Support to the development of methodologies for the certification of industrial carbon removals with permanent storage

Review of carbon removals through enhanced rock weathering

Prepared by Cato Sandford, Zara Popstoyanova, Chris Malins, Martin Johnson and Jonathan Lonsdale under service request CLIMATE-30420
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Submitted to:

Fabien Ramos

European Commission

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Review of carbon removals through enhanced rock weathering

A report submitted by ICF S.A., Cerulogy, Ecodiversity

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Executive summary

The goal of enhanced rock weathering (ERW) is to sequester atmospheric carbon dioxide (CO_2) as dissolved inorganic carbon (DIC). This is achieved by accelerating the dissolution of alkaline minerals on the Earth's surface, a process which shifts the equilibrium balance of DIC, and thereby creates the conditions for additional gaseous CO_2 to be dissolved from the atmosphere. This process has the potential to durably bind CO_2 over long time-scales: surface water containing dissolved inorganic carbon (DIC) eventually reaches the ocean, where DIC compounds are stable for many thousands of years.

This review seeks to elucidate the theoretical underpinnings of ERW, outline the real-world factors that affect its carbon removal efficacy, and highlight potential challenges for reliably quantifying the CO_2 durably stored by a given project. The work is delivered in the context of the EU's Carbon Removals and Carbon Farming Regulation (CRCF), under which a methodology to allow ERW to generate carbon removal units may be adopted. A handful of certification bodies (Puro.earth, Isometric, Rainbow, Carbon Standards International) have already published methodologies for crediting ERW in the voluntary carbon market; this existing work is reviewed to shed light on various quantification, baselining, and sustainability issues.

The main feedstocks identified as suitable for ERW are silicate minerals, which are commonly found in igneous rocks such as basalt – though other candidate rocks and anthropogenic industrial wastes may also be considered. Carbonate minerals may also be suitable for ERW in certain contexts. When spread onto land (primarily agricultural land) exposure to water and CO_2 initiates the geochemical reactions that may ultimately deliver carbon sequestration. A strong overlap exists between ERW and the current agricultural practice of liming (for controlling soil acidity); ERW projects using silicate rocks can take advantage of existing machinery and expertise, while achieving many of the desired effects of liming.

However, multifarious factors influence the rate and efficiency of CO_2 uptake from a given ERW project plan; these include climate and hydrology, soil conditions, and biotic factors (e.g. crops and soil fungi). It is not possible to directly measure the mass of CO_2 that has been sequestered by ERW, and so proxy measurements of weathering products in soil solids and porewater must be performed and interpreted, along with watershed modelling to understand the longevity of the geochemical carbon sequestration. Beyond this monitoring, reporting, and verification (MRV) challenge, further practical and sustainability issues remain to be addressed at both the general and the site-specific level – for example controlling the addition of toxic contaminants in ERW feedstock and avoid adverse impacts on soil health. Hence, while ERW has been discussed as a potential carbon removal mechanism since the 1990s, the voluntary carbon removals market has not yet fostered a commercial-scale ERW industry. As such, ERW researchers and practitioners have yet to settle on a uniform language for discussing the technology or a mature framework for dealing with uncertainties, or on a consensus on the risks of this technology.

Underpinning the CRCF are the 'QU.A.L.ITY' criteria – quantification, additionality, longevity, and sustainability. These frame the development of certification systems for each kind of carbon removal activity. This technical paper concludes by identifying key main issues for the European Commission to consider under these headings, including protocols for in-field measurements; accounting for secondary climate impacts; establishment of counterfactual baselines and modelling of carbon loss (to ensure that prospective projects genuinely deliver carbon savings on the long term); and requirements for controlling soil contamination through over-application of ERW feedstock.

1 Introduction and context

The European Union (EU) has adopted a Carbon Removals and Carbon Farming Regulation (CRCF) (European Union, 2024). The Regulation aims to boost innovative carbon dioxide removal (CDR) approaches and sustainable carbon farming solutions, and contribute to the EU's climate, environmental and zero-pollution goals. It is intended to improve the EU's capacity to quantify and verify carbon removals, with transparency to ensure trust from stakeholders. The European Commission, supported by experts, is developing tailored certification methodologies for carbon removal activities.

The Regulation sets out rules for the independent verification of carbon removals, as well as rules to recognise certification schemes that can be used to demonstrate compliance with the EU framework. To ensure the quality and comparability of carbon removals, the Regulation establishes four QU.A.L.ITY criteria:

1. **Quantification:** Carbon removal activities need to deliver unambiguous benefits for the climate and be measured, monitored, and reported accurately.
2. **Additionality:** Carbon removal activities need to go beyond existing practices and what is required by law.
3. **Long-term storage:** Certificates are linked to the duration of carbon storage and should ensure long-term storage.
4. **Sustainability:** Carbon removal activities must contribute to sustainability objectives such as climate change adaptation, circular economy, water and marine resources, and biodiversity.

One carbon removal approach that has been identified as having potential to meet these criteria is enhanced rock weathering (ERW).

1.1 Enhanced rock weathering

Rock weathering is a natural feature of the geochemical cycle, whereby rocks slowly dissolve in the presence of water. The dissolution of certain alkaline minerals in water has the effect of drawing CO_2 out of the atmosphere: this process is estimated to sequester approximately 3% of anthropogenic CO_2 emissions annually (Ciais et al., 2013; Taylor et al., 2016). Enhanced rock weathering (ERW) promotes this kind of reaction, using crushed and pulverised rock to create the conditions for mineral dissolution, and hence the conditions for an accelerated rate of CO_2 draw-down compared to the baseline. At the global level, ERW has been identified as having significant potential for achieving geochemical carbon sequestration[1] (Fuhrman et al., 2023; IPCC, 2018, Figure 4.2)[2].

ERW projects may be undertaken on agricultural land. Fields are accessible and farm machinery can be re-purposed for spreading the ERW feedstock[3], and the crushed rock can potentially deliver agricultural benefits in addition to carbon removal. The continuous use of nitrogen fertilisers, among other factors, tends to acidify soils through the formation of nitric acid; this reduces the soil's productivity. In response, farmers may routinely 'lime' their fields:

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[1] Geochemical sequestration is not to be confused with geological sequestration, which is where carbon dioxide is captured and injected underground into stable rock formations.

[2] It is outside the scope of this technical report to review the estimates on global or local carbon removal potential: these are highly dependent on feedstock availability, technology deployment scenarios, and the economics for future carbon removals. Similarly, we do not review the slate of current research or commercial deployments. This report focusses on the technology background and potential issues associated with certification.

[3] Note that the term 'ERW feedstock' is interchangeable with 'weathering material'.

spreading powdered limestone or related materials in order to raise the pH[4]. As such, a form of ERW is already commonly practiced, though the ERW community has often identified silicate rocks as the preferred feedstock, rather than the carbonates which are used in liming[5].

Chemically, the weathering of rocks containing alkaline minerals promotes the conversion of CO_2 from the atmosphere into dissolved inorganic carbon (DIC) compounds, namely bicarbonates and carbonate ions. As water becomes more alkaline, its capacity for holding DIC grows, and more gaseous CO_2 will dissolve to maintain chemical equilibrium. Both bicarbonates and carbonates can be stored in solution in bodies of water; providing the pH of the water body is relatively constant – true for the ocean even in light of acidification – DIC can remain stable on millennial timescales with low risk of reversal (Renforth & Henderson, 2017; S. Zhang et al., 2022).

It is worth noting that the practices and underlying chemistry of ERW share commonalities with ocean alkalinity enhancement (OAE). Whereas ERW involves the spreading of alkaline solid material on land leading eventually to greater DIC storage in the ocean, OAE encompasses pathways for increasing the alkalinity of the sea directly. This carbon removal pathway is considered in a separate technical review (Johnson et al., 2025).

1.2 Glossary of technical terminology

This section introduces some scientific concepts and terminology that are relevant for understanding carbon removal through ERW, as well as through OAE (cf. the review paper on OAE).

1.2.1 Ions

Ions form when atoms or molecules lose or gain electrons and therefore carry a net charge, or when neutral molecules dissociate into positively and negatively charged components. For example, calcium carbonate ($CaCO_3$) is an ionic compound composed of calcium ions (Ca^{2+}) and carbonate ions (CO_3^{2-}). Positively-charged ions are called cations (e.g. calcium, Ca^{2+}), negatively-charged ions are called anions (e.g. bicarbonate, HCO_3^-). An ion's 'valency' refers to its net charge: for example calcium forms divalent cations as indicated by the '2+', while bicarbonate forms monovalent anions as indicated by the '-'. Natural aqueous systems (e.g. soil moisture, seawater) are always electrically neutral at the macroscopic scale i.e. charge balance is always maintained (charges of cations and anions balance).

1.2.2 Thermodynamic equilibrium

Thermodynamic equilibrium exists between different states or forms of matter i.e. for any chemical reaction. In a system at thermodynamic equilibrium, there are no net flows of mass or energy. Hence, in a chemical system at equilibrium, the reactants and products have no tendency to change with time. For a chemical reaction, the concentrations (or activities – see below) at equilibrium can be determined from the equilibrium constant, K.

For the hypothetical reaction where a moles of A plus b moles of B are in equilibrium with c moles of C and d moles of D:

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[4] Application rates are in the low tonnes per hectare every 2-3 years (West & McBride, 2005). Soils with high clay and organic matter can require considerably more amendment compared with sandy soils with lower organic matter.

[5] An advantage of liming with limestone rather than silicates is that, all else being equal, the dissolution time is orders of magnitude faster (Hartmann et al., 2013, Table 1). We return to the use of carbonate feedstocks in Section 2.2.3.

$$aA + bB \rightleftharpoons cC + dD \qquad (1)$$

The ‘⇌’ symbol means that chemical reactions may proceed leftwards and rightwards until the rates going in each direction are equal, and equilibrium is achieved. We often imagine reactions proceeding from left to right, and refer to the species on the left-hand side as ‘reactants’ and those on the right-hand side as ‘products’; however, this is just a convention. The equilibrium state – i.e. the final concentrations of chemical species A, B, C, and D – depends on the reaction conditions (e.g. temperature and pressure) and can be determined empirically and/or can be calculated for any reaction from thermodynamic theory and data.

The equilibrium constant for the reaction is:

$$K = \frac{[C]^c . [D]^d}{[A]^a . [B]^b} \qquad (2)$$

where square brackets denote concentration, and by convention the reactants on the left/right side of Equation (1) are put in the denominator/numerator[6]. The value of K tells us where the equilibrium state is, and hence, for a given starting point, in what direction the reaction will proceed (though it doesn’t necessarily determine how quickly, as reactions may be ‘kinetically limited’ by a high activation barrier).

In a situation where K is large, the species on the right-hand side of Equation (1) (i.e. the products) dominate in equilibrium, and vice-versa when K is small. Considering a scenario where we begin with a high concentration of reactants A and B, and zero products C and D, a high K implies that the reaction may all but use up the reactants; conversely, a small value of K implies that the reaction will progress to a potentially negligible degree. For many reactions, however, the equilibrium state is such that there is an appreciable amount of both reactants and of products that coexist.

The equilibrium state is the most thermodynamically favourable state under a given set of conditions. Therefore, any change (e.g. in temperature, pressure, concentration of reactants or product) disturbing the equilibrium of a system, will result in a change in reaction rates, to counteract the change in conditions and re-establish equilibrium. For example, if the products of a chemical reaction (C and D) are removed from the reaction mixture, the forward reaction rate will increase to make up for the change in concentration until equilibrium is reached again.

These principles are relevant to all chemical processes, from fuel combustion to biochemistry. Here we focus on those which are directly relevant to ERW.

1.2.2.1 Solubility of gases in water (including soil moisture)

Take, for example, the reaction of CO_2 gas dissolving in water:

$$CO_{2(g)} \rightleftharpoons CO_{2(aq)} \qquad (3)$$

The equilibrium constant for this type of reaction is known as the Henry’s Law constant, K_H, or the air-water partitioning coefficient. In this case:

$$K_H = [CO_{2(aq)}] / [CO_{2(g)}] \qquad (4)$$

K_H can be expressed in a range of different units, the complexity of which is beyond the scope of this report; here we consider the dimensionless form (both air and water concentrations

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[6] The equilibrium constant for the ‘reverse’ reaction cC +dD ⇌ aA + bB would simply be the inverse of K as defined in Equation (2).

expressed in moles per unit volume). In this case the value of this equilibrium constant under standard conditions is around 1.3 (Sander, 2023), indicating that CO_2 roughly equally partitions between the air and water phases.

Rearranging Equation (4), the water phase concentration that would be in equilibrium with a given gas phase concentration is expressed as:

$$[CO_{2(aq)}] = K_H*[CO_{2(g)}] \tag{5}$$

Therefore, in the case where the CO_2 concentration in water is greater than the atmospheric concentration multiplied by K_H, (after accounting for temperature, salinity, and atmospheric pressure, which can affect the local value of K_H) there will be a spontaneous flow of CO_2 from water to air. Typically, soils are a net source of CO_2 to the atmosphere because respiration of plant matter yields CO_2 in the soil. In strongly alkaline soils, where CO_2 is drawn down by carbonation reactions (Section 2.1), the soil surface may be a sink for atmospheric CO_2 ($[CO_{2(aq)}] < K_H*CO_{2(g)}$). Note that dry soils are largely unable to accept CO_2 from the atmosphere to any great extent, and that drying soils will tend to release a substantial amount of the dissolved inorganic carbon in the soil water.

1.2.2.2 Acid-base equilibria

The simplest description of an acid is a compound which can donate protons (H^+ ions), and thereby of a base as a proton acceptor (which in practice can sometimes mean a donor of hydroxide ions, OH^-, which can react with protons to make water). The strength of an acid is represented by its acid dissociation constant, K_A (another equilibrium constant). Consider the generic acid dissociation reaction:

$$HX_{(aq)} \rightleftharpoons H^+_{(aq)} + X^-_{(aq)} \tag{6}$$

The acid dissociation constant, K_A, can be expressed as:

$$K_A = [H^+][X^-] / [HX] \tag{7}$$

The stronger the acid, the greater the value of K_A. Strong acids are fully dissociated in solution, e.g. hydrochloric acid, HCl has a K_A of the order of 10^8, indicating that it is effectively fully dissociated into H^+ and Cl^-. Strong bases, like sodium hydroxide (NaOH) have similarly high base dissociation coefficients (K_B).

1.2.2.3 The dissociation of water and the pH scale

Water dissociates spontaneously according to the following reaction:

$$H_2O_{(l)} \rightleftharpoons H^+_{(aq)} + OH^-_{(aq)} \tag{8}$$

The equilibrium constant for this reaction is $K_W = 10^{-14}$. In neutral solution (i.e. water with no other acids or bases), $[OH^-] = [H^+]$. We can consider the concentration of water (in water) to be unity (equal to 1), therefore $K_W = [OH^-][H^+] = 10^{-14}$ and:

$$[H^+] = [OH^-] = \sqrt{K_W} = 10^{-7} \tag{9}$$

The pH of a solution is a measure of acidity or basicity and is defined as:

$$pH = -\log[H+] \tag{10}$$

Under neutral conditions pH = -log[10^{-7}] = 7. With increasing concentration of H^+, acidity increases and the pH decreases. Therefore, pH < 7 means [H^+] > [OH^-], and vice-versa.

Compounds which, upon dissolving, release hydrogen ions or bind to hydroxide ions will make the solution more acidic, and vice-versa for alkaline compounds. When CO_2 dissolves in water, it reacts with water molecules, forming carbonic acid (H_2CO_3), which in turn dissociates into hydrogen (H^+) cations and bicarbonate (HCO_3^-) or carbonate (CO_3^{2-}) anions. This reduces the pH of the water, making it mildly acidic.

1.2.2.4 The solubility of ionic salts and their impact on pH

The degree to which solid ionic compounds (or salts) can dissolve in water is determined by the equilibrium constant of dissolution, K_{SP}, also known as the solubility product.

For a generic salt labelled as ‘XZ’:

$$XZ_{(s)} \rightleftharpoons X^+_{(aq)} + Z^-_{(aq)} \qquad (11)$$

As with liquid water, the concentration of a solid is unity (=1), so:

$$K_{SP} = [X^+][Z^-] \qquad (12)$$

In the situation where K_{SP} is greater than $[X^+]*[Z^-]$ in a solution, it is said to be undersaturated with respect to the salt, and the dissolution of solid salt would be expected, notwithstanding kinetic limitations. Where the concentration product $[X^+]*[Z^-]$ exceeds the K_{SP} for the corresponding salt, the solution is supersaturated and precipitation of the salt would be expected.

Sodium chloride (NaCl) is the salt of a strong acid (HCl) and a strong base (NaOH). It dissolves in water as:

$$NaCl_{(s)} \rightleftharpoons Na^+_{(aq)} + Cl^-_{(aq)} \qquad (13)$$

When dissolved in water, NaCl does not affect pH, because Na^+ will not mop up any OH^- and nor will Cl^- mop up any H^+: the respective base and acid dissociation constants are large. However, calcium carbonate ($CaCO_3$), whose dissolution reaction is:

$$CaCO_{3(s)} \rightleftharpoons Ca^{2+}_{(aq)} + CO_3{}^{2-}_{(aq)}, \qquad (14)$$

is the salt of a strong base (calcium hydroxide $Ca(OH)_2$) and a weak acid (carbonic acid H_2CO_3). Therefore, whilst Ca^+ will not associate with OH^- in solution, CO_3^{2-} will accept a proton (H^+) from water to form HCO^{3-} and OH^- (at e.g. neutral or typical seawater pH), leading to an increase of the pH of the solution:

$$CO_3{}^{2-}_{(aq)} + H_2O_{(l)} \rightleftharpoons HCO_3{}^-_{(aq)} + OH^-_{(aq)} \qquad (15)$$

In an acidic solution the same reaction can be thought of as:

$$CO_3{}^{2-}_{(aq)} + H^+_{(aq)} \rightleftharpoons HCO_3{}^-_{(aq)}. \qquad (16)$$

$CaCO_3$ is therefore considered a basic, or alkaline, salt. This means that the solubility of calcium carbonate is dependent on the pH, with more acidic solutions reacting away more of the dissolved carbonate and favouring further dissolution as the term $[Ca^{2+}]*[CO_3^{2-}]$ is less at lower pH for a given quantity of $CaCO_3$ dissolved.

1.2.3 Relevant compounds

Table 1.1 lists some chemicals and minerals of relevance to the present discussion. This is intended as an indicative rather than an exhaustive catalogue. It should be noted that a given chemical compound may have multiple associated minerals – for instance calcium carbonate can form calcite, aragonite, or vaterite, depending on the formation process and environmental conditions.

Table 1.1 Relevant chemicals and minerals

Category	Chemical name	Mineral / common name	Chemical formula
Acids	Carbonic acid		H_2CO_3
	Nitric acid		HNO_3
	Silicic acid		H_4SiO_4
	Sulphuric acid		H_2SO_4
	Phosphoric acid		H_3PO_4
Alkaline earth metal cations[7]	Magnesium		Mg^{2+}
	Calcium		Ca^{2+}
Inorganic carbon anions	Carbonate		CO_3^{2-}
	Bicarbonate		HCO_3^-
Silicates	Magnesium silicate	Forsterite (olivine group)	Mg_2SiO_4
	Magnesium silicate	Enstatite (pyroxene group)	$MgSiO_3$
	Calcium silicate	Wollastonite (pyroxenoid group)	$CaSiO_3$
Carbonates	Magnesium carbonate	Magnesite	$MgCO_3$
	Calcium carbonate	Calcite	$CaCO_3$
Solid oxides	Silicon dioxide	Silica	SiO_2
	Aluminium oxide	Alumina	Al_2O_3
	Calcium oxide	Quicklime	CaO
	Calcium hydroxide	Slaked lime	$Ca(OH)_2$

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[7] Alkaline earth metals are metals from Group II of the Periodic Table, the most common being magnesium (Mg) and calcium (Ca). When dissolved, they form divalent cations. These alkaline earth metals form the compounds and minerals that are most relevant for enhanced rock weathering.

2 Long-term carbon sequestration through ERW

2.1 Dissolved inorganic carbon chemistry

ERW involves spreading certain types of powdered rocks onto the land – and in particular onto agricultural land[8]. Exposure to water (from precipitation and/or irrigation) allows alkaline minerals in the powdered rock to dissolve, raising the water's pH and allowing it to absorb CO_2 from the atmosphere. Where this process can be achieved at scale, and where the absorption of CO_2 can be treated as durable (i.e. there is no net release to the atmosphere), ERW has potential as a carbon removal technology.

Chemically speaking, the goal of ERW is to augment the capacity of water – in soils, streams, lakes, and seas – to hold atmospheric carbon in the form of dissolved inorganic carbon (DIC). Gaseous CO_2 dissolves in water to form carbonic acid (H_2CO_3). The carbonic acid may dissociate to form hydrogen cations along with bicarbonate (HCO_3^-) and/or carbonate anions (CO_3^{2-}), lowering the pH[9] of the solution. This process is characterised in Equation (17).

$$\underbrace{\mathrm{CO_2}}_{\text{atmospheric}} + \mathrm{H_2O} \rightleftharpoons \underbrace{\mathrm{H_2CO_3}}_{\text{carbonic acid}} \rightleftharpoons \mathrm{H^+} + \underbrace{\mathrm{HCO_3^-}}_{\text{bicarbonate}} \rightleftharpoons 2\,\mathrm{H^+} + \underbrace{\mathrm{CO_3^{2-}}}_{\text{carbonate}} \qquad (17)$$

The equilibrium state of these reactions and therefore the relative concentrations of the corresponding DIC species is determined by environmental conditions like pH, salinity, pressure, and whether the system is exposed to the reservoir of atmospheric CO_2. For the present purposes our main focus will be pH. As illustrated in Figure 2.1 for sea water, increasing pH in water (which in its pure form has a pH of 7) drives a shift in the dominant form of the DIC, first from dissolved CO_2 to bicarbonate (at a pH between 5 and 7), and then from bicarbonate to carbonate (at a pH between 9 and 11). Significant quantities of carbonate can therefore only form in relatively high pH conditions; for instance, the distribution of the three species H_2CO_3, HCO_3^-, and CO_3^{2-} at the sea surface is about 1:90:9, meaning that a significant majority of oceanic DIC is in the form of bicarbonate (Hartmann et al., 2013). Looking at this another way, when the ocean is in equilibrium with the atmosphere its capacity to store carbon as dissolved CO_2 is relatively limited[10]; but the ability for dissolved CO_2 to react with water and form bicarbonate and carbonate species means that in fact the ocean stores 98% of the carbon in the atmosphere-ocean system (Zeebe & Wolf-Gladrow, 2001). Without a natural alkalinity source (i.e. natural rock weathering) the ability of the ocean to store atmospheric CO_2 would be considerably limited, as the pH decrease per unit of CO_2 dissolved would be greater.

