigation outcomes are balanced. However, it has been argued that carbon-market-based initiatives have not consistently delivered important co-benefits to local communities[188]. Governments and third parties have the most flexibility in shaping their desired outcomes. Governments and third parties supported by socio-ecological, transdisciplinary ERW deployment models should provide policy guidelines to shape the desired outcomes of ERW applications and to address local and national implementation and costs[189].

Finally, stringent measuring, reporting and verification standards will be needed to ensure that ERW CO_2 certificates are reliable. Given uncertainties in weathering rates and the fate of DIC during transport to the ocean, CO_2 certificate reliability will remain a key issue for actors choosing to invest in ERW. Although notable progress has been made to account for downstream losses[25], more effort is needed to reduce uncertainties and implement system models to quantify source-to-sink CDR rates. Taken together, the net CDR effect of ERW cannot yet be reliably determined owing to large and unconstrained uncertainties along the entire land-ocean continuum.

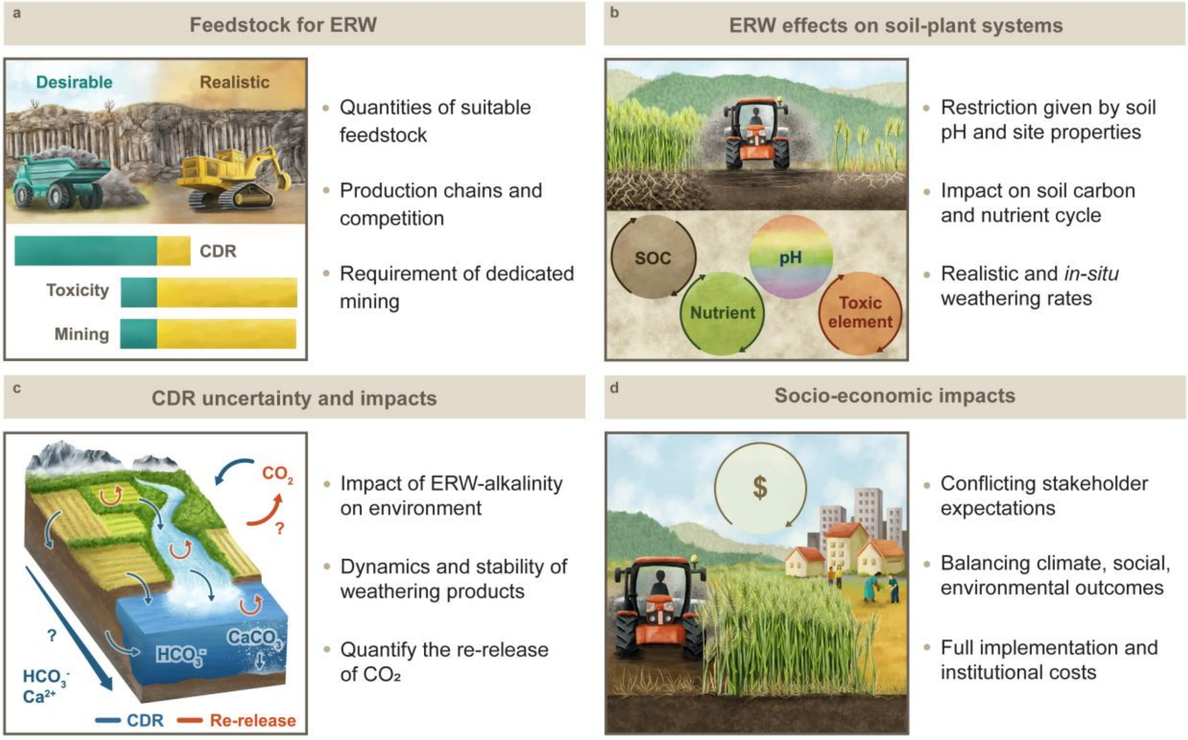


Figure 3 | Constraints on ERW as a climate-change mitigation measure. The potential of enhanced rock weathering (ERW) for carbon dioxide removal (CDR) is constrained by the global availability and suitability of feedstock material; on-site impacts on soils and in-situ effects; downstream alkalinity dynamics and CO_2 re-release along the land-ocean continuum; and socio-economic considerations.

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Author contribution statement

M.S., K.J.H, X.D., B.M. J.D.H., and E.F. wrote the manuscript. M.S. and B.M. analysed the GEOROC dataset. K.J.H, J.D.H. and M.S. analysed the GLORICH database. M.S. coordinated, revised and finalised the manuscript with contribution of A.D., R.G.H., S.D. and T.S.. The final version of the manuscript was approved by all authors.

Competing interests

B.M. is an employee of and holds share options in Flux Carbon, a for-profit company developing ERW projects in Africa. There are no elements of this manuscript that have been affected by this employment of B.M.. All other authors declare no competing interest or involvement in the used datasets or mentioned organizations.