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[8] While we will focus on agriculture-based projects in this review, Section 2.4 covers alternative contexts for ERW, where the underlying chemistry is similar but implementation and monitoring requirements may differ.

[9] Note that only a small fraction of dissolved CO_2 forms carbonic acid, , as the 'hydration equilibrium constant' is of the order 0.1%.

[10] From Henry's Law, the equilibrium concentration of aqueous CO_2 assuming no hydration and dissociation predicts approximately 10 µmol/l for water in contact with today's atmosphere, compared to the total dissolved inorganic carbon pool (CO_2 + HCO_3^- + CO_3^{2-}) being on the order of 2000 µmol/l.

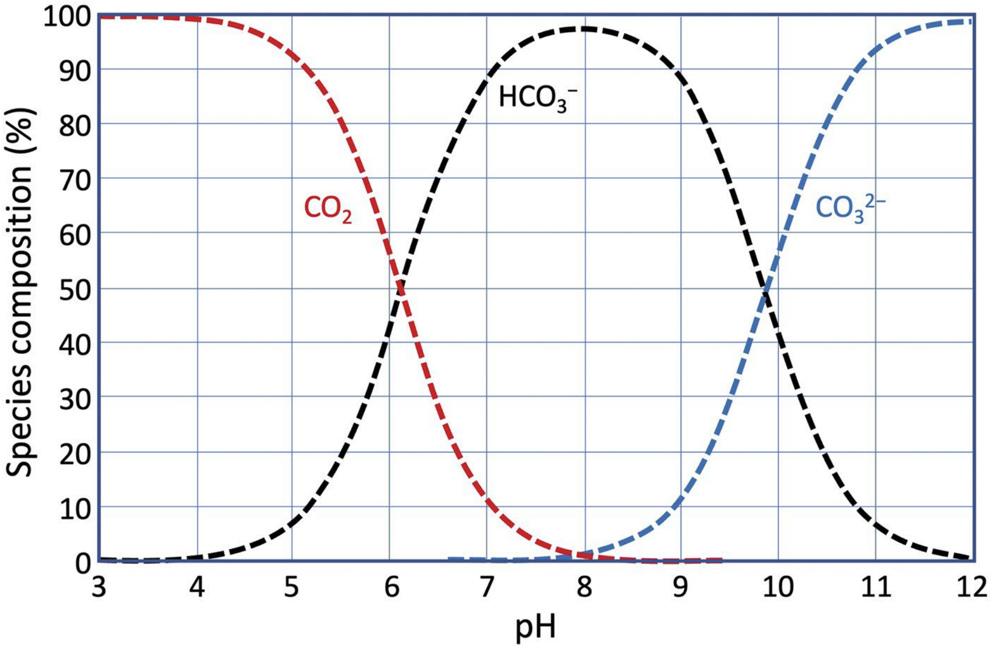

Figure 2.1 Bjerrum plot of carbonate species in sea water relative to pH

Note: the 'CO_2' line in this plot should be understood as the sum of dissolved CO_2 and carbonic acid.
Source: Rohling (2023)

When alkaline rock-derived minerals dissolve, they release metal cations (e.g. magnesium and calcium) and anions (e.g. silicate or carbonate) into solution. The anions act as an acidity sink, reacting and thereby neutralising available hydrogen ions, raising the pH, which in turn allows for more carbonic acid to dissociate to produce bicarbonate/carbonate ions. The result is a relative increase in bicarbonate/carbonate concentration and reduction in dissolved CO_2 concentration: this can be thought of as a shift towards the right of Equation (17). As the concentration of dissolved CO_2 diminishes, more atmospheric CO_2 can be dissolved in water, according to Henry's law. This process of absorbing additional atmospheric CO_2 acts to counter the increase in water's pH; it will reach a new equilibrium with a greater carbon content than before the alkalinity was increased.

Under the right conditions, bicarbonate and carbonate ions can durably sequester carbon on time-scales of centuries to millennia (Renforth & Henderson, 2017; S. Zhang et al., 2022). Such conditions exist in the ocean, which has a pH of around 8, as well as a considerable 'buffering' capacity to resist changes in pH. Under more acidic conditions (e.g. when fertiliser is added to a field and produces nitric acid) DIC may revert into dissolved CO_2, and then to atmospheric CO_2 by outgassing.

ERW feedstock dissolving into solution releases cation-anion pairs: for instance magnesium silicate hydrolyses to Mg^{2+} and SiO_4^{2-}. The (alkaline) silicate anion neutralises hydrogen ions; as described above. This raises the pH and shifts the balance of dissolved inorganic carbon species towards bicarbonate / carbonate (the right of Figure 2.1), causing a deficit of dissolved CO_2 with respect to the atmospheric concentration, and hence effects a flux of atmospheric CO_2 across the air-water interface.

Both carbonate and bicarbonate contain a single carbon atom. Bicarbonate has a valency of -1 while carbonate has a valency of -2 (Section 1.2.1): this means that a positively-charged divalent cation (e.g. Mg^{2+}) released from ERW feedstock into solution can balance the charge of two bicarbonates or one carbonate. Since the ratio of bicarbonate to carbonate that will be formed following alkalinity addition depends on the pH of the solution in question (cf. Figure 2.1), so too does the carbon stored in that solution. The final carbon reservoir for ERW is the sea, the carbon storage depends on the sea's pH: each divalent cation, which we use as a proxy for two units of added alkalinity, enables the conversion of 1.4-1.7 CO_2 molecules from the atmosphere into DIC (Renforth & Henderson, 2017). Another way of thinking about this is more explicitly in terms of the role of the carbonate system in buffering against pH changes. When a basic material is dissolved in ocean water, this base can either be neutralised by a free hydrogen ion (reducing the acidity in the system and raising its pH, and enabling atmospheric CO_2 draw-down), or it can be neutralised by a hydrogen ion donated by a bicarbonate ion becoming a carbonate ion (in which case there is no change in the pH of the system and no CO_2 draw-down. In reality, the precise origin of any particular hydrogen ion is immaterial, as once chemical equilibrium is re-established, thermodynamics fixes the proportions of acid neutralisation versus buffering for a given solution.

In principle, one can measure the DIC concentration in water samples from treated and untreated areas to estimate the additional dissolved carbon. However, the measurements can be hard to accomplish in practice, as well as being challenging to interpret (see Section 2.6.2 below). A common approach to circumvent this in the field and in the literature is to treat the metal cations as persistent 'conserved' proxies for the amount of alkalinity that has been added to the system (following the 'explicit conservative' expression for total alkalinity proposed by Wolf-Gladrow et al. (2007)); the focus is therefore on measuring the movement and fate of these cations, even though these do not directly participate in the geochemistry that sequesters carbon as DIC. The advantage is that cation concentrations can be measured more reliably, avoiding to some extent the complexities of DIC chemistry, where carbon shifts from one form into another.

Precipitation of solid carbonates following the reaction of a carbonate anion with a metal cation (an instance of secondary mineral formation) is another important phenomenon. The chemistry is covered in more detail in Johnson et al. (2025); for our purposes, the key observation is that precipitation creates a stable store of carbon in mineral form, but removes alkalinity from the solution and hence prompts the conversion of some DIC back into gaseous CO_2. This would constitute a net loss of carbon sequestration. It has been modelled as potentially significant during transport of DIC in rivers (Knapp & Tipper, 2022) and during residence in the upper layers of the ocean (Kantzas et al., 2022).

2.2 Minerals for ERW

2.2.1 Naturally-occurring minerals as feedstock

To be used at scale, ERW feedstocks need to be rich in soluble alkaline minerals, abundant, found close to potential project sites (i.e. agricultural areas), and relatively free from toxic contaminants; and crucially, weathering reactions should be energetically favourable. These requirements narrow down the list of candidate materials. In this paper, we shall focus on the main naturally-occurring minerals that have been identified as suitable: silicates of magnesium and calcium (cf. Bullock et al., 2022; Cipolla et al., 2021a; Seifritz, 1990), and to a lesser extent carbonates of the same (cf. Table 1.1)[11]. This focus notwithstanding, other minerals which are perhaps less widespread but locally abundant in certain regions may also be relevant[12]; it would be useful for further research to be undertaken to develop EU maps of mineral deposits

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[11] Alkali metal compounds (lithium, sodium, potassium) may be insufficiently soluble to undergo weathering, or may be too soluble to be found as relatively pure rock form (though two exceptions are noted in Footnote 12). Iron will tend to form oxides like haematite (Fe_2O_3) and precipitate out of solution. Compounds containing other metals like copper, nickel, and aluminium may poison soils if added in large quantities (Dupla et al., 2023).

[12] For example, leucite (a potassium-aluminium-silicate) and nepheline (a sodium-potassium-aluminium-silicate) are minerals that have similar dissolution rates to olivine and are abundant in certain settings.

and quarrying activity to come to a synthetic view of where the potential ERW feedstocks are located.

Magnesium and calcium silicates are found in especially high concentrations in igneous ‘mafic’ and ‘ultramafic’ rocks like basalt, gabbro, and peridotite. Within these rocks are many types of magnesium-containing mineral compounds characterised by their chemical compositions and structures. A simple and common example is magnesium orthosilicate (Mg_2SiO_4), also known as ‘forsterite’, from the olivine group of minerals[13]. When exposed to water and dissolved CO_2, forsterite will partially dissolve to produce magnesium cations, bicarbonate anions, and silicic acid[14], as shown on the right side of Equation (18):

$$\mathrm{Mg_2SiO_4} + 4\,\mathrm{CO_2} + 4\,\mathrm{H_2O} \rightleftharpoons 2\,\mathrm{Mg^{2+}} + 4\,\mathrm{HCO_3^-} + \mathrm{H_4SiO_4} \tag{18}$$

Further examples can be found in (Jiang et al., 2023, Table 5). From Equation (18), each molecule of forsterite that dissolves provides four units of alkalinity to the solution and can therefore potentially pull up to four molecules of CO_2 out of the atmosphere in the form of bicarbonate ions (two per magnesium atom). This should be treated as an upper limit, as carbonate system chemistry will tend to buffer the formation of bicarbonate in neutral-to-alkaline soils.

Once DIC has been formed, re-release of gaseous CO_2 from the DIC pool can happen if the aqueous environment becomes more acidic (i.e. lower pH)[15], pushing the Equation (17) balance back to the left. Conversely, if the pH of were to increase past a certain level, then magnesium carbonate (magnesite, $MgCO_3$) can precipitate out of solution as a solid secondary mineral (see Section 1.2.2.4). Per molecule of magnesite, this shift would store one carbon atom in the solid material, but would at the same time release one atom as CO_2 because alkalinity is removed from the solution[16].

As mentioned in Section 2.1, it is already common practice to lime agricultural fields to reduce acidity (and/or to reduce the bioavailability of heavy metal contaminants). This can be done with crushed limestone (calcium carbonate, $CaCO_3$), or with ‘quicklime’ (calcium oxide, CaO), which is produced by heating calcium carbonate until it releases a CO_2 molecule, or ‘slaked lime’ (calcium hydroxide, $Ca(OH)_2$)[17]. Calcium carbonate is soluble in water, especially in acidic conditions, and each molecule can react with a CO_2 molecule and a water molecule to yield two bicarbonate ions, delivering net storage of one carbon atom. This overall reaction is shown in Equation (19)[18]:

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[13] The literature frequently refers to ‘olivine’ where the metal component is provided by a mix of magnesium and iron: $(Mg,Fe)_2SiO_4$. Forsterite corresponds to an olivine of pure magnesium, and fayalite to an olivine of pure iron. The iron component of olivine will contribute little to carbon sequestration and may even lead to net CO_2 release to the atmosphere (Griffioen, 2017).

[14] A hydrated form of silica, SiO_2.

[15] Provided there is no reservoir of alkalinity – like further undissolved rock – to counteract the acidity. In realistic ERW scenarios, addition of acidity would stimulate further rock dissolution which would counteract the pH reduction.

[16] In the language of the cation-oriented perspective (discussed above), it is often said that each divalent magnesium or calcium cation can balance the charge of two monovalent bicarbonates, or that of a single divalent carbonate ion. In the first instance, two carbon atoms are stored; in the second, only one.

[17] Quicklime and slaked lime are produced by heating limestone in a high-temperature kiln, driving off carbon in the form of CO_2. The energy intensity and extra production steps means they are more expensive than simple limestone; but they dissolve faster in water and are more effective at neutralising acidity per tonne of material applied. Calcium carbonate dissolves significantly faster than other minerals (Hartmann et al., 2013), and calcium oxide / hydroxide even faster.

[18] Equation (19) considers calcium carbonate which contains one carbon atom per molecule. A similar equation holds for quicklime and slaked lime made from calcium carbonate via calcination at high temperatures, though there would be an intermediate step where a molecule of CO_2 is emitted to the atmosphere before being re-bound as bicarbonate.

$$\mathrm{CaCO_3 + CO_2 + H_2O \rightleftharpoons Ca^{2+} + 2\ HCO_3^-} \tag{19}$$

In the case of quicklime, each molecule can react with two molecules of CO_2 to yield two bicarbonate ions, potentially delivering net storage of one carbon atom when the CO_2 molecule released during calcination is considered. The storage per unit of added feedstock is reduced, as was discussed in Section 2.1, if carbonate ions are formed instead of bicarbonate ions. For silicate minerals the quoted ratio was 1.4-1.7 CO_2 molecules stored per two units of alkalinity added, but because the dissolution of calcium carbonate releases a carbon atom, the net impact is less by one (i.e. 0.4-0.7 CO_2 molecules per two units of alkalinity) (Renforth & Henderson, 2017).

Liming is not typically recognised as a carbon removal pathway: the IPCC's Tier-1 emission factors assume that all the carbon contained in liming feedstock is released into the atmosphere, meaning an emission of 0.12 tonnes of carbon[19] per tonne of $CaCO_3$ (IPCC, 2006, Chapter 11.3)[20]. This stands in tension with the understanding of added alkalinity providing a net carbon sink, and the IPCC guidelines acknowledge that the assumption amplifies apparent emissions. An empirical study by Hamilton et al. (2007) of field drainage waters reported that 25-50% of the carbon in applied limestone was actually sequestered (similar to the ocean range quoted above), and point to a more nuanced carbon-balance picture depending on field conditions and land-management practices.

Other studies in the academic literature have sought to quantify the net carbon balance of liming (e.g. Buckingham & Henderson, 2024; Robertson, 2014; West & McBride, 2005), reaching a range of conclusions that underscores context-dependency of measured outcomes as well as nuances arising from what is considered in scope for the analysis[21]. In any case, for ERW projects where application of silicate rock is known to displace some amount of liming activity, accurate determination of the carbon impacts will require consideration of the baseline carbon removals achieved by the liming. This was recently discussed in a Carbon Plan workshop (Hausfather & Kukla, 2025).

2.2.2 Rock sourcing: silicates

Much of the focus of the developing ERW industry is on the use of silicates as ERW feedstock. Basalt is considered to be an archetypical ERW feedstock, being abundant and rich in magnesium/calcium silicates. It is already mined around the world, primarily for building materials (concrete aggregate, masonry, etc.). The literature offers various estimates of rock production and mining capacity (e.g. Madankan & Renforth (2023) for the UK). One attractive option for ERW is the fine quarry dust ('fines') that is a by-product of the crushing and screening process and which has a sub-millimetre grain size. It has been estimated that 250-300 kg of fines are produced alongside each tonne of principal rock product (Manning et al., 2024); this dust may be used as fine aggregate for construction[22] (Altheeb, 2022; Valentin et al., 2021), for compact road beds, and as back-fill in landscaping and mining operations (Trimax Machinery, 2023). At present, it is understood that there is a surplus of quarry dust that could be used for ERW (Beerling et al., 2020; cf. Forrest & Wentworth, 2024b). It has been estimated

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[19] One tonne of carbon equals 44/12 tonnes of CO_2.

[20] The rules on liming were not updated in the IPCC's 2019 refinement to the inventory guidelines.

[21] The principal issue of weathering by acids other than carbonic acid will be discussed in greater depth in Section 2.7.2.

[22] Potentially replacing fly ash which is becoming scarcer in regions where coal burning is on the decline.

that global production of silicate mining waste is in the hundreds of Mt/year[23]. For the UK, the figure is 86 Mt/year, which would enable an application rate of over 10 t/ha/year on all UK arable land (Royal Society & Royal Academy of Engineering, 2017). This existing supply has the advantage that additional energy consumption for crushing and powdering would be minimised. Certification would require a life-cycle emissions analysis for feedstock sourcing and in the CRCF context the use of this existing supply would therefore reduce emissions in the $GHG_{associated}$ term.

Future scale-up of ERW might require additional mining and active crushing of rock, with implications for energy consumption and ecological impacts (Section 2.9.8). Or it might divert material away from existing uses. If, for instance, growth of the ERW industry meant that disused sections of quarry had to be stabilised with new purposely transported material in lieu of the locally available quarry dust, this would not only represent an additional expense but would also undermine the net greenhouse gas savings of the ERW application due to the implied indirect emissions.

There is also a baselining question associated with the use of existing supplies of powdered rock. Existing uses may in some cases already offer CO_2 uptake potential, meaning that the net benefit of using the material for ERW could be reduced. There are some contexts in which we would expect baseline CO_2 uptake to be quite limited, for instance when incorporated into concrete, used in roadbed, or buried under earth, we would expect the infiltration of water and CO_2 to be slow and carbon storage to be minimal (cf. the *Review of carbon removals through mineralisation into products* (Sandford & Malins, 2025b)). Similarly for backfilling old mines, where quarry dust is firmly compacted or bound together with water and cement, and buried underground. We are not aware of any evidence of significant CO_2 uptake under these conditions.

Baseline CO_2 uptake could be more of an issue in the period before use or disposal where quarry dust is left in heaps on-site, exposed to moisture and the atmosphere (Sairanen et al., 2018). It is unclear to what extent these storage conditions would enable CO_2 uptake, how durable that uptake would be (e.g. rock material may undergo cycles of dissolution and re-precipitation as rain falls and evaporates), and how this might change over time (the weathering rate may be consistent, or it may experience an initial bout followed by formation of a crust or some other kinetic limitation). Fine particles of dust that are blown off into the environment could be assumed to undergo weathering without any active intervention[24], meaning that this fraction of feedstock sourced for ERW projects may have to be discounted. In the interior of a quarry dust heap, mineral dissolution could be limited (Stubbs et al., 2025) by low moisture and CO_2 concentration and build-up of reaction products like silica; further research would be needed to characterise baseline CDR for a range of quarry dust storage practices and durations.

The ERW reaction generally proceeds faster for smaller particles which have higher surface area as this allows more rapid dissolution of minerals. Sizes in the low tens of micrometres have been identified as suitable (Rinder & von Hagke, 2021; Vanderkloot & Ryan, 2023), with particles of less than one micrometre in diameter dissolving on time-scales of years (de Oliveira Garcia et al., 2020; Renforth et al., 2015). Quarry dust contains a distribution of particle sizes (Hyde et al., 2017; Stubbs et al., 2025), which will vary between batches and between quarries. Some dust may be applied to fields as-is (McDermott et al., 2019); but efficient use of the resource may require further grinding and screening to reach the desired size distribution. This

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[23] To our knowledge, there is no coordinated collection of waste stream data from the global silicate mining industry. The presentation by Hays & Ghag (2023) quotes an estimated 600 Mt/year of basalt by-product mined worldwide, but we were unable to verify this figure in the peer-reviewed literature.

[24] Due to the health hazards that inhaled rock dust pose to humans and animals (see Section 2.9), operators may employ strategies like regular wetting and shielding to control fugitive dust. But some material will inevitably be lost.

would entail additional energy consumption and greenhouse gas emissions (Eufrasio et al., 2022; Future Forest Company, 2021). Similarly, the transport distance between ERW feedstock production in quarries and ERW projects on agricultural lands is a major determinant of project emissions, and some sites which would otherwise be suitable for ERW will be located too far away from feedstock production to be viable (Jerden et al., 2024; Madankan & Renforth, 2023). In summary, each prospective ERW feedstock will have to be screened to confirm whether the chemical content and grain size is appropriate given local conditions.

2.2.3 Rock sourcing: carbonates

Liu et al. (2011) estimated that silicate rocks contribute only 6% of the Earth's natural weathering carbon sink, and that the other 94% is due to carbonates. As an ERW feedstock, carbonate minerals hold some advantages over silicates. They are already quarried at scale – global production of limestone is in the billions of tonnes per year (Caserini et al., 2022), an order of magnitude more than silicates[25]. Their agricultural use is already normalised and well-studied. They dissolve extremely quickly compared with silicates[26] (Palandri & Kharaka, 2004), meaning they act faster to neutralise acids, and the grinding stage is less energy intensive: both because larger grain sizes can still yield tolerable dissolution rates[27], and because limestone is a relatively soft rock with a lower crushing energy. Faster action also means less material is needed to achieve the same pH change on the desired time-scale, which cuts transport costs and associated emissions. Indeed, an article for the Oxford Energy Forum (Bryson, 2025) argues that a portion of the ERW carbon removal signal observed from basalt application may in fact originate from the small amounts of fast-dissolving calcite found among the slow-dissolving silicates – a better understanding of this issue could have important ramifications for the fledgling ERW sector.

A main disadvantage of using carbonates as ERW feedstock is that they have only half the maximum theoretical efficiency of carbon removal per unit of alkalinity added (as described in Section 2.2.1), and less than half the efficiency when real-world buffering is taken into account. For calcium-rich carbonates, there is also a greater chance of downstream carbonate precipitation (a carbon loss) in water bodies that are close to saturation[28].

Limestone fines have a range of existing applications, overlapping with those for silicate rocks in the previous sub-section. Baselining of CO_2 uptake in these applications must therefore follow similar lines as discussed in Section 2.2.2. Limestone fines may also find other uses, for instance as an additive in cement mixes: since this particular use displaces some fraction of energy- and emissions-intensive Portland cement, diverting the material for use as ERW feedstock could incur some indirect emissions.

Given that liming with carbonates is already common practice in agriculture, certification of carbonate addition as an ERW activity would require careful consideration of questions of additionality and of baselining. Research studies have concluded that 'under-liming' is common

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[25] Carbonate minerals are integral to cement manufacture, water treatment, and a range of heavy and light industries (Grosso et al., 2020).

[26] Precise comparison of the dissolution rates will depend on the smoothness of the grains, the pH of the environment, the ambient saturation with respect to dissolution products (e.g. calcium ions and silica), temperature, and other factors. But roughly speaking, if a grain of calcite of a given size takes on the order of days to fully dissolve under acidic conditions found in the field, a grain of forsterite of the same size could take on the order of hundreds of years.

[27] It should be noted that the EU has rules about the size distribution of grains used for limestone amendments (see the Fertilising Products Regulation (European Union, 2019)). In order to deliver a comparable dissolution rate comparable with limestone, a given silicate mineral would have to be ground many times smaller.

[28] Magnesium-rich carbonate feedstocks like dolomite may be preferable in these situations (if they are available nearby), as magnesium concentrations that approach the local saturation point are much rarer than near-saturation calcium concentrations.

on European arable and grassland, with for instance 40-60% at sub-optimal pH in the UK and Germany (Goulding, 2016; Müller et al., 2022). This suggests that there is an opportunity for ERW as a carbon removal that would also improve agricultural outcomes, though the additionality issues would need to be carefully assessed.

2.2.4 Non-natural materials as feedstock

Certain industrially-generated materials may also have potential for use as ERW feedstock. These include industrial wastes (such as iron and steel slag or fly ash from coal-fired electricity production) and concrete construction and demolition waste (McDermott et al., 2024; Renforth, 2019). Soluble alkaline earth metal compounds like calcium hydroxide and magnesium silicates contained in these materials could in principle deliver carbon removals in the same way as the silicate rocks discussed above (Grosso et al., 2020). Renforth et al. (2011) estimated global production of silicate-bearing wastes is 7-17 Gt/year, while Bullock et al. (2021) estimated 9.5-16.7 Gt/year, with a central estimate of 12.9 Gt/year.

Much of this material will have complex mineral composition and would weather slowly, and in some cases would contain toxic elements that make it inappropriate for use on agricultural lands (or indeed earn it a hazardous waste classification). Still, significant potential has been identified for a range of industrial wastes, especially if interventions are taken to accelerate weathering (Yin, 2014). The production of alkaline materials as by-products of other processes – and their associated carbon capture capacity – is projected to rise throughout the 21st century (Renforth, 2019). These industrial wastes will experience some baseline rate of natural weathering depending on storage conditions (e.g. Wilson et al., 2009), and this would have to be subtracted off to determine the net carbon sequestration potential of additional interventions. The geographical distribution of legacy deposits of slag and demolition waste is different from the distribution of basalt quarries, and the use of these materials could significantly expand the range of viable ERW projects (cf. Madankan & Renforth, 2023), while potentially creating a new disposal pathway (for true wastes) or a new revenue pathway (for by-products that have existing low-value uses).

Depending on the form of the waste materials as produced, there may be additional energy requirements (and hence carbon and economic costs) for crushing and grinding into ERW-compatible powder. Conversely, those wastes which already have a high surface area and are exposed to the atmosphere may undergo weathering in-situ. As in Section 2.2.1, this could complicate assessment of the carbon removal baseline, as some amount of CO_2 would be absorbed into DIC in the absence of an ERW-type project. Studies of slag heaps' hydrogeology tend to focus on the rate at which heavy metals and other toxic compounds get flushed into surrounding water and soil systems, rather than the formation of DIC; but this nevertheless establishes that water flow and material dissolution in these kinds of deposits are non-negligible phenomena; further investigation of natural weathering rates would be needed to come to a firmer conclusion about the carbon removal baseline.

Contaminants in industrial wastes may render them unsuitable for agricultural application (cf. Bunzl et al., 2001). Slag from non-ferrous metal production has been generally identified with more heavy metal contamination than slag from ferrous metal production (Piatak et al., 2015). The sulphur content of applied industrial wastes also warrants consideration; sulphates boost soil fertility as an essential nutrient (Saha et al., 2018; Sharma et al., 2024); at the same time, some chemical pathways for producing sulphate may counterproductively acidify the soil through the formation of sulphuric acid[29]. A more thorough understanding of how enrichment

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[29] For instance, calcium and magnesium sulphide (CaS and MgS) in blast furnace slag (Schrama et al., 2017) will chemically hydrolyse in water to form alkaline hydroxides and hydrogen sulphide. The hydrogen sulphide may then be passively oxidised or metabolised by soil microorganisms to produce sulphuric acid. In the mining industry, 'acid

with sulphur-containing compounds may affect fertility and carbon sequestration capacity is therefore needed.

It has been proposed that concrete waste should be preferred over industrial slag due to its lower contaminant content (Choi et al., 2021). However, testing on a case-by-case basis may allow the identification of appropriate ERW feedstock material from a range of sources.

2.3 Silicate rocks in agriculture

As mentioned above, conventionally field acidity is controlled by the application of fast-dissolving crushed calcium carbonate, or even faster-dissolving calcium oxide or calcium hydroxide. But silicate amendments are also currently used in agriculture for pH management, fertilisation, and pest control, irrespective of any benefits of ERW as a carbon sink. These may be called silicate rock powders (SRPs), rock dust, stone meal, agrominerals, or remineralisers, but are essentially the same as ERW feedstocks (Swoboda et al., 2022).

Silicates contain silicon, and this can be absorbed by plant roots (Caubet et al., 2020; Cornelis & Delvaux, 2016; Swoboda et al., 2022), and silicon is present in plants in quantities comparable to macronutrients such as calcium, magnesium, and phosphorous (Epstein, 1999). Though the generic importance of silicon as a plant nutrient has long been recognised, its precise role in physiological and metabolic processes for different plant taxa is still a topic of active research. It has a clear role in plants' ability to respond to stressors: both abiotic stressors like salinity, water availability, and temperature (Cooke & Leishman, 2016), and biotic stressors like herbivores and pathogens (Frew et al., 2018). There is also growing evidence of silicon's importance for fundamental metabolic and growth processes (Frew et al., 2018). Silicon fertilisation may therefore be beneficial where removal from soils by plants and water flow exceeds replenishment (Cooke & Leishman, 2016)[30].

Many studies have identified positive growth impacts of ERW soil amendments more broadly (i.e. beyond silicon availability). These have highlighted factors like the fertilising effect of micronutrients and phosphorous present in the feedstock rocks (Beerling et al., 2024; Goll et al., 2021; Kelland et al., 2020; Porder & Ramachandran, 2013; Skov et al., 2024; X. Wang et al., 2025); the stimulation of soil biotic activity (Dupla et al., 2023; Kelland et al., 2020; X. Wang et al., 2025); and pH-mediated reduction in the availability of toxic metals like aluminium[31] (Skov et al., 2024).

Clay formation from the products of weathering is an important instance of secondary mineralisation, and has been associated with improved water-holding capacity in soils (de Oliveira Garcia et al., 2020) as well as productivity-boosting improvements to soil structure, microfauna, and organic matter retention (Buss et al., 2024; Goulding, 2016). It should be noted that the binding of weathering products into clay particles represents a loss of carbon sequestration potential – this is further explored in Section 2.5.6. Moreover, where the ERW feedstock has a similar chemical profile to the underlying soil, as in Ramezanian et al. (2013), it may be impossible to identify any positive effect on yield[32].

Concerns have also been raised about potential negative effects of silicate application such as heavy metal accumulation, soil compaction, sodification (i.e. increased salinity), and priming

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mine drainage' is a known environmental issue associated with sulphide-rich mineral deposits (Akcil & Koldas, 2006).

[30] An example where this is relevant is the continuous industrial agriculture systems of the USA (Tubana et al., 2016).

[31] ERW may decrease availability of micronutrients like manganese and zinc through the same mechanism (Buss et al., 2024).

[32] The feedstock analysis in Ramezanian et al. (2013) indicates a high iron content, which, as discussed in Section 2.5.6.3 below, would counteract DIC formation in some circumstances.

of soil respiration (Dupla et al., 2024). We elaborate on various sustainability considerations in Section 2.8 and Section 2.9.

2.4 Non-agricultural ERW pathways

Though this technical review focusses on agricultural ERW deployments – reflecting scalability of agricultural application and the natural overlap with existing agricultural practices – in this section we introduce some other technologies using similar chemistry principles. We split these into contexts where alkalinity augmentation causes the dissolution and sequestration of atmospheric CO_2; and contexts where a concentrated CO_2 stream from a carbon capture facility is stored. The former has more obvious overlap with agricultural ERW projects, whereas the latter can be thought of as using DIC and/or solid carbonate minerals as an alternative to geological carbon storage.

We shall only consider terrestrial ERW deployments in this document; ocean alkalinity enhancement (OAE) using crushed minerals is addressed in a separate review paper.

2.4.1 Application to non-agricultural soils

Spreading of crushed and powdered rock has been trialled in a number of land types. Researchers have investigated how storage and distribution of the basalt dust by-product around quarries could promote carbon removal, and how this could be maximised via the management of existing dust stockpiles (Harrison et al., 2013; Stubbs et al., 2025).

Washbourne et al. (2015) and others have highlighted the carbon sequestration opportunity in urban soils following application of silicate- and hydroxide-rich demolition waste. Bell et al. (2024) found ERW carbon sequestration on grass lands to be effective, without significant adverse impact on species or soil toxicity.

Several studies have examined forest landscapes such as degraded, acid-rain impacted forests (Taylor et al., 2021), tropical rubber plantations (Bi et al., 2024; X. Wang et al., 2025), oil palm plantations (Larkin et al., 2022), and reforestation sites (Waring et al., 2025). Some studies have also noted increased organic carbon stocks in soils and/or vegetation – including faster-growing trees – arising from mineral fertilisation and the effect of silicates as discussed in Section 2.3 (Bi et al., 2024; Gaucher et al., 2025; Waring et al., 2025; Xu et al., 2024).

Several studies nevertheless urge caution against ERW application in natural or semi-natural forests, and particularly in tropical forests, due to their high biodiversity and complex biological interactions (Bi et al., 2024; Edwards et al., 2017a; Gaucher et al., 2025; X. Wang et al., 2025). In these systems, unforeseen ecological consequences of ERW processes could outweigh the carbon benefits if they disrupt forest systems.

2.4.2 Concentrated CO_2

The weathering reaction rate can be accelerated by using a concentrated CO_2 stream in some form of reactor vessel – for instance from a direct air capture facility or from point source capture. To qualify as ERW, the treatment must increase the concentration of DIC in the geochemical system (and eventually in the ocean); the formation of solid carbonates is considered in a different technical report (Sandford & Malins, 2025a).

One approach relies on multi-stage thermochemical reactors which extract and purify target compounds from a rock feedstock before exposing them to CO_2 under controlled conditions (Daval et al., 2009; Kakizawa et al., 2001; Lackner et al., 1995). This increases the rate and efficiency of the reaction. Depending on the setup the reactor may produce an aqueous stream of DIC which is released into water courses and ultimately reaches the ocean, or solid carbonates which can be stored or disposed of.

Another similar approach omits the chemical extraction step and just relies on temperature and pressure control along with CO_2 concentration to optimise the CO_2 mineralisation process. This is the setup that was imagined by Seifritz (1990) who first introduced the possibility of ERW as a carbon abatement strategy, and various experimental protocols have been explored since (Daval et al., 2009; Lackner et al., 1995; Nduagu et al., 2012; Rau et al., 2007; Xing et al., 2021). The voluntary certification body Isometric has recently released a draft methodology for certifying these kinds of projects (Isometric, 2025b). Again, the output can be either aqueous DIC or solid carbonate minerals.

2.5 Factors affecting weathering rate and CO_2 uptake

2.5.1 Overview

ERW initiates the sequestration of CO_2 when alkaline minerals dissolve into water and stimulate the formation of DIC ions from dissolved CO_2. This tips the balance of Equation (17) and causes CO_2 from the atmosphere to enter the water to re-establish equilibrium. This sequestration is durable when alkalinity (or the chemical buffering of pH) prevents the re-release of CO_2 – such conditions are present in the ocean, so the ocean can be considered a long-lived store of carbon[33].

It is straightforward to place an upper bound on the amount of CO_2 sequestered by weathering reactions; much more difficult is to determine the rate of weathering, which depends on how quickly the feedstock releases the relevant ions into solution, the rate at which CO_2 dissolves, and competition between the formation of different reaction products. Techniques for analysing and quantifying weathering on large scales continue to be developed (Wittmann et al., 2024). The potential for re-emission of CO_2 (called 'CO_2 loss') is covered in the next section, Section 2.6.

In this section we outline the main factors influencing these behaviours, noting at the outset that the literature tends to focus on candidate materials which are deemed abundant and sufficiently fast-weathering (cf. Golubev et al., 2005) to make a difference to CO_2 balances on the time-scale of decades. Baseline dissolution rates of different minerals under different conditions are discussed in Bullock et al. (2022).

2.5.2 Climate and hydrological conditions

Since the chemical reactions that underpin acidic weathering take place in solution, water is a critical ingredient. Acidic weathering will progress slowly if at all in arid conditions, and fastest in humid conditions (Hartmann et al., 2013). As with most chemical reactions, silicate weathering rates generally increase with temperature (Renforth, 2012), and so the hot and humid climate in the tropics has been identified as being optimal for ERW (Baek et al., 2023). Of course, this should not rule out other geographies; and in the case of calcium carbonate, solubility actually reduces as temperature rises (this is because the dissolution reaction is exothermic and the solid state is favoured at higher temperature).

The products of the weathering reaction remain in the soil until they are washed out by rain or by irrigation. The residence time and chemical conditions in various carbon reservoirs are discussed in more detail in Section 2.6; what concerns us for now is that a build-up of reaction products would tend to slow and inhibit further reactions. For example, saturation of silica or silicic acid would impede further weathering of silicate rock (Harrington et al., 2023; Köhler et

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[33] For the purpose of this review, we shall treat the ocean as being 'well buffered' and as providing long-term bicarbonate stability. More detail about the subtleties of this assumption can be found in the *Review of carbon removal by ocean alkalinity enhancement* (Johnson et al., 2025).

al., 2010). Continuous flushing out of reaction products refreshes chemical pools and allows for a higher rate of weathering and hence carbon draw-down (Deng et al., 2023). In field situations the details of these processes are not simple, and are bound up with hydrology-dependent activity of soil organisms (Cipolla et al., 2021a) – for instance, uptake of silica by diatoms would achieve the same flushing effect (Schuiling et al., 2011).

2.5.3 Material characteristics

Basalts contain a diversity of minerals with different levels of water solubility (Hartmann et al., 2013; Lewis et al., 2021). The same is true of alkaline industrial wastes. The rate of carbon removal will typically rise with increased feedstock solubility, but only insofar as it is alkaline minerals that are dissolving. For example, dissolving of sulphide compounds would work in the opposite direction and lower the soil water pH (Bufe et al., 2021). Since these dissolve significantly faster than silicates, in the short term even small concentrations may cause a net release of CO_2 to the atmosphere. Other components may well dissolve faster or slower, but will not directly influence carbon removal (Power et al., 2025).

As noted above, smaller feedstock grains with larger surface area tend to dissolve faster (Bullock et al., 2022; Rinder & von Hagke, 2021). Some models assume a linear relationship between surface area and dissolution rate (Lewis et al., 2021), though the domain of validity of this approximation may be limited by other reaction bottlenecks like CO_2 concentration or acidity (cf. Amann et al., 2022). Certain feedstock materials will present more inhomogeneous granular surfaces than others – for example, steel slag is identified as tending to form more jagged grains. A batch of ERW feedstock will include a distribution of particle shapes and sizes; of these, the more jagged and smaller particles will tend to weather faster, with the surface area to volume ratio of each grain changing over time. In addition to pulverising feedstock grains to the desired size distribution, which is commonly identified as an energy-intensive process, research has explored thermochemical pre-treatment techniques for accelerating weathering (Maroto-Valer et al., 2005; Olsen & Donald Rimstidt, 2008). It should be noted that pre-treatment with industrially-produced acids may reduce the total weathering potential of a given batch of feedstock.

One mathematically simple approach that is commonly used in the literature to model weathering is the ‘shrinking core model’ (cf. McDermott et al., 2024). This describes how a spherical solid particle immersed in a solvent shrinks over time, by moving the dissolution interface uniformly inward towards the unreacted core. As may be expected from the discussion about particle inhomogeneities, the model does not precisely capture the dissolution rates of real particles, and its domain of validity is not guaranteed in all settings (Didyk-Mucha et al., 2016). This has prompted various attempted modifications (Renforth et al., 2015; Rinder & von Hagke, 2021), and the proposal of experimentally-tested alternative models (e.g. Starnoni et al., 2024). Again, the suitability of any given empirical / phenomenological / theoretical model for a given dissolution scenario will be context-dependent.

The chemical heterogeneity noted above will tend to lead to a slowing of mineral dissolution over time (Oelkers et al., 2018; Power et al., 2025), meaning that early measurements can be misleading if simply extrapolated into the future. Larger, heavier particles are more likely to remain in the project vicinity – lighter particles that are moved by wind before much weathering activity has taken place will fail to raise the pH of the target soil and bring less agricultural benefit (Section 2.9), and will of course not show up in in-situ soil measurements used for crediting carbon removals.

2.5.4 Soil water acidity

Acidity in the soil is a major determinant of ERW feedstock dissolution and the extent to which DIC provides a carbon sink. Acidity has a number of origins (Bufe et al., 2021), but in the ERW context there is a convention of distinguishing carbonic and non-carbonic acids and treating them differently as we shall see. Carbonic acid in soil water originates from the dissolution of atmospheric CO_2 into rain (which naturally has a pH around 5.7), and from the dissolution of atmospheric and biogenic CO_2 directly in the soil pores. Non-carbonic acids include nitric acid (HNO_3) from fertiliser, sulphuric acid (H_2SO_4) from industrial emissions, and organic acids from plants.

Carbonic acid (H_2CO_3) is a ‘diprotic’ acid because it can produce two hydrogen ions (i.e. two moles of acidity); but it is also a ‘weak’ acid because its acid dissociation constant is small (cf. Section 1.2.2.2). For the first ionisation:

$$\mathrm{H_2CO_3 \rightleftharpoons H^+ + HCO_3^-} \quad \Bigg| \quad \mathrm{K_a^{(1)}} = \frac{\mathrm{[H^+][HCO_3^-]}}{\mathrm{[H_2CO_3]}} = 4.3 \times 10^{-7} \tag{20}$$

And for the second ionisation:

$$\mathrm{HCO_3^- \rightleftharpoons H^+ + CO_3^{2-}} \quad \Bigg| \quad \mathrm{K_a^{(2)}} = \frac{\mathrm{[H^+][CO_3^{2-}]}}{\mathrm{[HCO_3^-]}} = 4.7 \times 10^{-11} \tag{21}$$

The small values of K_a mean that the acid tends not to ionise under standard conditions (and moreover, the concentration of bicarbonate ions is four orders of magnitude higher than that of carbonate ions).

Some non-carbonic acids like nitric and sulphuric are considered ‘strong’ acids because they have significantly higher dissociation constants[34]. When these are present in the soil, they will supply hydrogen ions that weather the ERW feedstock. For example, under the right conditions ammonium nitrate fertiliser will nitrify into nitric acid and react with forsterite to yield the overall reaction in Equation (22)[35]:

$$\mathrm{Mg_2SiO_4 + 4\ HNO_3 \rightleftharpoons 2\ Mg^{2+} + 4\ NO_3^- + H_4SiO_4} \tag{22}$$

The reaction products include silicic acid as before, but the dissolved anions are nitrates rather than bicarbonates. Some important caveats about the downstream fate of the nitrate ions are discussed at the end of this sub-section. For now, the important point is that many approaches to quantifying ERW-based carbon removal discount the non-carbonic weathering because such reactions do not show gaseous atmospheric CO_2 becoming aqueous DIC: e.g. compare the carbonic weathering Equation (17) with the non-carbonic Equation (22).

If we were to consider nitric acid weathering of calcite or magnesite – i.e. solid carbonates instead of solid silicates – then the aqueous neutralisation reaction would produce CO_2 on the right hand side rather than absorbing it on the left, as in Equation (24):

$$\mathrm{CaCO_3 + 2\ HNO_3 \rightleftharpoons Ca^{2+} + 2\ NO_3^- + H_2O + CO_2} \tag{23}$$

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[34] For (monoprotic) nitric acid the dissociation constant is ~23.4, and for (diprotic) sulphuric acid the dissociation constants are ~1,000 and ~0.01.

[35] As with many of the chemical equations presented here, several intermediate reaction steps have been omitted.

Another way this may be framed is to say that in Equation (17), the metal cations are ‘charge balanced’ by carbon-containing bicarbonate, whereas in Equation (22) (and (23)) they are ‘charge balanced’ by some other anion. This may be a useful short-hand in some contexts, but it is not necessarily consistent with how carbon is sequestered through DIC chemistry, and, as we will explore in Section 2.7.2, it may obscure the full picture. However, this ‘charge balance’ perspective, and the commensurate discounting of non-carbonic weathering, is prevalent in the ERW literature and warrants further elaboration.

Non-carbonic acid weathering will be significant in soils that are either naturally acidic or have been acidified through human activity. Even in lightly acidic soils, a large component of the overall weathering may be attributed to non-carbonic acids: Dietzen & Rosing (2023) suggests that the effect of other acids can be neglected only when the soil pH is greater than 6.3; but as can be seen in Figure 2.2, acceptable agricultural soil pH may be rather lower than this. At an extreme end, one study on a tropical sugarcane plantation estimated that only 2% of weathering was carbonic acid (Holden et al., 2024). It has been suggested that “given the impact of non-carbonic acids, soils with a pH below 5.2 may not be ideal candidates for mineral applications aimed at CO_2 uptake” (Dietzen & Rosing, 2023). A similar conclusion was reached by a study using crushed concrete as a feedstock, where the authors concluded that nitric acid negated the expected rise in dissolved inorganic carbon concentration (McDermott et al., 2019). It is repeated by Mills et al. (2024), Bertagni & Porporato (2022), and by the certification schemes covered in Section 3. Ironically, these acidified soils are the ones which may be in greatest need of alkalinity amendment, meaning that under a framing that discounts non-carbonic weathering, agricultural and carbon-removal drivers for ERW treatment do not align. Many ERW practitioners target moderately acidified soils with pH in the region of 5.5 as a compromise position (Cascade Climate, personal communication).

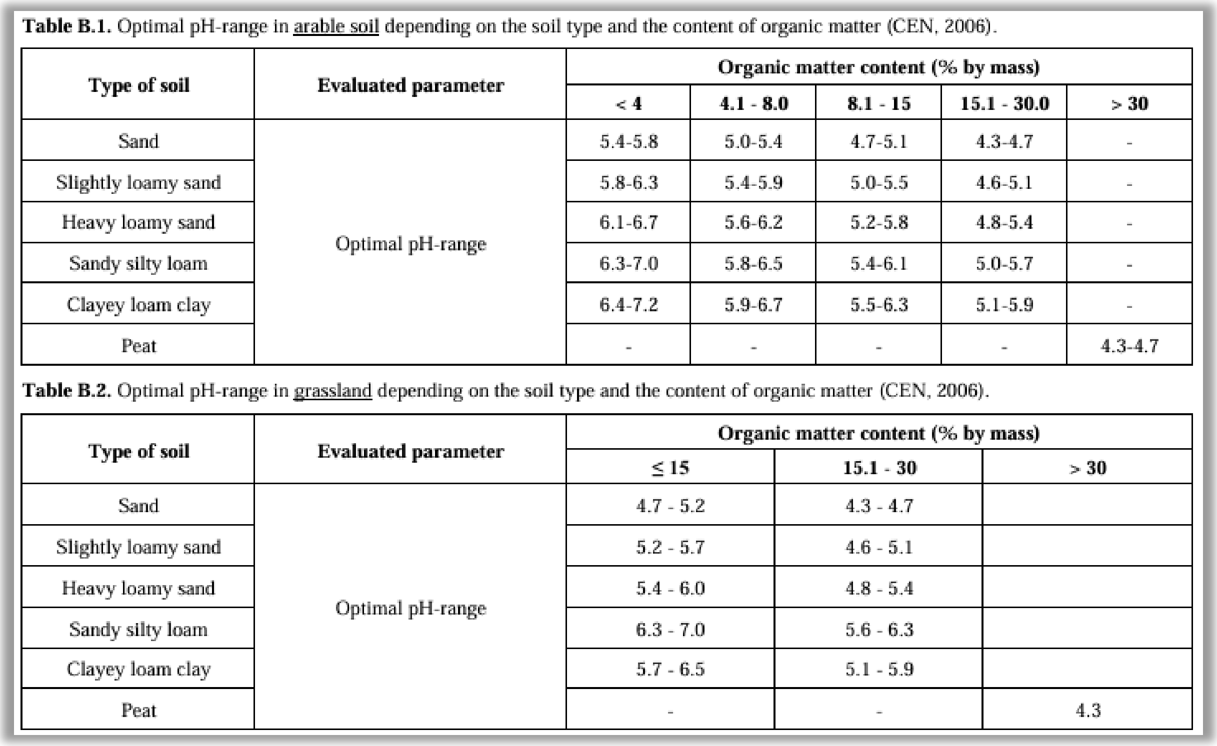

Table B.1. Optimal pH-range in arable soil depending on the soil type and the content of organic matter (CEN, 2006).

Type of soil	Evaluated parameter	Organic matter content (% by mass)				
		< 4	4.1 - 8.0	8.1 - 15	15.1 - 30.0	> 30
Sand	Optimal pH-range	5.4-5.8	5.0-5.4	4.7-5.1	4.3-4.7	-
Slightly loamy sand		5.8-6.3	5.4-5.9	5.0-5.5	4.6-5.1	-
Heavy loamy sand		6.1-6.7	5.6-6.2	5.2-5.8	4.8-5.4	-
Sandy silty loam		6.3-7.0	5.8-6.5	5.4-6.1	5.0-5.7	-
Clayey loam clay		6.4-7.2	5.9-6.7	5.5-6.3	5.1-5.9	-
Peat		-	-	-	-	4.3-4.7

Table B.2. Optimal pH-range in grassland depending on the soil type and the content of organic matter (CEN, 2006).

Type of soil	Evaluated parameter	Organic matter content (% by mass)		
		≤ 15	15.1 - 30	> 30
Sand	Optimal pH-range	4.7 - 5.2	4.3 - 4.7	
Slightly loamy sand		5.2 - 5.7	4.6 - 5.1	
Heavy loamy sand		5.4 - 6.0	4.8 - 5.4	
Sandy silty loam		6.3 - 7.0	5.6 - 6.3	
Clayey loam clay		5.7 - 6.5	5.1 - 5.9	
Peat		-	-	4.3

Figure 2.2 Optimal soil pH for agriculture

Source: Grosso et al. (2020)

Returning briefly to the nitrate ions produced in Equation (22), we note some important second-order effects. Once the nitrate ions enter the nitrogen cycle, they may be taken up by plant roots or metabolised by denitrifying bacteria. The latter pathway provides an acidity sink (hydrogen ions are reacted with the nitrate to ultimately produce gaseous molecular nitrogen and neutral water), meaning that any intervention that enhances denitrification activity can further raise the pH of the soil system and hence absorb CO_2 from the atmosphere. However, this will be difficult to measure if denitrification happens on nitrates that have leached out of the project area, and may happen on long timescales, further diluting the signal. Moreover, enhanced nitrous oxide emissions from incomplete denitrification will act to increase climate impacts (Baggs et al., 2010; Wan et al., 2021). This is an area of ongoing research.

2.5.5 Biological weathering

Plants, microbes, and invertebrates can have a significant effect on weathering dynamics and carbon sequestration; it has been suggested that an understanding of biotic factors is needed for explaining discrepancies between the results of laboratory and field weathering experiments (Vicca et al., 2022). Plants and soil biota (including earthworms) break down compounds in the soil – including rock particles – to release nutrients (Adeleke et al., 2017; Calogiuri et al., 2025). Secretion of ‘root exudates’ into the soil allows for the uptake of positively-charged nutrient ions by swapping them for positively-charged hydrogen ions (i.e. acid) at the root boundary (Taylor et al., 2017). An ERW project that boosts plant growth may accelerate the dissolution of feedstock via increased secretion of organic acids and enzymes from the roots (Drever & Stillings, 1997); similarly for soil bacteria, fungi, and other fauna (Calogiuri et al., 2023). The community consensus appears to be that the short-lived nature of these acids means that reaction with ERW feedstock results in the direct formation of DIC (Mills et al., 2024, Section 8.3): thus, under methodological framings which exclude weathering by non-carbonic acids (see the previous sub-section), organic acids would in fact be treated in the same way as carbonic acid. These effects are recognised in some existing certification schemes (Chapter 3) and in soil models referenced by them.

The extent to which different crops and soil populations respond to ERW treatment by increasing productivity and production of organic acids, and the effect this has on the net carbon balance of the ERW project, is a matter for further research (Finlay et al., 2020). Biological enhancement has been suggested as an important mechanism for accelerating weathering in the field (Van Den Berghe et al., 2024; Verbruggen et al., 2021), and both laboratory and field trials have been carried out to quantify the magnitude of the effect and determine the key influencing factors (Fuenzalida-Meriz et al., 2024; Lunstrum et al., 2023).

2.5.6 Soil chemistry

2.5.6.1 Cation exchange capacity (CEC)

A key soil fertility parameter is the ‘cation exchange capacity’ (CEC) which quantifies a soil’s ability to maintain a bank of dissolved ions available for uptake by plants. A major determinant of the CEC is the presence of soil particles that can become negatively charged[36] and hence hold on to cations. The order in which cations adsorb to soil particles is typically Al^{3+} first, followed by H^+, Ca^{2+}, Mg^{2+}, K^+, NH^{4+}, and finally Na^+, though as with all soil dynamics the sequence for a given plot will depend on ion concentrations and other soil conditions (Bijma et

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[36] This happens, for instance when an abundance of lower-valence cations (e.g. Mg^{2+} from dissolving silicates) displaces higher-valence cations (e.g. Al^{3+}) from compounds in the soil particles.

al., 2025, preprint). The presence of the cation reservoir has several implications, but the main ones for our purposes are:

1. It can function as a buffer, releasing hydrogen ions and/or precursors like NH^{4+} in response to changing cation concentrations in its environment. This would tend to counteract additional CO_2 from dissolving into the soil water (cf. Lackner, 2002; cf. Paessler, 2022).
2. It can slow the flushing of reaction products out of the soil (Paessler et al., 2025; Pogge von Strandmann et al., 2021), which both increases the likelihood of 'gross' CO_2 re-emission (e.g. if the soil is then exposed to acidic conditions), and may hinder further weathering through the build-up of reaction products.
3. It may complicate some measurement approaches for ascertaining the rate of weathering, as the number of free mineral cations in solution will be reduced if the CEC acts as an ion sink (Paessler et al., 2025).

Clay soils have particularly strong CECs, meaning that in these soils the transfer of carbon from the atmosphere to stable bicarbonates in the ocean may be slowed. Over time, compounds like aluminosilicates deriving from the ERW amendments themselves may encourage clay particle formation, enhancing the CEC.

2.5.6.2 Clay formation

Aside from enhancing the CEC, clay formation binds weathering products up into the chemical complexes of soil particles. This brings agricultural benefits (see Section 2.3), but removes them from solution such that they are incapable of participating in the geochemistry that delivers carbon removal. For example, Pogge von Strandmann et al. (2019, 2025) used isotope tracers to demonstrate rapid clay formation following ERW amendment (though these experiments did not fully replicate field conditions). Secondary mineral formation may be a permanent, transient, or cyclical phenomenon, which from a carbon accounting perspective influences whether it should be treated as a loss or a delay. At the same time, clay-rich soils offer greater nutrient and water retention, as well as SOC formation: where this benefits plant growth, there would be a tension between permanent carbon sequestration objectives and agricultural productivity objectives.

The diversity of clay types and behaviours makes this a rich seam for further research, and it would be of great interest to explore the extent to which clay formation can be expected in different contexts. ERW project operators may seek to avoid deployments in areas that are prone to clay formation as the feedstock-to-carbon-removal efficiency will be low.

2.5.6.3 Acidifying clay compounds

Clay formation may have more specific acidifying influences beyond the effects on the CEC. During weathering and clay formation, some rocks (e.g. feldspars like orthoclase, $KAlSi_3O_8$) release iron or aluminium ions into solution where they participate in hydrolysis reactions that lead to net acidity. Dissolved aluminium may undergo the following reactions that consume hydroxide ions from water molecules:

$$\begin{gathered}\mathrm{Al^{3+} + H_2O \rightleftharpoons AlOH^{2+} + H^+} \\ \mathrm{AlOH^{2+} + H_2O \rightleftharpoons Al(OH)_2{}^+ + H^+}\end{gathered} \tag{24}$$

It is commonly the case that the mineral matrix from which the Al^{3+} originates retains its negative charges in the solid phase, and there is no 'balancing' reaction to mop up the liberated hydrogen ions in Equation (24): consequently, the reaction produces net acidity. Moreover, aluminium and iron hydroxides act to buffer pH in a low range by adsorbing and releasing extra hydrogen or hydroxide ions in response to changing conditions (e.g. Al-OH becomes Al-OH-H^+ at low pH, and releases the hydrogen ion as pH increases).

Whereas cations like Ca^{2+} and Mg^{2+} are relatively soluble and can leach out of the soil in drainage water, aluminium and iron hydroxides form solids and tend to accumulate. Thus, weathering products which enhance alkalinity and buffer at a high pH are washed away, while weathering products which enhance acidity and buffer at a low pH remain. In highly weathered tropical regions, the net effect of this selective removal may perversely be to drive soils to low pH values.

ERW feedstock that promotes formation of aluminium and iron hydroxides into the soil will introduce a source of acidity, counteracting the carbon removal benefit of the project. The net effect on CO_2 will depend on the concentration of acid cations in the feedstock versus that of base cations, as well as whether soil conditions are amenable to dissolution of the different minerals (e.g. acidification from aluminium compounds may only happen below a threshold pH). Since the minerals containing these cations will have different solubilities, an ERW amendment may at first produce an alkaline effect contributing CO_2 removal, and only later an acidification effect that is a CO_2 source. More study will be needed to fully elucidate the dissolution timescales and effect on soil chemistry of different compounds in different soil conditions.

The influence of the CEC and clay-formation processes on CDR rates and efficiencies is a topic for further research, as it may influence which kinds of measurements are appropriate to quantify removals for projects operating in different soil types.

2.5.7 Other soil conditions

2.5.7.1 Biogenic CO_2

Biological activity has an impact on the CO_2 gas concentration profile of the soil. Generally speaking, a higher gaseous CO_2 concentration will result in more carbonic acid in soil-water and faster weathering (Hartmann et al., 2013); the precise relationships depend on whether CO_2 availability is a reaction bottleneck in the situation under consideration, which in turn depends on soil conditions such as water saturation (Deng et al., 2023). Observed CO_2 levels depend on the soil type and the flourishing of its biota; but they may easily be tens or hundreds of times higher than atmospheric levels, peaking at some depth into the soil (Amann et al., 2022; Amundson & Davidson, 1990; Hashimoto & Komatsu, 2006; Railsback, 2006). Ploughing ERW feedstock into the soil would maximise its exposure to elevated carbonic acid levels (Yonemura et al., 2009), as well as reducing material loss to the wind. But the effect on soil organic carbon of any additional ploughing and soil disturbance aimed at optimising ERW rates would have to be considered (cf. Sapkota, 2012).

2.5.7.2 Mineral crusts

As has already been mentioned, certain chemical conditions can lead to the formation of solid secondary minerals, such as calcite and magnesite. Bouts of rapid secondary mineral precipitation can lead to mineral crusts forming on the surface of ERW feedstock, which impedes further weathering (Oelkers et al., 2018). The presence of certain soil organisms can have a similar effect (Oelkers et al., 2015; Van Den Berghe et al., 2024).

2.6 Monitoring ERW outcomes

2.6.1 Monitoring approach

Monitoring of ERW projects can be conceptualised in three stages. The first stage, ‘feedstock characterisation’, entails understanding the properties of the ERW feedstock – rock type, mineral composition, and grain size distribution. The second stage, ‘in-field weathering’ (sometimes referred to as the ‘near-field zone’ (NFZ)), seeks to quantify the amount of feedstock that weathers and binds atmospheric CO_2 into dissolved ions over a given time period. The third stage, ‘downstream transport and sequestration’ (sometimes referred to as the ‘far-field zone’ (FFZ)), is where dissolved ions and reaction products are washed into water-courses and eventually into the ocean or deep groundwater (potentially after protracted residence times in intermediate reservoirs such as the ‘phreatic’ zone of the soil or surface water bodies (cf. Mills et al., 2024)). Figure 2.3 shows a schematic of the zones and some of the major processes at play.

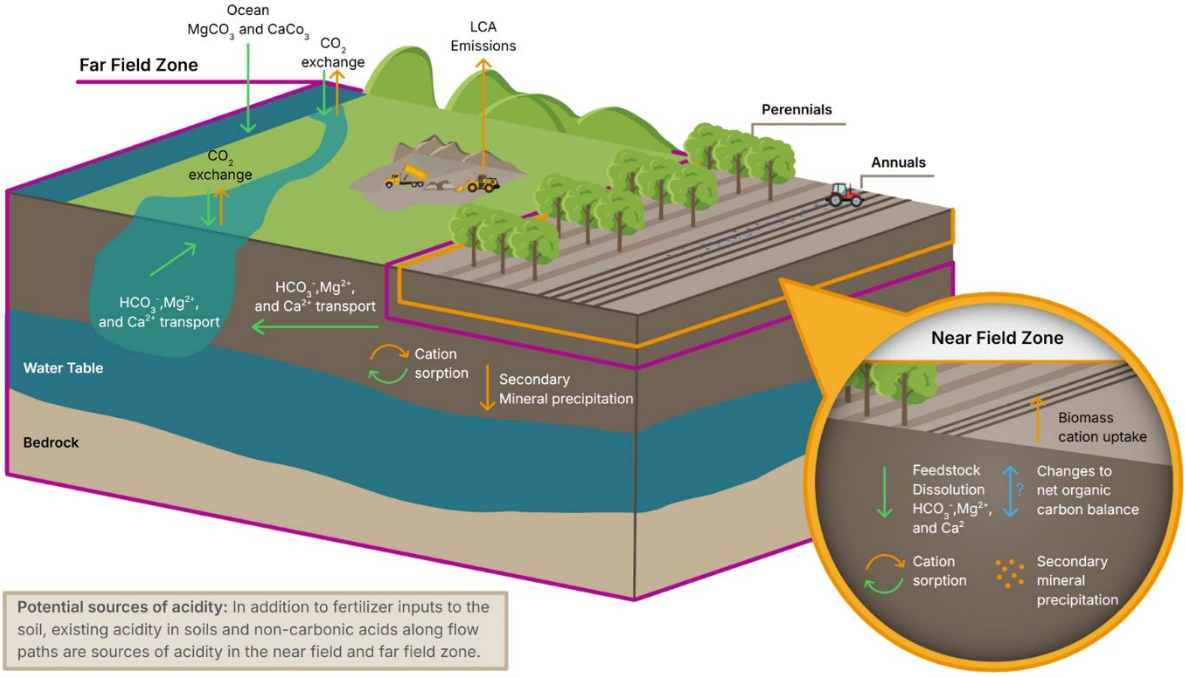

Figure 2.3 Near- and far-field zones of an ERW project and relevant chemical and transport processes

Source: Mills et al. (2024)

When it comes to carbon sequestration efficiency, it is convenient to distinguish two related terms: ‘loss’ and ‘reversal’. Carbon losses refer to re-emissions that happen before the carbon reaches its final reservoir – for example, DIC that outgasses from chalk streams as CO_2 could be counted as a loss. Carbon reversals refer to re-emissions once the carbon has reached its final reservoir. A geological example would be CO_2 leaking from a supposedly sealed underground sequestration site. Geochemical examples are less clear-cut; arguably a hypothetical outgassing from the ocean in response to falling atmospheric CO_2 concentrations could be viewed as a reversal (cf. Johnson et al., 2025), there is also the possible case that some element of our understanding of the carbonate system improves and leads to a downgrade of a previous estimate of carbon removal delivered by an activity; for ERW we will mostly be focussing on losses.

Quantification strategies relating to the three life stages above are necessarily quite different (see the Annex for a list of variables and techniques). In the first stage, lab techniques can be employed to directly characterise feedstock samples, and estimate its maximum carbon sequestration potential based on its mineral composition. The ‘modified Steinour Formula’ (Clarkson et al., 2024; Renforth, 2019) can be used for this purpose: this formula is expressed in different ways in the literature, but essentially it multiplies the concentrations of each type of metal cation in a hypothetical feedstock by the molar mass of carbonate that that type of cation could charge balance with in principle, sums the contribution of all cations, and multiplies the total by a factor that expresses the fraction of carbon that is expected to be stored as bicarbonate anions. This factor would be 1 if 100% of storage is delivered by precipitated carbonates, and 2 if 100% of storage is delivered by aqueous bicarbonate. A value of about 1.7 may be considered reasonable ocean storage.

In the second stage, once a known quantity of ERW feedstock has been spread, various indicators can be used to infer the rate of material dissolution and therefore to book-end the rate of carbon removal. However, the details of soil chemistry and biotic interactions make it non-trivial to reliably determine the fate of the various weathering products and the extent to which they have bound or facilitated the binding of atmospheric CO_2. A variety of physical and chemical measurement techniques have been used by demonstration projects, each giving insight into a different aspect of the weathering process; these can be cross-referenced to build a picture of how much carbon has gone where – including in-field carbon losses – though variability in sample measurements add uncertainty to these estimations.

In the third stage, ‘downstream transport and sequestration’, the area of interest encompasses the deep soil and the watershed: thus, any signature from the ERW project (e.g. increased bicarbonate concentration) will get rapidly diluted and hard to detect. Modelling of the local hydrology and chemical environment based on scientific study and (in the context to carbon removal certification) conservative assumptions therefore becomes necessary. We discuss some modelling approaches in Section 2.6.3.

2.6.2 In-field measurements

A variety of measurement strategies may be combined as a basis to quantify the amount of carbon dissolved by weathering. Variables that can be monitored in the field are split into solid-phase, aqueous-phase, and gas-phase.

Solid-phase measurements track the presence of feedstock-derived metals bound to soil particles as a basis to infer the quantity of metal cations (generally calcium/magnesium) that have gone into solution. This is related to the alkalinity added to water in the soil, which is in due course expected to flow with the cations into watercourses and hence the ocean.

Aqueous-phase measurements identify the weathering reaction products in the soil water or in local watercourses. Reaction products (e.g. mineral-derived cations, bicarbonate and carbonate ions, silicic acid) may be measured directly or through proxy measurements (e.g. electrical conductivity, total alkalinity).

Gas-phase measurements track the amount of gaseous CO_2 entering and leaving the soil (either by enclosing a part of the soil or through remote monitoring[37]). In agricultural contexts organic CO_2 normally dominates the overall CO_2 flux, making it difficult to distinguish the inorganic ERW signal from the noise of the short-term carbon cycle (e.g. Lockhart et al., 2023). Gas-phase monitoring may therefore be more relevant as a check against associated changes in organic carbon flows or flows of other greenhouse gases (CH_4, N_2O) than as a direct

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[37] Using techniques such as spectral and eddy covariance analysis.

measurement basis for carbon uptake by ERW. This form of monitoring is not considered by certification schemes, and is not further discussed here.

Considerable variability – spanning more than four orders of magnitude – has been observed in the efficiency of carbon removal per unit of ERW feedstock (Kukla et al., 2024). Aside from the influence of the chemical environment, crops, and biota, and heterogeneity introduced by sampling[38], different approaches to quantifying carbon removals may produce radically different results depending on what is measured, where it is measured, and on what time-scale: for instance, cation concentrations at the top of the soil column shortly after ERW feedstock application may bear little relation to the concentration of bicarbonates leaching out of the soil into the watershed (Power et al., 2025; te Pas et al., 2025; Vienne et al., 2025, preprint). Linking these various measurements to the efficiency of in-field CO_2 removals relies on assumptions that may vastly simplify the complexities of soil chemistry (e.g. cation exchange capacity and what ion types may be exchanged for a given soil under given conditions). These assumptions are critically examined or taken for granted to varying degrees in the literature; an ongoing programme of testing and validation would be needed to arrive at a more nuanced picture and to provide assurance that protocols for estimating carbon removals in the field are yielding credible results.

A list of parameters that can be measured and tracked is provided in the Annex; further description, pros and cons, and recommendations for deployment can be found in Table 1 of Mills et al. (2024) and in Table 1 of Clarkson et al. (2024).

2.6.2.1 Solid phase

Solid phase measurements are performed on samples of dry soil (where of course there is no DIC). The concentration of metals and metal ions bound in rock and soil particles is measured, ideally both before ERW feedstock application and at intervals afterwards. These measurements allow estimation of the total weathering that has taken place up until that time – this is sometimes referred to as a ‘time-integrated’ estimate, in contrast to the momentary snapshot of the DIC resident in the soil that is offered by aqueous-phase measurements (Section 2.6.2.2 below). This gives the solid-phase approach the advantage of indicating how much alkaline material has been added to the Earth system[39] without relying on assumptions about how to integrate flows over time; the disadvantage is that these measurements only constrain the ‘upstream’ process of material dissolution, rather than directly measuring the formation of DIC, and so provide no information on loss terms in the field.

As soil sampling is already widely practiced by farmers and landowners to monitor fertility, solid-phase measurements can slot into existing management systems. The quantification approach here begins with an estimate of the carbon sequestration potential of the ERW feedstock that has dissolved so far (see the modified Steinour Formula discussion in Section 2.6.1); it then subtracts off estimated reductions due to inefficiencies in converting that dissolved feedstock to additional stored carbon, and due to subsequent losses of carbon from storage. Reductions can include uptake of weathering products by plants and other biota (meaning they can’t participate in DIC chemistry), weathering by non-carbonic acids, and secondary mineral formation (cf. Mills et al., 2024, Section 5.4.1.2). The existing certification methodologies outline how these terms are to be calculated and what measurements these

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[38] The details of adequate sampling strategies and densities are out of scope for this technical review paper. Commentary on the statistics and bias in sampling rules – and the advantages of aggregating plots – can be found in Suhrhoff et al. (2025).

[39] Measuring the concentration of added metal cations can be treated as equivalent to measuring alkalinity added if the chemical make-up of the input feedstock is known. As mentioned above, the cations here are ‘conserved’, so it is easier to determine the contribution from the ERW feedstock versus other sources.

calculations are to be based on (though they still afford a degree of interpretive flexibility to project operators).

Samples are sent to a laboratory for testing. Since traditional chemical detection may struggle to distinguish small additions of weathering products above baseline concentrations, physical techniques like isotope-dilution inductively-coupled plasma mass spectrometry (ID-ICP-MS) may be employed (Reershemius et al., 2023; Van Acker et al., 2023). This is well suited to detecting trace elements in a sample and has been used in a range of ERW studies and projects.

All measurement approaches will suffer from some degree of uncertainty, as spatial variation in sample measurements and low signal-to-noise makes it challenging to reliably obtain representative values[40]. Cascade Climate recommends transparent treatment and propagation of statistical uncertainty by project operators, along with the flexibility to adjust sampling density and strategy in response to variability for each individual project (Mills et al., 2024). These measurement regimes imply a non-trivial level of expertise being required to perform the measurements for each project.

Once measurements have been obtained, the mass of relevant metals remaining in a soil sample may be compared to an estimate of the quantity applied, giving an indication of the fraction of the potential cations of each metal that have entered solution and leached out of the measurement area.

Solid phase measurement is made more challenging because of inhomogeneity in the applied material and in initial application rates – a single sample or even a set of samples may not give an accurate account of the quantity of metal that has entered solution. This is somewhat analogous to the difficulty of establishing the decay rate of carbon in biochar using in field measurements (Malins et al., 2024). In the case of ERW, however, these measurement challenges may be addressed by comparing the concentration of alkaline metals from the ERW feedstock to concentrations of immobile ‘tracer’ elements that are also present in the ERW feedstock and that are persistent in the soil. For example, if an application of weathering material were to introduce calcium and aluminium into the soil in the ratio 2:1 and the aluminium is immobile, then measuring an above-baseline calcium:aluminium ratio of 1:1 would imply that half of the applied calcium had gone into solution. Provided that the ratio of elements in the ERW feedstock is relatively homogeneous these techniques based on comparing elemental concentrations can give a more reliable basis to identify total dissolution. See (Baum et al., 2024, Section 2) for an overview, including statistical limitations of this approach.

In analysing in-field measurements one has to control for the presence of the relevant metals in the soil prior to the application of ERW feedstock. Baseline rates of weathering must be accounted for, along with application of agricultural chemicals that could introduce additional material. As discussed in Section 2.5.4, there is also the possibility that the ERW cations may be weathered by acids other than carbonic acid (cf. comments by Höglund, 2025).

2.6.2.2 Aqueous phase

Aqueous-phase measurements can be performed on water gathered from the soil at an application site (e.g. using a lysimeter) or on water in local rivers or channels. An advantage of aqueous over solid-phase measurement is that it’s one step closer to where CO_2 is actually absorbed from the atmosphere, and can avoid some modelling assumptions about how weathering products transition between phases. It is also possible to monitor chemical concentrations in water flowing away from the project site, i.e. into deep soil or into streams

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[40] A statistical treatment of measurement uncertainty as it pertains to ERW projects can be found in Suhrhoff et al. (2024).

and rivers: after this point, the in-field complications of plant uptake, non-carbonic weathering, and secondary mineral deposition no longer need to be estimated as they have all happened already[41].

A disadvantage of conventional aqueous-phase measurements (using rhizons and tension lysimeters) is that they provide only a snapshot of the carbon in the soil-water at any one time, and frequent measurements combined with understanding of hydrology and uptake of water by plants is be needed to join these snapshots up into a characterisation of how carbon is flowing through the system. This introduces a significant source of uncertainty, as there is no guarantee that samples taken at a particular time will be representative of the average behaviour in that period[42]. Moreover, sample collection protocols are laborious, and may be less familiar and less standardised than in the solid-phase approach (Baum et al., 2024). Depending on the approach taken, it may be necessary to set up monitoring stations underground (which can interfere with mechanised farming processes, as well as potentially disturbing the soil water conduits and leading to non-representative sampling) and/or at a number of water egress points. Finally, samples extracted from the soil and brought to the surface will begin to re-equilibrate with the atmosphere: this destroys the sought-after signal and hence requires either immediate readings to be taken (which can be challenging in the field) or a hermetic storage solution.

The development of new measurement devices may provide practical solutions for in-situ monitoring in future. Wey et al. (2021) describe an ion-exchange resin that accumulates ions over time (overcoming the snapshot issue and reducing sampling costs), while G. Jones et al. (2025) describe a centrifuge technique that more efficiently separates water from the soil and obviates the need for equipment burial. It is hoped that systematic validation of these devices' results will be forthcoming. Monitoring chemical composition further downstream gives a clearer picture of the behaviour and efficiency of the total soil system, including deep soil, rather than just the shallow dynamics at the application site; but measurements made downstream may also be noisier as weathering products become diluted. The optimal monitoring strategy will depend on the hydrology and characteristics of the site.

A further limitation of aqueous phase monitoring is that carbon removals that are achieved through the precipitation of secondary carbonates are not reflected in these measurements (Mills et al., 2024), and additional direct measurements of carbonate formation may be needed to accurately quantify net carbon removal (Hasemer et al., 2024).

On-site measurement can be performed using equipment to extract soil pore water samples for lab analysis, by burying devices that accumulate water for later sampling, or by burying devices that continuously aggregate readings about ion flows. Proxy measurements of electrical conductivity or alkalinity are also possible in principle, as these are correlated with ion concentration and relatively cheap and straightforward to perform (Amann & Hartmann, 2022). In practice, however, the readings are highly dependent on soil moisture and flushing events (i.e. heavy rain), and soils with high cation exchange capacity (e.g. clays) present a low signal-to-noise (Rieder et al., 2024). A standardised testing regime that has been calibrated to each soil would presumably be needed as a minimum for this approach to be effective.

Greater resolution on chemical outflows can be obtained by taking soil water readings at graded depths (Reershemius et al., 2023); but we would expect this to be more appropriate

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[41] More precisely, aqueous-phase measurements incorporate the effects of these phenomena up until the sample point – for example, secondary mineral formation that takes place in the deep soil below the sample point will not be captured.

[42] Indeed, the need for sampling to fit around the agricultural calendar may bias samples towards particular non-representative times in the season.

for research studies than commercial ERW projects, and we are not aware of any existing certification scheme that requires this approach.

Directly measuring bicarbonate concentration can be difficult using traditional methods (Manning et al., 2024)[43], and so it may have to be calculated based on other supporting measurements (Isometric, 2025a, Section 9.3.5.1.2). As with solid-phase monitoring, aqueous-phase may focus on measuring changes in the concentration of metal cations, which serve as a signature of dissolving feedstock, can be more directly quantified, but there is more uncertainty as to the correspondence between measurement results and achieved carbon removal in the field[44]. Bijma et al. (2025, preprint) argue that cation-based measurements based on the approach developed by Wolf-Gladrow et al. (2007), are superior to other options both in terms of cost and complexity, and in terms of fidelity to the underlying CDR dynamics. References to studies using such techniques can be found in Reershemius et al. (2023). On the other hand, scholars have cautioned that approaches based on inorganic carbon may need to be combined and reconciled with those based on cations to validate carbon removal estimates (Hasemer et al., 2024).

In all cases, it is important to remember that the rate of feedstock dissolution may vary in time as the weathering process progresses, potentially slowing down as the more soluble minerals are exhausted; this can skew estimates of the weathering rate if simply extrapolated to longer times (Power et al., 2025), so time-resolved tracking of weathering product concentrations and flows – both in the lab and in the field – is critical for accurate carbon removal accounting. Existing certification methodologies (Chapter 3) have put forward sampling frequencies for capturing the dynamics; development of a CRCF methodology will have to consider this issue and whether it is appropriate to establish minimum requirements.

Sampling of water from the fields where ERW feedstock is applied can be complemented by analysis of local streams and other water bodies to understand the flow of weathering products downstream (Knapp et al., 2023; Larkin et al., 2022); though the ability to detect ERW-related signals away from the project site will be dependent on details of the local hydrology, interference from background signals and on the quantities of material applied (Clarkson et al., 2024). An advantage of such approaches is that aggregation of field outflow may smooth out some of the heterogeneity associated with on-site sampling protocols.

2.6.3 Modelling CO_2 uptake and sequestration

Computational models have been used to quantify the carbon flows in all parts of the ERW process – feedstock dissolution, soil chemistry, biological feedbacks, hydrology and watershed dynamics, and inorganic carbon balance in the ocean. This is an ongoing area of research, and models are not yet ready to be relied upon for full quantitative prediction of ERW carbon dynamics. A push towards public sharing of ERW project data could expand understanding of the effectiveness of different approaches in different scenarios, and provide a more comprehensive foundation for testing and training models (Carbon Removal Alliance, 2025; Hausfather & Kukla, 2025). Cascade Climate’s Data Quarry (Cascade Climate, 2025a) is a key open data repository, and may prove to be a nexus for the future development of ERW.

At this stage of development, models can still be used to aid site selection, inform the expectations of ERW project operators, estimate downstream carbon loss that would be difficult or impossible to measure, and also deepen researchers’ understanding of the physical,

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[43] Owing, for instance, to CO_2 outgassing during sample collection and handling (Clarkson et al., 2024), a lack of uniformity in sample extraction methods and conditions, and interference from similar anion species during titration readings, which can obscure the relative proportions of each species.

[44] There is also the possibility of tracking other reaction products like silicic acid (in the case of silicate feedstock), though these are generated in a number of chemical pathways which may be less directly relevant to weathering.

chemical, and biological processes at play. An ERW project operator will typically have to combine their in-field measurements with results from downstream modelling in order to estimate the total carbon removal delivered by their activities. Modelling can also be used prior to any measurements being taken to estimate the expected carbon removal benefit of an activity (this is required by some certification bodies).

Research groups have independently developed and tested soil chemistry models to understand the speed of rock dissolution and CO_2 uptake in different environments. These range from the relatively simple and focussed (e.g. Jerden et al., 2024) to more comprehensive approaches that explicitly include hydrological and crop interactions (e.g. Lewis et al., 2021). A number of existing sophisticated soil models were identified as being suitable for adaption for ERW purposes in Taylor et al. (2017), but as the earlier references show, the slate of options continues to grow and evolve. A general account of common assumptions and simplifications can be found in Bertagni et al. (2025), which describes a newly-developed model and argues that many current theoretical estimates of weathering rates are systematically overestimated compared to experimental results.

Modelling is especially important outside of the project boundary where measurements become increasingly difficult and noisy – in the deep soil, ground-water, and streams and rivers. Watershed modelling is needed for understanding how weathering products from a specific project will get washed from the field out to sea, and how long they will spend in each reservoir along the way. Coupling this with the relevant chemical dynamics gives critical insight into the opportunities for carbon loss (Cipolla et al., 2021b, 2021a; Clarkson et al., 2024 and references therein), the role of dissolved carbon species in buffering the pH of water courses, and the extent to which carbon is re-emitted to the atmosphere through carbonate precipitation (Beerling et al., 2025; Harrington et al., 2023). Similarly, modelling of mixing and biotic activity in the coastal oceans will inform the longevity of dissolved carbon that is carried by rivers out to sea.

The accuracy of such models in predicting each stage of the carbon removal process is a topic for further investigation. While there appears to be a growing level of sophistication and integration in the modelling frameworks, the results from different models have yet to satisfactorily converge (Bertagni et al., 2025; Reershemius et al., 2023), and significant discrepancies with experimental data have been reported (Buss et al., 2024; Vienne et al., 2022).

2.7 Long-term storage

2.7.1 Carbon reservoirs and potential for losses

The optimal outcome for ERW is to durably store carbon as dissolved bicarbonate ions in the ocean or as solid carbonate minerals in terrestrial, riverine, hydrogeological, or oceanic environments (see Figure 2.4). However, there are circumstances in which bicarbonate ions and even solid carbonates can turn back into gaseous CO_2 if the chemical environment changes (Harrington et al., 2023). The main driver of loss of carbon storage would be change in pH. If the pH of water storing the bicarbonate is adequately reduced (e.g. the water flows into an acidified stream) this could lead to reconversion of the bicarbonate ions into gaseous CO_2 and subsequent outgassing of the CO_2. Similarly, if carbonate minerals are exposed to acid, they will produce CO_2.

The formation of carbonate minerals may be simultaneously considered a durable store of carbon and a carbon loss pathway. Carbonate minerals precipitate from solution when the pH of water containing bicarbonate ions and/or its saturation with metal cations and carbonate ions increases (IPCC, 2018, Section 4.3.7.4). Each unit (i.e. mol) of solid carbonate stores a unit of carbon atoms; but the precipitation simultaneously reduces the alkalinity of the solution by two units, and hence two potential units of DIC are lost (cf. Harrington et al., 2023). Mineral precipitation is treated as a permanent carbon removal loss by voluntary schemes (e.g. Isometric, 2025a, Section 7.4.1).

It should be noted that the existence of solid carbonate minerals as a carbon reservoir may complicate the quantification of in-field carbon removal. In solid-phase measurements that infer carbon removal from the amount of feedstock that has dissolved, the reduced efficiency owing to carbonate precipitation must be accounted for. Aqueous-phase measurements on the other hand will fail to recognise the additional carbon sequestration pool, and so may have to be complemented with solid-phase measurements (Mills et al., 2024).

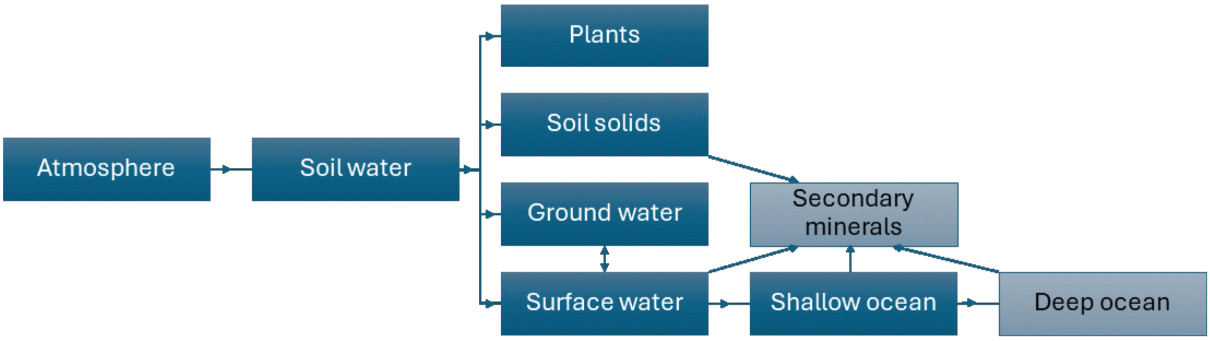

Figure 2.4 Flow of atmospheric carbon into downstream reservoirs under an ERW project; long-term storage indicated with lighter shading

A comprehensive modelling of net carbon storage from ERW requires quantification of expected carbon losses in each reservoir that the weathering products might encounter on their journey to the sea (Beerling et al., 2025; Mills et al., 2024). The modelling study by Kanzaki et al. (2025, preprint), which focussed on soil reservoirs, concluded that carbon removal may lag feedstock weathering by years to decades. There can be significant variation in the expected residence time of carbon depending on whether it is stored as bicarbonate or carbonate ions or as precipitated carbonate minerals in water bodies or soils – the latter called soil inorganic carbon (SIC)[45]. Some studies have identified only limited rates of carbonate precipitation (Beerling et al., 2025), though others have found environments where the phenomenon is more common (Jariwala et al., 2022; Kelland et al., 2020). Precipitated carbonate minerals in humid tropical climates or heavily-irrigated soils may be subject to an annual cycle of dissolution and precipitation, but longer-term accumulation of solid soil inorganic carbon may be possible in drier climates (Mills et al., 2024). Patterns of carbonate formation and dissolution have also been noted in riverine systems (Striegl et al., 2007). A reduction in the soil pH – for instance following fertiliser application – would also tend to liberate some of the carbon that had been stored as soil inorganic carbon.

When dissolved inorganic carbon washes out of soils and into streams and other bodies of surface / ground water, whether that carbon remains stored depends on the local geology[46] and on human activities which affect the pH. Acidity would again tend to promote formation of gaseous CO_2; high alkalinity (for instance in chalk- or limestone-rich areas) would promote the bicarbonate ions to dissociate to carbonate ions, and would therefore also lead to the formation of gaseous CO_2 to maintain charge balance with the available cations (cf. the discussion of re-emission in Section 2.1). The variety of possible environments, the time lags associated with

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[45] SIC can be easily distinguished from soil organic carbon (SOC) in laboratory analysis owing to the different decomposition temperature of organic matter and inorganic minerals (Manning et al., 2024, Figure 1).

[46] E.g. chalk streams are naturally alkaline with a pH between 7.4 and 8.7. Other water bodies can be naturally acidic (Mattson, 1999).

chemical kinetics, and the possible biological response to changing compound / ion concentrations, means that the residence time of dissolved carbon atoms in these waters is highly context-dependent (Stets et al., 2017).

As an example, hydro-chemical modelling of UK river catchments concluded that after wide adoption of ERW practices 10-27% of dissolved CO_2 would be re-emitted due to carbonate formation alone (Harrington et al., 2023). A global study (Knapp & Tipper, 2022) further underscores the importance of carbonate precipitation and CO_2 re-emission. It compared the potential CO_2 dissolution capacity in the soils of global river basins with those rivers' capacity to transport that CO_2 to sea, and concluded that, while most focus is on quantifying the former, the efficacy of ERW is in fact strongly determined by the latter. An indicative loss factor of 15%, encompassing all downstream carbon losses, is sometimes adopted (e.g. Kukla et al., 2024), but it is far from clear in which situations this can be considered representative. This should be considered an area for development. Estimated loss factors for each downstream stage can be found in the literature and in certification methodologies (see Chapter 3).

Ultimately, the expectation is that the carbon that remains in solution will reach the sea, where around 90% of DIC is in the form of bicarbonate ions. It is generally accepted that bicarbonate is highly stable in the ocean, with residence times of tens or hundreds of thousands of years (S. Zhang et al., 2022), although precisely modelling the longevity of bicarbonate ions in the sea is complicated by the existence of multiple equilibria and various chemical, physical, and biological forcings (Middelburg et al., 2020). It is worth pointing out, as an illustration, that certain marine calcification processes (e.g. mollusc shell growth) can convert dissolved carbon into solid carbonates such as calcite, and that this will (as with the case of carbonate precipitation in rivers) decrease CDR efficiency.

2.7.2 Full system alkalinity

As discussed above, some approaches to modelling carbon removal associated with alkalinity enhancement consider the degree to which ERW feedstock becomes associated with carbon-containing ion species only, as opposed to delivering neutralisation of other strong acids. The ERW methodologies of voluntary schemes include requirements to discount weathering that neutralises non-carbonic acids (e.g. Isometric, 2025a, Section 7.4.1), and it has been explicitly recommended that the CRCF ERW methodology deduct non-carbonic acid weathering from the carbon sequestration account (Cascade Climate, 2025c). Taking a step back, a broader argument could be made about the 'full-system' implications of adding alkalinity, which might allow one to dispense with some of the detailed modelling of carbon loss. The foundational observation for this full-system framing is that atmospheric CO_2 represents the marginal source of acidity in the Earth system. If this simplification broadly holds, then any measure that adds alkalinity[47] to the Earth's surface will in one way or another reduce the concentration of CO_2 in the atmosphere (after some equilibration period).

When strong acids in the soil weather ERW feedstock, the hydrogen ions that dissociate from those acids bind to the mineral anions (as discussed in Section 2.5.4 on soil acidity). A common premise in the literature is that under these circumstances less ERW feedstock will be available to neutralise carbonic acid from dissolved CO_2, and hence that there is less potential for carbon sequestration[48]. This treatment is in tension with the understanding that hydrogen ions in the soil water are interchangeable (irrespective of what acid they were originally associated with), and with the conclusion that, based on a full-system picture, any

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[47] To be clear, in this context, the new source of alkalinity is the crushed rock that would otherwise have been inaccessible to the terrestrial system on short time-scales. One can make an analogy with fossil fuels, which lock carbon out of the carbon cycle until they are brought to the surface and burned.

[48] As already noted, there is some evidence that carbonic acid weathering may represent a minority of total weathering in acidic soils (Holden et al., 2024).

measure to raise the pH of the soil will, all else being equal, ultimately drive net uptake of CO_2 from the atmosphere with respect to the counterfactual of no ERW application[49] (cf. Chay et al., 2025; Holden et al., 2024).

Existing certification methodologies, e.g. Isometric, treat this kind of non-carbonic acid neutralisation as a permanent loss of CO_2 removal potential, which decreases the carbon sequestration efficiency of the ERW feedstock. But when the system is considered more broadly, neutralising these strong acids in soil water encourages the formation of bicarbonate (Figure 2.1) and/or prevents CO_2 outgassing that would occur when acid washed out of the field encountered bicarbonate downstream. Hence, strong-acid weathering will still ultimately increase net CO_2 storage compared to a baseline scenario (Hausfather & Kukla, 2025; Mills et al., 2024). Another way of looking at this is that the neutralisation of soil acidity may prevent carbon losses due to downstream CO_2 emissions (Stets et al., 2017).

An inclination towards the conventional framing may arise in part from the use of cation and/or DIC concentrations in the field as a proxy for carbon removal. Carbonic-acid weathering produces DIC immediately, and this DIC can, in principle, be measured there and then (remembering that equilibration with the atmosphere means some fraction of the carbon may eventually turn into CO_2 and escape). By contrast, non-carbonic-acid weathering at low pH reduces acidity in a way that doesn't generate DIC at the project site: it could be that net DIC generation happens only when field effluent reaches a higher-pH reservoir due to avoidance of CO_2 outgassing that would have happened if the effluent were more acidic. This effect of non-carbonic-acid weathering is therefore difficult to measure as it's outside the project boundary and concerns a counterfactual (i.e. acidic field runoff that is or is not neutralised).

Another way of framing this is that carbonic acid weathering creates a local CO_2 sink, while lower-pH non-carbonic-acid weathering reduces distributed CO_2 sources. But semantics aside, they ultimately have a very similar net effect on the global balance sheet. We note that in-field measurements of DIC give a direct and verifiable quantification of carbon stored formation in the field – critical for the credible certification of removal units – but they mainly give insight into carbonic-acid weathering, and may miss the overall geochemical effect of ERW application.

The fact that the carbon sequestration associated with strong acids generally happens downstream of the project boundary produces a lag between feedstock weathering and the carbon removal event. The same can be said of the soil exchange interactions that can bind up feedstock-derived cations and anions for extended periods: even if we are confident that eventually DIC will be produced and transported to the sea to be durably sequestered, there is high uncertainty over how long this process will take for any given project. For this latter phenomenon, delay intervals spanning years to decades are certainly plausible, and this creates a timing problem for issuing carbon removal credits for a voluntary market that expects credited carbon removals to happen unambiguously at a given point in time (as is the case for geological sequestration). A research effort to systematically review and further understanding of the various time-delays in chemistry, soils, and watersheds would help to clarify consideration of what effects should be treated as a loss, what effects might be a reason to delay unit issuance, and what effects could be treated as transient and reasonably ignored.

It should be noted that even the aqueous-phase measurements of DIC produced from carbonic acid weathering in the field – i.e. the measurements that are held to be the least ambiguous basis upon which to estimate carbon removal – also suffer from lags. Measured DIC levels do not guarantee that permanent carbon removal has been delivered by the time of the

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[49] Left uncompensated, acidity would push the soil-atmosphere CO_2 balance towards the atmosphere; neutralising that acidity via ERW would lead to a net reduction the emission to the atmosphere relative to the non-ERW case.

measurement, because the DIC will continuously re-equilibrate with the atmosphere on its journey to the sea[50]. Again, it would be useful to understand this further.

To sum up, aspects of the approaches adopted by certification schemes systematically understate the net carbon removal benefit delivered by ERW in some contexts, and require costly measurement and modelling effort to quantify the creditable fraction of the potential carbon storage (Hasemer et al., 2024). The strong acid weathering pathway will in many cases be reasonably rapid, operating on the time-scale of field runoff reaching downstream water bodies (the same cannot be said for slower soil interactions). Critically, discounting weathering by strong acids deters the deployment of ERW on acidified soils – precisely those contexts where ERW progresses most rapidly and promises greatest agricultural benefits[51]. Penalising carbon removal through strong-acid weathering also constrains farmers' timing of ERW amendment application (as the activity will be treated as delivering more removal if they wait until fertiliser-derived acidity has leached out of the field into waterways); this could introduce a tension with the agricultural cultivation schedule. On the up-side, the existing approaches offer the three advantages of being conservative, being measurement-based, and of shifting focus away from sequestration pathways that take an unclear amount of time to deliver atmospheric CO_2 reductions.

2.7.2.1 Possible changes to management practices

While we have argued above that it may not be necessary to treat strong-acid weathering as a loss in carbon removal quantification, this is predicated on an implied assumption that total rates of introduction of exogenous acidity are independent of rates of ERW. Nitric acid in agricultural soils derives principally from nitrogen fertiliser use, and build-up of this acid reduces soil fertility. It is clear that ERW delivers the agricultural benefit of reducing soil acidity, but it is possible that it could also have an indirect effect of enabling greater fertiliser use and therefore the introduction of more exogenous acidity to the soil system (cf. West & McBride, 2005). This would reduce total DIC in the system, acting against the alkalinity enhancement.

We are not aware of any literature that directly discusses this type of feedback, but it may warrant further consideration, especially if ERW is adopted at a much wider scale than currently.

2.8 Other climate-related impacts

2.8.1 Organic carbon

Understanding how ERW affects SOC is complex, as it involves interactions between soil mineralogy, plant-microbe dynamics, and carbon protection mechanisms. Impacts have varied depending on study conditions (Anthony et al., 2025; Sokol et al., 2024), and a thorough predictive understanding of the carbon dynamics has yet to emerge. But it should be emphasised that biological sources and sinks dominate the overall carbon flux (Kantola et al., 2023), and an ERW project that stimulates even a small relative shift in organic carbon dynamics could produce a large absolute change that completely wipes out any potential carbon sequestration achieved through augmenting the DIC or carbonate mineral pool. (Mills

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[50] Note that irreversible downstream losses of carbon – e.g. through mineral precipitation – are accounted for in existing methodologies. What we are concerned with here is reversible, time-dependent shifting between reservoirs that may introduce more or less CO_2 into the atmosphere until the time that the weathering products reach the ocean.

[51] Reducing acidity, supporting soil biota, immobilising toxins, etc.

et al., 2024, Section 8.8.1) identifies land types that represent a high risk of substantial SOC loss.

Raising the pH of acidified soils through ERW amendment will typically increase the activity of soil microbes, which is suppressed by acidic conditions. Elements released during mineral dissolution (e.g. silicon and phosphorous) could also stimulate biological productivity (Hartmann et al., 2013). It has been demonstrated that, at least in the short term, it may stimulate respiration of soil organic carbon and hence the emission of CO_2 to the atmosphere (Dupla et al., 2024; Fuentes et al., 2006; Lei et al., 2025a; Paradelo et al., 2015; Yan et al., 2023). Other studies have pointed to a net positive impact on carbon balances over agricultural cycles using ground silicate rock (Kantola et al., 2023; Kelland et al., 2020), and iron furnace slag (Galgo et al., 2024). Additionally, Steinwidder et al. (2025) reports organic carbon stocks distributing into more stable pools[52] following ERW treatment. Existing soil models reviewed and described by certification schemes endeavour to represent these kinds of effects, and schemes' methodologies require SOC measurements to be performed as part of the ongoing monitoring plan; but the results are not incorporated into the carbon removal calculation, and no action is prescribed in response to a rise or fall in SOC stock.

Research conducted over a longer time-frame has associated liming with increased growth of soil biota, which can enrich the SOC (de Oliveira Garcia et al., 2020; Goll et al., 2021; Grosso et al., 2020). Y. Wang et al. (2021) calculated a carbon stock increase in a meta-analysis of arable soils. Xu et al. (2024) performed a field study in managed forests and concluded that the SOC gain from ERW may surpass the intended carbon sequestration through DIC. But the literature on the effect of liming still appears to be mixed (see e.g. Aye et al., 2016; X. Wang et al., 2016). In some circumstances, such as on peat lands, ERW may be linked to a significant reductions in net carbon sequestration (Klemme et al., 2022).

Long-term storage of SOC depends on its stabilisation in soil aggregates, or through adsorption to clay and metal oxides (Abramoff et al., 2018; Hemingway et al., 2019; Lavallee et al., 2020; Poeplau et al., 2020). Both of these processes are mediated by polyvalent cations like Ca^{2+}, Mg^{2+}, Fe^{3+}, and Al^{3+} (Kleber et al., 2015; Singh et al., 2018) which may be made more available through dissolution of ERW feedstock, and secondary minerals formed from weathering products act as clay precursors that can bind SOC. This is a matter of ongoing research (Buss et al., 2024; Ramos et al., 2024), and represents a possible tension between organic and inorganic forms of carbon storage as secondary mineral formation typically introduces lags to production and stabilisation of DIC levels, but simultaneously encourages SOC retention. The presence of plants introduces complexity, as changing soil chemistry may cause them to change their nutrient uptake strategies. Some studies have pointed to a destabilising effect on soil aggregates and/or SOC (Keiluweit et al., 2015; Li et al., 2021; Lin et al., 2022), while others have found a positive effect (Bi et al., 2024).

In the same vein, ERW material and weathering reaction products that are washed out of the field into downstream water bodies will influence pH and nutrient levels, and this will have knock-on consequences for the respiration and fixation of organic carbon in those water bodies. For instance, liming has been linked to growth in biotic populations and dissolved / suspended organic material, with a net increase in organic carbon (Brandrud, 2002).

In summary, the evidence suggests that under certain conditions ERW amendments can enhance organic carbon formation alongside inorganic. Due to the temporary and reversible nature of SOC stocks, this may potentially be recognised as a type of carbon farming, separate from the carbon removals delivered by manipulation of inorganic carbon geochemistry (cf. Manning et al., 2024).

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[52] Meaning the organic carbon was found to associate with minerals and aggregates in the soil.

2.8.2 Methane and nitrous oxide

ERW amendments which alter the biochemistry and biota of the application sites may have climate impacts that go beyond the geo-carbon system, affecting the net climate impact. At present, these phenomena are poorly understood.

Feedstocks like basalt contain trace metals which have been shown to promote the methane-producing activity of a type of microbe found in a diverse range of environments (D. Zhang et al., 2024), including agricultural soils. Conversely, it has been suggested that increasing soil pH could reduce methane emissions (Forrest & Wentworth, 2024a, 2024b), and there is some positive (Anthony et al., 2025; Geoghegan, 2024) and some mixed (Poblador et al., 2022) evidence for this in lab and field settings respectively.

Altering the alkalinity may also have impacts on the flux of nitrous oxide (N_2O) from agricultural soils. The use of nitrogen fertiliser is linked to about half of anthropogenic N_2O emissions (Shcherbak et al., 2014; Y. Wang et al., 2018); this originates largely from microbial action, but there are also abiotic chemical pathways (Butterbach-Bahl et al., 2013; C. M. Jones et al., 2022). The IPCC Tier-1 methodology used by many countries for national emissions reporting puts a standard 1% emission factor on nitrate fertiliser applied to mineral soils[53] (IPCC, 2006, Chapter 11.2), but the actual value in any given location will be sensitive to soil conditions (including pH), climate, the timing of fertilisation, and the local response of soil biota.

A global meta-analysis by Y. Wang et al. (2018) concluded that acidic conditions significantly increase N_2O emissions and their sensitivity to nitrogen availability (meaning that changes in fertilisation rate have outsized impacts in acidic soils), and that the standard IPPC factor generally underestimates agricultural emissions. The importance of pH is echoed by Qiu et al. (2024), which found maximum emissions in moderately acidic soils (pH around 6) that favoured N_2O-producing micro-organisms. It has been claimed that the acid-neutralising effect of ERW soil amendment in general will produce net emission reductions (Forrest & Wentworth, 2024b), and experimental (Anthony et al., 2025; Chiaravalloti et al., 2023) and modelling (Blanc-Betes et al., 2021) studies lend support to this under the specific project conditions concerned[54]. However, other studies have found no discernible effect on N_2O emissions (Poblador et al., 2022). Research on agricultural liming – the conventional approach to controlling soil pH – has similarly found instances of both positive and negative impacts on emissions (Grosso et al., 2020), which is consistent with a more complicated set of interactions in the soil.

As such, there is no simple relationships governing the effect of ERW on either methane or nitrous oxide soil emissions, and depending on the project circumstances there is potential for diminished net carbon removals once these greenhouse gases are taken into account. At present, monitoring of net fluxes requires on-site soil measurements; it is possible that future availability of sufficient calibration data will improve the accuracy of modelling estimates. As in the case of methane, it has been claimed that ERW amendment will produce net reductions (Forrest & Wentworth, 2024a), and there is some experimental support for this (Anthony et al., 2025; Chiaravalloti et al., 2023), though another study found no discernible effect (Poblador et al., 2022).

2.8.3 Albedo

Analogous to the discussion in the *Review of carbon removals through biochar* (Malins et al., 2024), the spreading of material at large scale may change the reflectiveness (albedo) of the ground, and hence the net radiative forcing. The albedo impact of ERW (Goll et al., 2021) will

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[53] The 1% figure means that 1% of the mass of applied nitrogen gets incorporated into gaseous nitrous oxide. A multiplicative factor of 44/28 gives the mass of nitrous oxide.

[54] A meta-analysis of 121 field studies reported a 21% nitrous oxide emission reduction (Y. Wang et al., 2021).

vary by feedstock: basalt tends to be dark in colour, and would hence absorb more solar radiation; limestone used for conventional liming is light in colour and would absorb less. A simple back-of-the-envelope estimate with assumed parameters suggested that the magnitude of an albedo effect could overpower the climate benefit of ERW on the decadal time-scale (Marston & Ibarra, 2025, preprint), but the literature lacks a more rigorous and specific analysis[55]. The importance of soil albedo changes will also be strongly affected by the degree of soil cover from the cropping systems in a given field.

2.9 Broader sustainability considerations

The literature identifies potential agricultural advantages from ERW, as well as a number of sustainability risks. When assessing these, it is important to consider that some ERW activity may partially or wholly displace existing liming activity, and so any net sustainability impact should be evaluated relative to the baseline of existing agricultural and soil practices.

We start with sustainability benefits before considering the risks.

2.9.1 Crop growth and soil health

The agricultural benefits of silicate application to soil (covered in Section 2.3) may allow for reductions in the use of chemical fertilisers like potassium and phosphorous fertiliser (Lewis et al., 2021), and pest control products (Edwards et al., 2017b); more research is needed to quantify these effect over longer timescales (Eufrasio et al., 2022). It has been suggested that, by protecting crop productivity and resilience, silicate amendments may aid global food security (Beerling, 2017). However, not all effects of pH and feedstock compounds on crops and soil biota are well understood at this time: laboratory and field research testing will be needed to constrain the major sources of uncertainty in terms of impacts, and to elucidate the underly chemical interactions.

2.9.2 Use of by-products

In agriculture, ERW performs the same acidity-neutralising role as conventional liming. Where industry by-products – e.g. quarry dust, metallurgical slag, demolition waste – is used as ERW feedstock, it may displace some demand for freshly quarried limestone, and also avoid the costs of processing into quicklime and slaked lime. This would reduce the environmental footprint of the quarrying industry.

Similarly, the use of certain by-products may minimise leaching and ground/water contamination from storage sites. It may also reduce the costs and land use of disposal, and promote more circular resource use.

Some by-product feedstocks which can currently be treated as a ‘free resource’ at the point of production may in time develop applications in other sectors. In this eventuality, competition may drive up feedstock prices, and indirect emissions arising from using feedstocks that could be put to other (possibly more efficient) uses may have to be considered.

2.9.3 Ocean acidification mitigation

Rising atmospheric CO_2 increases the equilibrium concentration of carbonic acid in the ocean and decreases the availability of carbonate; this threatens the marine ecosystem by impairing the activity of organisms like corals and molluscs. ERW projects can reduce the acidity flowing

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[55] It is plausible that the differentiated reflectivity of different crop types is a far more significant factor.

into coastal seas from rivers, and add buffering bicarbonate ions that moderate pH changes (Taylor et al., 2016).

2.9.4 Soil contamination

Turning now to potential costs, certain ERW feedstock may contain heavy metals (e.g. nickel, chromium) or radioactive elements (Dupla et al., 2023, 2024; IPCC, 2018, Section 4.3.7.4; McDermott et al., 2019). This goes for both naturally-occurring rocks and industrial wastes. There is a concern that these contaminants may accumulate and become bioavailable over time, especially with repeat application, potentially making crops unsuitable for consumption or affecting neighbouring ecosystems and soil biota. Depending on their mobility, contaminants may wash into surface water or percolate into ground water, threatening ecosystems and natural resources.

In principle, the suitability of a given feedstock-site combination will depend on the contaminant content and solubility of the various minerals in the feedstock, as well as pre-existing contaminant levels in the soil; both of these would have to be measured for an accurate determination of the risks (Cascade Climate, 2025b). Though we are not aware of field trials that have found evidence of problematic accumulation in the short term, the general consensus is that more experimental evidence should be gathered on trace metal behaviour under different environmental and soil conditions (Beerling et al., 2024; Skov et al., 2024). Long-term monitoring of individual projects – including crop toxicity assessments – will be necessary going forward. It has been suggested that certain ERW feedstocks could be co-deployed with crop types which mitigate or avoid the contamination issue (Suhrhoff, 2022).

2.9.5 Soil compaction

Application rates envisaged for ERW is in the tens of tonnes per hectare per year. This is significantly higher than the rates for conventional agricultural lime. ERW feedstock spreading thus entails an intensification of agricultural activity that could lead to soil compaction, with negative consequences for its structure, ecology, and productivity (Horn et al., 1995). This effect has been noted in the literature (Haque et al., 2023), but we are not aware of a thorough study of the potential broader impacts on agriculture. This may be of high concern for farmers, and will understandably present a barrier to uptake.

2.9.6 Over-alkalisation

The value of carbon removal credits could in principle induce excessive ERW application, including in certain contexts where it is not agriculturally or ecologically appropriate, or where it delivers sub-optimal climate benefit (Beerling et al., 2025; Merry et al., 2022). As an example, model research suggested that heavy ERW application in certain UK river systems with high initial pH would result in the precipitation of solid carbonate and hence loss of some proportion of dissolved inorganic carbon to the atmosphere (Harrington et al., 2023). In one studied case, pH exceeded an ecological threshold for riverine health, which may have a range of adverse effects such as interference with fish olfactory systems and promotion of toxic ammonia formation (cf. U.S. Environmental Protection Agency, 2025).

2.9.7 Sensitive ecosystems

An ERW project may affect the concentration of nutrients and toxins, as well as the pH, in downstream water courses and ecosystems of conservation significance. While this subject is relatively under-researched, there are certainly grounds to adopt a cautious approach over the risk of adverse effects to species and habitats. It is important to remember that ecosystems close to or downstream of the project site could in principle be affected by ERW application

(Harrington et al., 2023; Taylor et al., 2021). Under the EU's Water Framework Directive (WFD) (European Union, 2000), Member States overlapping each given watershed must collaborate to develop a River Basin Management Plan (RBMP) to protect and improve the body to good ecological and chemical status. Where warranted by ecological grounds in specific geographies, it may be appropriate to include in the RBMP mechanism cross-border restrictions or buffer zones for ERW projects.

While the long-term effects of ERW on soil health are under-researched, they are likely to be highly site-dependent (Forrest & Wentworth, 2024a, 2024b). Dust may impact vegetation and arthropods (DeLucia et al., 2022; Farmer, 1993; Fisher, 2024; Taylor et al., 2016), with photosynthetic organisms like lichens, liverworts, and sphagnum mosses identified as particularly vulnerable (Levy et al., 2024).

During the freshwater transport stage, ERW feedstock and weathering products may alter water chemistry and turbidity, altering species composition and food web structure (Edwards et al., 2017b; Farmer, 1993; Levy et al., 2024; Royal Society & Royal Academy of Engineering, 2017). In principle, excess sediment deposition could smother fish spawning beds and aquatic invertebrate habitats (Edwards et al., 2017b; Kemp et al., 2011), though we are not aware of definitive research relating to ERW.

Naturally acidic soils (pH < 5.5), such as those found in bogs, peatlands, and wetlands, support microbial and faunal communities that are sensitive to alkalinity changes (Forrest & Wentworth, 2024a, 2024b; Levy et al., 2024). And these land types are rich stores of carbon precisely because of the acidic conditions, and disrupting the equilibrium could have considerable greenhouse gas as well as ecological effects. Indeed, (Mills et al., 2024, Section 8.8.1) identifies various acidic soils, including peat, as having a high risk of SOC loss following chemical and pH changes that might be associated with ERW projects.

2.9.8 Land conversion

In the case that agricultural land was been recently brought into production by land use change, this is generally linked to land use change CO_2 emissions. In the context of carbon farming, projects on recently converted land may be excluded from consideration for unit issuance (cf. Section 2.2 of Allen et al. (2022)). It may be appropriate to consider similarly limiting eligibility under an ERW methodology.

2.9.9 Feedstock supply

Supply of ERW feedstock to fields is constrained by the production of suitable by-product material from existing quarries and by the cost of transport. If demand were to increase in quantity and/or in geographical distribution, then this may drive mining activity in existing and new mines (Goll et al., 2021). This would likely have some negative environmental impacts (Royal Society & Royal Academy of Engineering, 2017). Similarly, any additional crushing and transport requirements may increase energy demand, with environmental and economic implications (Eufrasio et al., 2022).

2.9.10 Air quality

Quarrying, transport, and field application of dust pose health risks, as the fine silicate particulates can pose respiratory hazards to workers and nearby populations. Around 5% of the UK's PM10 emissions come from the quarrying sector (MPA, 2022), and any increase in demand for ERW could increase crushing and grinding activities could have localised health impacts (MinEx, 2008). Spreading rock dust (with average grain size in the tens of micrometres) at project sites would represent an additional source of airborne particulate matter, which may affect project and farm workers, local residents, and ecosystems (MinEx, 2008). Additionally, some rock and mineral types are associated with high asbestos content (Forrest & Wentworth, 2024b), and may not be appropriate for ERW applications.

Adoption of best practices (e.g. timing of spreading on wet and still days and covering trucks during transport to avoid additional PM dispersal), and air quality monitoring may be necessary for controlling health risks (Forrest & Wentworth, 2024b; Skov et al., 2024). Combining feedstock application with existing farming practices such as seeding or irrigation may Adoption of best practices (e.g. timing of spreading on wet and still days and covering trucks during transport to avoid additional PM dispersal), and air quality monitoring may be necessary for controlling health risks (Forrest & Wentworth, 2024b; Skov et al., 2024). Combining feedstock application with existing farming practices such as seeding or irrigation may help to stabilise feedstock particles in the soil.

2.9.11 Community engagement

Engagement with the local community, who may be unfamiliar with ERW and concerned about potential risks (instances of this have been documented by Cohen (2024)), is required in some existing schemes. Voluntary schemes may include consultations and/or information distribution into their protocols to improve trust and legitimacy. The value of transparent engagement is underscored by instances of public backlash to proposed carbon removal deployments (e.g. BBC News, 2025). This may be particularly relevant where sensitive or protected landscapes and species are involved.

2.9.12 Protections

The regulation and enforcement of health, social, and environmental protections vary by country, and so projects that uphold their statutory obligations may still deliver markedly different outcomes depending on where they're based (Oppon et al., 2024).

Certification bodies' requirements may go beyond local regulations. In our review we found the level of detail of such provisions to be limited, with discretion presumably left in the hands of project operators and/or auditors. For example, Isometric exclude projects that negatively impact crop yields (Isometric, 2025a, Section 4.0). They require yields on treatment and control plots to be gathered (Isometric, 2025a, Section 9.3.2), but there is no guidance on how to compare or interpret these numbers given the variability inherent to farming projects. Similarly, the standard states that measures must be taken to "identify and eliminate potential risks to terrestrial and aquatic ecosystems and biodiversity" (Isometric, 2025a, Section 5.0), but the only ERW-specific risks that are identified are "the release of metals such as nickel (Ni) and chromium (Cr) or other harmful contaminants such as asbestiform minerals" (Isometric, 2025a, Section 5.2). It is therefore unclear what the tolerances will be in practice for certification of projects that could be associated with any of the other negative impacts discussed above.

3 Assessment of relevant methodologies from private standards

3.1 Puro.earth ERW methodology

Puro.earth's ERW methodology is advertised as the first carbon removal crediting standard for enhanced rock weathering brought to the voluntary carbon market. It was first introduced in 2022, followed by an updated version released in October 2024 (Puro, 2024) which is reviewed in this paper[56]. The methodology is aligned with the Integrity Council for the Voluntary Carbon Market (IC-VCM) Core Carbon Principles.

The ERW methodology is based on a Life Cycle Assessment (LCA) approach which includes an assessment of the entire project's emissions resulting from the ERW process. The LCA is required to follows the ISO standards 14040/44. The processes in the system boundaries are split into five phases, with the last two phases occurring simultaneously:

- Mining of rock;
- Processing of rock;
- Transportation to the application site;
- Application to site;
- Weathering phase;
- Carbon fate in environment.

Mining or extraction of ERW feedstock can be considered "burden-free" if the material is identified as a waste or secondary product. The weathering phase is when the material is weathered, leading to carbon capture and storage, and involves monitoring activities such as sampling, analysis, and simulation. The 'carbon fate in environment' phase covers the processes that occur when stored carbon travels in the environment, with potential re-emission of CO_2, which is required to be quantified in the quantification of carbon removals. If co-products are generated, the operator may either ignore treat them as 'burden free' (i.e. give the main process no credit for the generation of co-products) or may allocate burdens between them, with a requirement to document and justify the choice of allocation factors.

Spatial and temporal aspects of the ERW projects are defined and accounted for in the LCA. The spatial boundary includes the mine location where the rock or waste originates, the processing facility, the application site (ideally to be specified with a map or geospatial vector data as a shapefile), and the catchment area where the stored carbon is expected to evolve (soil, groundwater, watershed, rivers, ocean entry point, aquifer). The 'time boundaries' that must be declared are the timing of each process from rock mining to soil application, which are assumed to happen within a year and are therefore allowed to be described with a single date, a conservative estimate of the time needed for to reach various weathering thresholds (i.e. weathering completion (%) over time (years)), and an order of magnitude estimate of the time scales for the movement of captured carbon in the environment.

The ERW project activity is defined as the application of a given type and amount of material to a specific soil location, with specified application rates, granulometry, and soil incorporation depth. Application at multiple sites may be included in a single project activity provided that all application sites adhere to the same environmental and other relevant regulations, and have broadly consistent geographic location, climatic conditions, type of applied feedstock, soil type and risk profile related to potentially toxic elements (PTE risk). The application sites can be

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[56] Puro's updated methodology is expected in the second half of 2025 (Puro, 2025).

physically separated and do not need to be contiguous. Projects at sites that do not meet these criteria, and/or ERW applications in different years, must be treated as separate activities.

The methodology requires that third party auditors verify the LCA approach and project emissions by visiting the production site and examining the accuracy of the data submitted.

The optimal conditions for weathering are identified as low pH, high water availability, high temperature, high CO_2 partial pressure and increased reactive surface area of rock. In general, warm tropical regions are known for faster enhanced rock weathering. It does not specify particular rock types or ERW feedstocks; however, it does provide an environmental risk assessment (ERA) framework in relation to toxicity and require that operators comply with any local legal requirements for material applied to agricultural soils (e.g. the EU Fertilising Products Regulation), This is discussed further in Section 3.1.5.

3.1.1 Quantification

The methodology states that despite various studies, “comprehensive experimental data on weathering rates in field conditions is still largely missing”, and notes that there is no scientific consensus on the assessment of ERW nor an IPCC methodology. Operators are therefore required to propose a simulation approach for the amount of CDR delivered and an in-field measurement approach to validate and if necessary to calibrate the simulation results. A written description of the approach used to quantify the weathering process and the expected carbon removal from the activity is required. This must specify the empirical quantification and site-specific modelling of the weathering process. The operator must also use an LCA approach to assess associated GHG emissions. The LCA analysis and the scientifically justified estimation of the expected carbon removals must demonstrate a “net-negative overall carbon footprint” for the whole project activity to be eligible for the ERW activity.

CORCs issued are based on estimated actual amounts of CDR, which must be evidenced by in-field measurements, and Puro’s CDR certificates (also known as CORCs) are issued gradually over the lifetime of the project as the required measurements provide evidence of weathering. The methodology notes that ERW will therefore be associated with long financial return periods, dependant on how quickly the weathering reactions occur. As carbon removal by ERW cannot be directly measured (the main medium of carbon storage is bicarbonate ions in solution in dispersed bodies of water) these CDR estimates must be obtained by combining the simulation approach with the in-field measurements.

The amount of Carbon Removal Credits (CORCs) generated during a monitoring period is calculated as the gross amount of CO_2 sequestered through weathering reactions minus the supply chain emissions, potential losses and economic leakage, following the Equation (25) with all units in tCO_2e:

$$\mathrm{CORCs} = C_{\mathrm{stored}} - E_{\mathrm{project}} - E_{\mathrm{leakage}} - E_{\mathrm{loss}} \tag{25}$$

Where:

CORCs is the net amount of CO_2 equivalents removed by the applied weathering material.

C_{stored} is the gross amount of CO_2 stored via weathering of the applied rock during the monitoring period. This includes the generation of carbonate and bicarbonate ions and potential precipitation of solid carbonate minerals. The quantification approach of C_{stored} is based on solid or aqueous phase measurements from in-field soil or water samples. It includes measurement of weathering signals relative to control site measurements. Control sites must be representative of the application site in terms of pH, soil texture, and climate conditions, and statistically significant in sample count and/or total area. The operator must propose a quantification approach for consideration by the issuing body, and this approach must include solid or aqueous phase measurement of the applied material. The issuing body analyses the quantification approach and determines whether it is acceptable.

$E_{project}$ is the total life cycle emissions arising from the whole supply chain of the ERW activity, derived from the full LCA. For any activity included in the life cycle inventory, a full scope of emissions must be accounted for, including all life cycle stages. Project emissions must be updated in each monitoring period with actual measured and recorded activity data (e.g. transport distances, fuel, energy, and material consumption).

$E_{leakage}$ is the total GHG emissions due to negative economic leakage. The LCA should include primary material extraction if economic leakage is likely to occur. This ensures that all indirect emissions are accounted for. Economic leakage can affect the net carbon footprint of the ERW project, potentially reducing the overall effectiveness of the carbon removal activity.

E_{loss} is the total re-emissions of initially sequestered CO_2. These should be accounted for based on loss pathways (re-emission pathways) which are defined as biogeochemical events or processes that occur after the initial CO_2 sequestration through chemical weathering. They result in the release of sequestered carbon back into the atmosphere or convert it into a form that is not considered permanently stored. Any unknown re-emissions after the CORCs issuance are referred to as reversals.

The total GHG emissions due to loss pathways are calculated by summing the emissions from each identified loss pathway (S):

$$E_{\text{loss}} = \sum_{i \in S} mCO_2e_i \tag{26}$$

Where:

mCO_2e_i is the total mass of GHGs emitted from each loss pathway in tCO_2e.

Common loss pathways include:

- neutralisation of acids other than carbonic acid:
  - measured by a combination of direct measurement of the concentrations of major anions in the soil porewaters using ion chromatography and an estimate of the total volume of water infiltrated into the soil.
  - conservative estimation outlined by Dietzen and Rosing (2023) to determine the proportion of weathering by sources of acidity other than carbonic acid. This involves calculating the carbonate system speciation based on soil pH and pCO_2 (or other parameters like total alkalinity and dissolved inorganic carbon (DIC)).
  - alternative method that should be approved by Puro.
- plant uptake of major cations (Ca^{2+}, Mg^{2+}, Na^+, and K^+):
  - Emissions to be set at zero if the effect of plant uptake of major cations is already accounted for in the quantification approach or if no crops are being harvested from the application site.
  - Measure the total base cation content of harvested plant biomass and estimate the total plant biomass removed from the field. Compare these measurements with control plots to identify the baseline plant cation uptake.
  - If direct measurement is not possible, an estimation of 5% of the value of C_{stored} to be applied.
- secondary mineral formation:
  - Emissions to be set at zero if the effect of precipitation of secondary pedogenic carbonates is already accounted for in the quantification approach used (e.g.,

  measuring the alkalinity flux at the base of the soil profile through tile drainage, deep lysimeters, or catchment drainage waters).
  - Direct measurements of the soil inorganic carbon stocks of the application sites using methods like calcimetry and to compare the measurements with control plots to identify the baseline changes in soil inorganic carbon stocks.
- losses from surface water systems to be estimated at 5% of the value of C_{stored};
- losses from marine systems to be estimated at 10% of the value of C_{stored}.

3.1.2 Indirect emissions and leakage

The methodology mentions indirect emissions under the term "economic leakage" referring to GHG emissions and removals outside the system boundaries of the project, resulting from the implementation of the project. Economic leakage is different from the physical leakage, which is when the carbon is release back into the atmosphere. Economic leakage could occur when the weathering material was already used for another product or service. If demand persists, this could entail the extraction of additional primary material, which must be included in the LCA.

The weathering phase of the LCA approach includes a requirement to assess emissions due to 'direct land use changes'. The definition of direct land use change emissions used in this methodology differs from e.g. the definition used in the Renewable Energy Directive, in that rather than focusing on changes in biomass and soil carbon stock due to a change from an existing land use to agricultural land use, it relates to any potential change in soil GHG emissions (biogenic carbon, methane, and nitrous oxide) after the application of the weathering material, relative to a non-application baseline. The methodology notes that precise quantification of these emissions would require on-site measurements over multiple years, but that for the LCA these emissions must be estimated with best available knowledge. If direct land use change leads to increased emissions, it must be included in the quantification of the carbon removals; otherwise if it is decreased emissions then it cannot be included in the quantification of carbon removals but may be presented as co-benefits.

3.1.3 Additionality and baselining

Projects are required to demonstrate that the CO_2 removals are a direct result of the project and would not have occurred without it. Even if the project receives substantial non-carbon finance support, it can still be considered additional if investment is required, risk is present, and/or human capital must be developed. The project must not be required by existing laws, regulations, or other binding obligations, referencing the "Microsoft criteria for high-quality carbon dioxide removal".

The financial test requires full project financials and a counterfactual analysis which is based on baselines that must be project-specific, conservative, and periodically updated. This analysis should demonstrate what would likely have happened without the carbon finance, ensuring that the removals are indeed additional. Puro refers to measuring baseline plant uptake and baseline changes in soil inorganic carbon stocks in the control plots but a more explicit specification of how this can be achieved is not included. It does not specify what the counterfactual analysis could include e.g. carbon removals from natural weathering or pre-existing agricultural practices.

3.1.4 Long-term storage and liability

Puro states that enhanced weathering is "among the most permanent forms of carbon removal, with relatively low risk of natural or anthropogenic reversibility." There is the potential risk of outgassing CO_2 in the groundwater, surface water and through long-term carbonate formation in oceans, however, these emissions are addressed by estimation in the quantification of the net carbon removals.

3.1.5 Sustainability

The Puro methodology provides guidance (Section 4.2) on assessing risks associated with PTEs in ERW projects. PTEs include heavy metals (e.g., nickel, chromium VI, cadmium, uranium), radionuclides, and other contaminants like asbestos. These elements can have detrimental effects on ecosystems and human health if present above certain levels.

The focus of the environmental assessment is to prevent environmental risks through proper selection of crushed materials, site selection, and analysis of food safety impacts. In addition to a requirement for a safe weathering material sourcing, the project developed must provide evidence regarding the right or authorisation to spread the weathering material onto the application sites. The ERA must be considered by Puro and validated by an independent third party, but the methodology is not explicit (other than in relation to the toxicity limit values) about the circumstances under which Puro would withhold certification due to environmental risk.

The ERA must follow guidance from the U.S. Environmental Protection Agency, the European Food Safety Agency, or other locally relevant institutions. It is required to be reviewed by an independent third-party.

- Problem Formulation: Define the problem, scope, and goal of the risk assessment specific to the ERW activity.
- Hazard Characterisation: Identify elements of concern, their pre-existing levels, concentrations added by the project, and associated dynamics and mechanisms.
- Exposure Characterisation: Determine exposure pathways for humans, fauna, flora, and water resources, and assess their likelihood and seriousness.
- Risk Characterisation: Combine findings from hazard and exposure characterisation to determine the level of risk and evaluate the legality of the activity.
- Risk Mitigation Measures: Define measures to mitigate risks to acceptable levels, such as cultivating specific crops, using phytoremediation methods, or adding biochar for metal immobilisation.
- Conclusion: Summarise the overall risk, applicability, and legality of the ERW activity, stating whether it is permitted, permitted with mitigation measures, or not permitted.

The project developer is required to assess the environmental risks at the specific site of application and the receiving catchment. The final stage is the conclusion regarding the ERW activity, which may result in one of the following: “permitted, with no major risks”, “permitted provided that adequate mitigation measures are implemented”, or “not permitted due to non-manageable risk”. It is not clear whether there is any obligation to update the ERA after the ERW activity has commenced.

The methodology states that it is the developer’s responsibility to manage any identified risks and implement the mitigation measures defined in the risk assessment. It is required that the conclusions on the environmental risk assessment be supported by laboratory results from analyses of weathering material and soil samples, determining concentrations of PTEs. It is required that the risk assessment includes at least:

- a metric based on maximum concentration in soil;
- a metric based on maximum concentration in the weathering material.

The methodology requires that the operator shall ‘follow’ the impact on crops after the application of weathering material to agricultural soil, but is not prescriptive about what this should include. It recommends monitoring the absorption of major cations (Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}) by plants. Periodic measurements of soil organic carbon content at the application sites must be conducted in accordance with the monitoring plan. Note that this requirement is for research purposes and does not affect the quantification of CO_2 removals.

There are regulations that exist to control heavy metal concentrations in soils, with limits on maximum concentrations in applied materials, application rates, and soil concentrations. The methodology allows for the option of combining limitations provisions.

The EU Fertiliser Regulation presents maximum metal concentrations in applied material. These limits apply to inorganic soil improvers, such as rock dust and crushed rocks within an agricultural context.

There is also the *Finnish Governmental Decree on Soil Contamination* which has values for maximum metal concentrations in soil (top 30cm).

The *EU Sludge Directive* sets limits on the concentration of heavy metals in sewage sludge used in agriculture, as well as the maximum allowable concentrations in soils and the maximum addition over a 10-year period. Each member state has ratified the directive with adjustments to specific local conditions. Some regulations also consider the bioavailability of heavy metals. This approach is less conservative but can be relevant in specific contexts, such as determining the speciation of chromium or assessing nickel bioavailability in regions with naturally high nickel concentrations. More detail on examples of toxicity limits is available in section 4.2 of the methodology.

The methodology has requirements for social (section 4.3) and environmental (section 4.4) safeguards. The ERW project must ensure there is no negative impact on local communities and the environment, with the focus on minimising harm and maximising positive outcomes. One special requirement for this type of carbon removal activity is the provision of ongoing communication to the local community about the acceptability limits for contaminants followed in the project, including the potential health risks and limits concerning toxic contaminants in the applied weathering material.

3.1.6 MRV

At the start of an activity the estimated amount of carbon removals must be provided based on site-specific simulation modelling.

The simulated result is validated with post-application measuring and reporting. The in-field measurements are used to determine whether the simulation represents the measured weathering signal over time. The in-field measurements can be used to update the weathering model (Section 7.4.1).

The project developer is required to conduct ongoing in-field measurements throughout the project's lifetime to quantify the amount of CO_2 sequestered. They are required to quantify and keep records of all parameters needed to quantify CO_2 removal: quantity of weathering material extracted and applied, direct use of energy and fuels, and GHG emissions from the process.

A geochemical assay is required to be performed at least once per every 5000 tonnes of weathering material applied or when sourcing from a new supplier or quarry. The geochemical assay informs about particle size distribution, chemical oxide composition, mineralogy, and major/minor chemical elements.

Baseline sampling for soil analysis before application of weathering material is specified to determine soil type texture, pH, cation exchange capacity, organic and inorganic carbon content, major cations, and potentially toxic elements. The sampling information should be statistically representative of the project activity with recommended sampling density of 1 sample per hectare.

The methodology includes guidelines for correct soil and water sampling for measurements of weathering signals. Two collection methods for soil sampling are suggested, one for homogenous plots and a second one recommended for heterogenous or large areas, with recommended sampling depth between 0-20cm. It is not quite clear whether the same sampling depth is required for aqueous measurements.

For water sampling i.e. the collection of water from soil pores, the methodology discusses vacuum tools (i.e. rhizon samplers and porous ceramic cups) and passive tools such as lysimeter to measure leachate soil water, particularly effective for measuring leached alkalinity. Time of collection of the sample and time of actual measurement should be considered in the interpretation of the results.

The methodology suggests parameters used for quantifying CDR and also used as indicators for weathering:

- Total alkalinity: increased total alkalinity suggests that weathering is taking place. The flux of ions (alkalinity flux) into solution during mineral dissolution is the key measure for enhanced weathering rate. Total alkalinity can be measured via titration or colorimetry.
- Concentration of major cations: Higher concentrations of major cations (e.g., Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}) in soil pore water indicate active weathering processes. This suggests that minerals are dissolving and releasing cations, which can then participate in CO_2 sequestration. It is measured via spectroscopic methods.
- Total inorganic carbon: measures the total amount of inorganic carbon in the form of carbonates, bicarbonates, carbonic acid, and CO_2. The increased TIC suggests weathering has occurred. It can be performed with a CN analyser or other methods.
- pH: measures the concentration of hydrogen ions in soil pore waters, indicating acidity. In acidic conditions (low pH), minerals tend to dissolve more rapidly, releasing cations that can react with CO_2. An increase in soil pH can indicate the release of alkaline substances from weathering minerals, suggesting active CO_2 sequestration processes. Soil pH influences microbial activity, which can affect the breakdown of organic matter and the release of CO_2. Certain microbes thrive in specific pH ranges, impacting the overall CO_2 dynamics in the soil. pH will also affect the speciation of dissolved inorganic carbon between carbonic acid, bicarbonate, and carbonate ions.
- Soil electrical conductivity: measures the ability of soil pore waters to transmit electrical current, indicating water-soluble salt content and it can be used to predict CO_2 sequestration. Higher electrical conductivity values indicate increased concentrations of dissolved ions, suggesting active weathering processes. There are some relationships between electrical conductivity and total alkalinity, however more research is required to determine total alkalinity from electrical conductivity.
- Isotope ratios: Changes in isotope ratios can indicate specific processes, such as the dissolution of minerals and the formation of carbonates, which are involved in CO_2 sequestration. Isotope ratios can be used to identify the origin of carbon in the soil.

3.1.6.1 ERW simulation models

Puro does not prescribe the use of one specific simulation ERW model: instead it provides suggestions for project-specific and more scientifically robust simulation results. Puro states that models should be built to specifically simulate enhanced weathering and that they should be based on published, peer-reviewed research and they should be theory-based and data driven rather than purely empirical ones. The model should present the dynamic interactions of the “rock-soil-plant-atmosphere system” and be project specific. The parameters can vary but they should include thermodynamic and kinetic data such as temperature and partial pressure, mass and density, activities and solubilities, reaction rate constants. Further processes to be included are relevant biogeochemical reactions, transport of fluids and chemical species, changes due to environment, particle size effects and more. The output of the model should be able to quantify carbon removals over time relevant to the application of weathering material. The model should be able to model uncertainty related to both the behaviour of the ERW field and input parameters, with offering options to optimise performance and reduce uncertainty. Puro states that the models should be open for validation, transparent and backed by experimental EWR data. Some of the models referenced in the Puro methodology are Kelland model, SCEPTER, Dynamic river network and cGENIE.

Kelland model is a one-dimensional soil geochemical reactive transport model that estimates CDR linked to basalt weathering over periods of several decades (Reershemius et al., 2023). The model assumes the movement of rainwater or irrigation water through a 30 cm soil profile, with basalt added to the top 5 cm of soil.

SCEPTER model stands for Soil Cycles of Elements simulator for Predicting TERrestrial regulation of greenhouse gases (Kanzaki et al., 2022) and is one part of the Lithos enhanced weathering modelling framework, which estimates carbon capture during weathering and the amount of carbon storage in rivers and the oceans over a given timeframe. SCEPTER is a one-dimensional soil geochemical reactive transport model and it is stage one of the model framework. The model describes the movement of rainwater or irrigation water through soil layers using advection and diffusion processes, while mineral dissolution and precipitation are determined by geochemical thermodynamics and kinetics. Grain size, porosity, and permeability are treated as variables that can change over time.

Dynamic river network and cGENIE are separate models within the Lithos framework which monitor enhanced weathering products in surface waters and oceans over multiple decades to thousands of years (Reinhard et al., 2020; Ridgwell et al., 2007). The models assume the movement of water from the weathering zone through a soil column into surface waters, with river waters subsequently entering a homogenous surface ocean area at a designated point. The processes modelled include carbonic acid system re-equilibration in rivers and oceans, clay and carbonate minerals formation and mixing of various water masses in rivers and oceans.

3.2 Isometric Enhanced Weathering in Agriculture

This paper reviews version v1.1 of Isometric’s protocol for Enhanced Weathering in Agriculture, dated 22nd January 2025 (Isometric, 2025a). Isometric enhanced weathering protocol follows the Isometric Standard as the main guiding document, and is further informed by ISO standards (ISO 14064-2: 2019, ISO 14064-3: 2019, ISO 14040: 2006, ISO 14044: 2006). Additional reference standards for enhanced rock weathering that were reviewed with attempts to align with best practice include: Global Rock C-sink, Guidelines for the Certification of Carbon Sinks created by Enhanced Rock Weathering in Croplands, v0.9, Carbon Standards International and Puro.earth ERW methodology (version 2022). Other references that Isometric points out include:

- Criteria for High-Quality Carbon Dioxide Removal, Carbon Direct, Microsoft, 2023
- BS EN 15978:2011 Sustainability of construction works - Assessment of environmental performance of buildings - Calculation method
- Foundations for Carbon Dioxide Quantification in Enhanced Rock Weathering Deployments, Cascade Climate, 2024

Isometric's enhanced weathering protocol is applicable to projects using crushed silicate rocks (containing metals Ca, Mg, K, Na) on agricultural land as defined by the United Nations Food and Agriculture Organization (FAO), including row cropland and pastureland. The methodology covers alkalinity export from soils to the ocean via riverine transport. Ineligible projects include any that apply non-silicate material (e.g. carbonate feedstocks), any that operate in non-agricultural settings, and projects that "lead to a sustained and substantial net decrease on crop yields". Secondary mineral formation in soils is monitored but it is not credited towards carbon removals.

Isometric requires a cradle-to-grave lifecycle assessment of GHG emissions from the entire system boundary (all sources, sinks and reservoirs), including activities from the project establishment through operations to the end-of-life activities. Isometric outlines the required GHG emissions by activity, scope and timescales of emissions and accounting allocation.

Project establishment activities (e.g. quarrying, crushing, grinding, and processing, transportation of material) and end-of-life emissions are accounted for in the first reporting period while operation emissions from sampling and analysis and CO_2 stored are accounted for in the relevant reporting period. All direct and indirect emission sources related to the ERW project activity must be identified including any emissions outside the defined categories, which are referred to as miscellaneous emissions.

Unlike the Puro ERW methodology, secondary impacts on the GHG emissions due to enhanced weathering in agricultural soils such as emissions of nitrous oxide (N_2O) and methane (CH_4) are not included but something to be reviewed in the future as more scientific evidence is available. Still, project developers are encouraged to monitor the flux of soil gas (CO_2, CH_4, N_2O) in project areas.

Embodied emissions from waste products can also be excluded if specific criteria are met (Isometric protocol section 7.1.1.3). Additionally, activities that were already occurring and would continue without the EW project (e.g. rock spreading during seeding) can be omitted from the system boundary.

3.2.1 Quantification

The Isometric methodology defines two quantification approaches for determining carbon stored, namely **soil-based quantification** (quantify the loss of base cations from treated soil over time) and **porewater-based quantification** (quantify the accumulation of base cations in soil porewater).

Isometric specifies crediting based on the quantification approach. Where soil measurements are used, removals occur when the base cations are exported from the "near-field zones" i.e. the upper soil column[57]. If aqueous measurements are used, base cations can be measured and credited within the near-field zone if they appear in the porewater. In this context, Isometric is more specific than Puro in determining where the removals and losses could be quantified by the terms of near- and far-field zones.

The reporting period represents the interval of time over which carbon removals are calculated and reported for verification. The reporting period begins after the spreading of the material has occurred and ends once the weathering potential has been realised and MRV activities have ceased. Total net CDR is calculated for each reporting period as follows with all units in tCO_2e:

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[57] Isometric recommends the depth of the near-field zone to be the deeper of 20cm below the surface, or the depth of tillage plus a buffer of 5 to 10 cm.

$$CO_2e_{Removal,RP} = CO_2e_{Stored,RP} - CO_2e_{Counterfactual,RP} - CO_2e_{Emissions,RP} \tag{27}$$

Where:

$CO_2e_{Removal,\ RP}$ is the total net amount of CO_2 equivalents removed for the reporting period.

$CO_2e_{Stored,\ RP}$ is the total CO_2 removed from the atmosphere and stored as inorganic carbon for the reporting period, assessed by either a solid-phase or aqueous phase measurement approach, discussed further below.

$CO_2e_{Counterfactual,\ RP}$ is the total counterfactual CO_2 removed from the atmosphere and stored as inorganic carbon in the solid or aqueous form for the reporting period. This is the CO_2 that would have been removed and stored in the baseline scenario without the project. It is calculated using the control plot and business as usual farming practices. The counterfactual must also consider the natural weathering of feedstock if it was stored without project intervention. It can be quantified by combining direct measurements and models and the equation for **$CO_2e_{Stored,\ RP}$** from aqueous phase.

$CO_2e_{Emissions,\ RP}$ is the total GHG emissions for the reporting period. This includes emissions associated with project establishment, emissions during the reporting period form operations, end-of-life emissions that would occur after the reporting period but allocated to the reporting period, and leakage emissions outside the system boundary. Within this term, project establishment emissions are allocated either as a one off term in the first reporting period, or across the first half of the anticipated project lifetime (calculated as the point at which 50% of weathering potential will be realised).

Isometric does not rely on models to simulate weathering rates as much as Puro; however, they do acknowledge that reactive transport models and other models could improve due to more calibration data and thus remains open to include models in the future revisions of the protocol. Isometric refers to several types of models for enhanced weathering with different purposes: geochemical models, equilibrium models, carbonic acid system models.

Projects are required to report on all input variables in the carbon removal calculation and their uncertainty caused at a project scale. A sensitivity analysis describes the impact of each input parameter on the overall net carbon removal. Any input variables with <1% impact change in the net carbon removal can be omitted.

3.2.1.1 Solid-phase approach, $CO_2e_{Stored,RP}$

If quantified using the soil-based approach, $CO_2e_{Stored,RP}$ is calculated by subtracting the inorganic carbon losses[58] from the total CO_2 removed from the release of base cations from feedstock. The latter is measured via direct geochemical observation of the project area before and after feedstock application. A statistically significant loss of cations must be demonstrated, the criteria of which is also outlined. This is equation 3 in Section 7.4.1.2 in the Isometric Protocol.

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[58] The inorganic carbon losses can be due to soil column biogeochemical processes and also downstream riverine and marine losses. These losses include plant uptake of base cations, formation of secondary silicates, carbonate precipitation, non-carbonic acid neutralization, sorption of base cations to cation exchange sites, outgassing due to re-equilibration of the carbonate system in rivers and oceans and any other relevant processes. These are considered permanent losses of base cations and are corresponding to reduction in CO_2 removals. Details for each of these carbon losses are outlined in section 7.4.1.2 of the Isometric protocol, as part of the soil-based quantification approach.

$$
\begin{aligned}
CO_{2^e Stored,RP} = {} & CO_{2^e weathering,RP} - CO_{2^e BiomassLoss,RP} \\
& - CO_{2^e NetNewCarbonate,RP} \left[-CO_{2^e NetNewSilicate,RP}\right] - CO_{2^e NetSorption,RP} \\
& - CO_{2^e NonCarbonicNeut,RP} - CO_{2^e Riverine\ Loss,RP} - CO_{2^e MarineLoss,RP}
\end{aligned} \tag{28}
$$

- The first term in this equation, $CO_{2^e weathering,RP}$, is to be calculated by measuring the loss of cations from the applied material (generally through comparing the abundance of the relevant metal to the abundance of an immobile tracer element) and multiplying this by a factor for “the pH dependent conversion of alkalinity to carbon stored in the aqueous phase”.
- $CO_{2^e BiomassLoss,RP}$ reflects the reduction in delivered CO_2 removal due to uptake of base cations by plants in the treated area, measured by direct, representative sampling of the treatment and the control plots.
- $CO_{2^e NetNewCarbonate,RP}$ is the average net change in soil inorganic carbon (SIC) measured via either calcimetry or thermo-gravimetric analysis (i.e. carbonate precipitation, formation of new carbonate minerals). The value may be positive (increase in SIC and less carbon stored), zero or negative (dissolution of carbonate materials i.e. more carbon stored, if reaction is with carbonic acid). The development of soil inorganic carbon as carbonates is associated with a reduction in storage of carbon as bicarbonates in solution, and thus reduces the maximum potential carbon removal by the bound cations by 50%.
- $CO_{2^e NetNewSilicate,RP}$ is the estimated average net change in silicate minerals beyond the near-field zone (i.e. outside the domain of direct measurement). Isometric states the inclusion of this parameter in the equation is currently only for completeness, measurement is not required at this time, and we have therefore marked the term with square brackets in the equation.
- $CO_{2^e NetSorption,RP}$ is the net sorption of base cations to cation exchange sites (noting this value may be negative, indicating a net release of cations in previous reporting period).
- $CO_{2^e NonCarbonicNeut,RP}$ is the neutralisation of acids other than carbonic acid, to be determined by direct measurements of anions in porewaters (e.g. NO_3^-, PO_4^{3-}, Cl^-, SO_4^{2-}) or estimates based on documented fertiliser application.
- $CO_{2^e Riverine\ Loss,RP}$ is riverine losses: CO_2 removal expected to be released back to the atmosphere due to outgassing in river systems, typically estimated via models.
- $CO_{2^e MarineLoss,RP}$ is marine losses: CO_2 removal expected to be released back to the atmosphere due to outgassing in the ocean, assumed to be estimated via models.

ISO standards are referenced for correct measurement procedures where relevant for each parameter and any additional standards or methodologies are allowed but to be approved by Isometric first. Isometric allows for alternative soil-based quantification as long as they contain the above-mentioned terms and parameters as part of $CO_{2^e Stored,\ RP}$.

3.2.1.2 Aqueous phase approach, $CO_{2^e Stored,RP}$

Under the aqueous approach, carbon removal is assessed based on measurement of ions in porewater, or if the water catchment is ‘well defined’ of catchment waters. To be considered well defined the applicant must show that groundwater runoff occurs at known points and is isolated from other water sources, and that the alkalinity flux from the project is resolvable in the catchment water. If quantified using the aqueous-based approach, it is assumed that all major soil column processes are accounted for by assessment of the aqueous geochemistry of the sample water. In the aqueous phase approach $CO_2e_{Stored,\ RP}$ is calculated by measuring cation abundance either directly or implicitly by assessment of the carbonic acid system, and then deducting the expected CO_2 release back to the atmosphere due to outgassing in rivers and oceans from the carbon removals as determined from infiltration of carbonate alkalinity. It is noted the equation does not include base cations removed by plant uptake, however, alkalinity may still be taken up by plants below the observation window (e.g. plant roots extend below the 20cm depth of porewater sampling) and in those cases it has to be quantified.

$$CO_{2^e}{}_{Stored,RP} = CO_{2^e}{}_{Aqueous,RP} - CO_{2^e}{}_{RiverineLoss,RP} - CO_{2^e}{}_{MarineLoss,RP} \tag{29}$$

Where the terms $CO_{2^e}{}_{RiverineLoss,RP}$ and $CO_{2^e}{}_{MarineLoss,RP}$ are calculated as for the solid phase.

3.2.2 Indirect emissions and leakage

Leakage emissions outside the system boundary but occurring due to project activities are considered and must be attributed to the reporting period in which they occur. Where potential leakage is identified, it must be quantified and deducted from the CDR. Leakage emissions are associated with activity-shifting, market leakage, and ecological leakage. Isometric requires that potential sources of leakage emissions are identified. The minimum requirement is that feedstock replacement must be considered as part of the leakage assessment.

A framework is provided to identify feedstocks as 'waste inputs' with no embodied emissions if the material was obtained without payment or if the quantity of material sold for ERW was not already being used and the payment was not a 'significant share' of revenue for the material producer.

3.2.3 Additionality and baselining

Projects must demonstrate additionality by showing that the carbon removal would not have occurred without the project intervention. The Isometric general standard (Isometric, 2025c) introduces four pillars of additionality: financial, common practice, environmental and regulatory.

- Financial additionality can be demonstrated if either a) removals are the only source of revenue for the project, or b) that without carbon finance revenue the project has an IRR that is zero or lower or that is below the cost of capital or required return on equity for the project, and that the revenue from carbon credits will make that IRR positive or above the required rate of return (as appropriate), although there is provision made for project proponents to justify a higher IRR for the assessment. The standard is not prescriptive about what target IRR can be considered acceptable.
- According to the common practice analysis, projects below TRL 8 or 9 are considered additional without further analysis. Alternatively, a full analysis must justify that similar activities are not common practice in the project's geographic area.
- Regulatory additionality requires that the project is not legally required, though removals beyond the minimum legal requirement may be certified.
- Environmental additionality is defined as a net negative climate impact, which is presumably trivial for any carbon removal project generating credits, and therefore does not seem to be a substantive addition to the additionality framework.

Additionality is determined at the time of initial verification and is to be reviewed if there are significant changes to project operations, new regulatory requirements or changes to project finance indicating carbon finance is no longer needed. If the project becomes not additional, it will be ineligible for future credits. Carbon credits issued under current or past crediting period will not be affected.

The baseline scenario is a quantification of what would have happened in the absence of the enhanced rock weathering activity. It represents CDR from natural weathering or pre-existing land practices. The baseline scenario is project specific and is determined by the control plots and aims to account for liming agricultural practices that could influence the net impact of the ERW project. The baseline scenario includes the calculation of the counterfactual carbon removals and they are subtracted from the total net amount of CDR in the main quantification formula. Isometric highlights that some projects may have to consider removals from the weathering of feedstock that could be expected in the counterfactual scenario where the feedstock was not applied to fields.

3.2.4 Long-term storage and liability

The standard states that the durability of carbon storage in ocean DIC is considered to be at least 10,000 years, with DIC constituted of bicarbonate (90%) and carbonate (9%). The residence time of dissolved inorganic carbon in the ocean is between 10,000 to 100,000 years.

ERW projects are categorised as having a very low risk level of reversal; however, the potential impacts and reversals in the global ocean dissolved inorganic carbon reservoir are not directly observable and it is acknowledged that more research is needed. The methodology notes that reversals could occur from the ocean if anthropogenic alkalinity management leads to increased carbonate precipitation, but that there is no way to monitor reversals in the DIC reservoir that would allow attribution to individual projects. A 'very low' reversal risk means that 2% of removals would need to be placed into buffer pools. It is not entirely explicit under what circumstances a reversal would be identified by Isometric – we presume that this would not be possible at the individual project level but could occur if an improved understanding of the DIC pool led to a need to apply a downward adjustment to estimates of carbon removed by previous projects[59].

3.2.5 Sustainability

Projects must consider environmental and social impacts at all locations, including quarrying, deployment sites, and feedstock transportation. They must implement measures to identify and eliminate potential risks to ecosystems and biodiversity, develop tailored mitigation plans, and ensure comprehensive safeguards for environmental protection, social equity, community engagement, and respect for cultural values. Regular assessment of environmental impacts, including heavy metal concentrations in crops and groundwater, is required, with data shared publicly. Adaptive management strategies must be employed based on monitoring results and feedback. Heavy metal concentrations must not exceed local regulatory limits. In the absence of local regulations, projects will adhere to EU, WHO, or US EPA standards.

Project developers are required to select feedstocks as per Isometric's Rock and Mineral Feedstock Characterisation Module (Isometric, 2024). Projects with pre-existing elevated metal concentrations can still be considered for credits but they must provide evidence of undertaking remediation strategies to mitigate contamination – for instance, adjusting the type or amount of feedstock used, applying soil amendments, or using phytoremediation practices with plants that can absorb heavy metals. If the project further aggravates the soil contamination, then it will be deemed ineligible.

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[59] Isometric indicate that the buffer pool is also intended to cover credits that have to be negated due to abandonment of the project.

3.2.6 MRV

Isometric's MRV approach includes pre-deployment, deployment, and post-deployment monitoring requirements.

Pre-deployment stage includes baseline establishment, field management, and climatic monitoring. It is required that soil samples are collected to quantify heterogeneity in soil characteristics such as pH, texture, permeability, cation exchange capacity (CEC), and base saturation. Field management prior to feedstock deployment takes account of crop type, productivity levels, tillage practice, and fertiliser use. Climatic monitoring provides data on rainfall, temperature, humidity, wind speed, and solar radiation.

The monitoring approach that projects can choose from can be 2-plot or 3-plot. The 2-plot approach includes control and treatment plots, while the 3-plot approach includes control, treatment, and deployment plots. The control plot represents CO_2 removal under business-as-usual practices, including liming and it is used to observe counterfactual CDR. The treatment plot is the monitored area where material is applied and the deployment area is the larger area, which is monitored less intensively. Isometric also includes a requirement for a research plot for projects exceeding a total 1,000 hectares, where alkalinity measurements are measured at a recommended depth of 1 meter.

The maximum CDR potential from feedstock application (Section 9.3.4.4.1) i.e. feedstock-based alkalinity during deployment is required to be measured by multiplying the average application rate by the moles of alkalinity per kilogram of rock. The theoretical maximum CDR potential is also calculated via an adjusted version of the Steinour equation (Section 9.3.4.4.3).

Along with the two quantification approaches based on soil and porewater measurements, Isometric outlines a list of validation approaches which aim to validate the selected quantification approach. At least one of the proposed validation approaches must be conducted and within the control and treatment plots.

The validation approach must not be the same as the quantification approach. Isometric proposes the following validation approaches:

- Local soil-based validation;
- Local porewater-based validation;
- Local ion exchange resin validation.

Measurements and chemical analysis are required for both control and treatment plots (each representing at least 2.5% of the project area). Direct measurements and sampling are essential for the verification of carbon removals, with recommended depth for sampling of 20cm or the depth of tillage plus a buffer, also subject to site-specific characteristics. The number of samples required is not fixed, but it varies based on project.

Isometric protocol also requires **statistical significance** for the verification of carbon removals: a 99.7% confidence threshold is required to validate weathering measurements, and the two techniques used for **cross-validation** of weathering signals (i.e. soil measurements and aqueous measurements) must also agree to within a 99.7% confidence interval (Section 7.4.1.1).

Isometric requires characterisation of soil parameters in baseline and post-deployment samples; the full details are available in Section 9.3.4.6.7:

Required soil-based measurement parameters for all sampling events and/or baseline samples:

- Concentration of immobile tracers and mobile cations that will be used for weathering determinations
- Cation Exchange Capacity (CEC)

- Base cation saturation
- Soil Inorganic Carbon (SIC)
- Soil pH
- Soil Texture
- Soil Organic Carbon (SOC)
- Total Sulphur Content
- Soil bulk density

Recommended soil-based measurement parameters for all sampling events:

- Soil Moisture
- Soil Permeability
- Total Carbon Content
- Secondary clay formation
- Soil CO_2 flux
- Soil texture (recommended for subsequent sapling events)
- Soil bulk density (recommended for subsequent sapling events)

Optional for all sampling events:

- Carbon isotopes

Similarly, the protocol requires porewater sampling plan which should consider events which impact the soil moisture such as irrigation and heavy rainfall. Baseline data is required e.g. water table depth, hydrologic maps, watershed boundaries, etc. Recommended and optional parameters for fluid samples include (with full details available in Section 9.3.5.1.1 of the methodology):

Required measurement parameters:

- pH
- Temperature
- Total alkalinity
- Concentration of any tracers and mobile cations that will be used for weathering determinations
- Non-carbonic acid acidity (anions)
- Dissolved inorganic carbon (DIC)

Recommended measurement parameters for porewater samples:

- Electrical conductivity
- pCO_2

Optional for porewater samples:

- Stable isotopes

An independent third-party verification includes checking feedstock characterisation, quantification approach, monitoring plan and environmental and social safeguards. Validators also conduct site visits at least once every two years at each control, treatment, and deployment plot.

3.3 Rainbow (formerly Riverse)

Rainbow (formerly Riverse[60]) published their enhanced rock weathering methodology in April 2025, and it is currently undergoing public consultation (Rainbow, 2025a). Riverse methodology is informed by Cascade Climate's report "Foundations for Carbon Dioxide Quantification in Enhanced Rock Weathering Deployments", also referenced by Isometric in their methodology. It complies with ISO 14064-2: 2019.

Rainbow defines the project area as all sites where rock spreading and monitoring occur. Project scope includes all spreading activities within eligible sites during the project's duration, along with related upstream and downstream activities, such as greenhouse gas emissions from rock sourcing and transport.

Rainbow permits spreading activity on various soils, including agricultural fields, forests, prairies, golf courses, and parks. The methodology does not require a predefined maximum area, but sites must be within the same administrative, jurisdictional, and geographic area.

Rainbow outlines the process of issuing credits under the methodology in the following stages:

- **Pre-project sampling** includes pilot samples and measurements for site and feedstock characterization. A sampling plan should be proposed.
- **Project Validation** includes the submission of the necessary documentation to Rainbow such as a site characterisation report[61] and developing an optional stratification approach. The project then undergoes an ex-ante validation audit conducted by a Rainbow-accredited Validation and Verification Body (VVB). A technical peer review may also be required, the final decision of which sits with the Rainbow Certification Team. At this stage expected carbon removals are estimated.
- **Project operations and monitoring** includes developers spreading crushed rocks on eligible sites, taking samples, and following the monitoring plan. If monitoring shows no statistically significant CDR has occurred, there will be no credit issuance, and the process is repeated until results are sufficient for the next step of CDR verification.
- **CDR verification** includes supplying evidence that the "actual" ex-post carbon removals have occurred, result in issuance on the Rainbow registry. The statistical significance requirements for substantiating this process must be detailed.
- **End of project:** monitoring and verification continue until the end of the crediting period. Rainbow's maximum duration of the crediting period is 5 years.
- **Renewal of the crediting period** if removals are still occurring, the monitoring plan can be extended.

The near-field zone (NFZ) is set at 15 cm, or the tillage depth plus a 10 cm buffer, whichever is deeper. Deeper or shallower NFZs may be considered based on site hydrology or agronomic practices.

Rainbow states that the carbon removal initiation begins upon feedstock dissolution and is "complete and permanent" when alkalinity is either exported from the NFZ or remains in the soil porewater at the NFZ depth. Rainbow defines two methods for carbon removal verification and credit issuance and project developers must choose one of two methods: Method 1 Direct measurement of export (i.e. an aqueous-phase approach) and Method 2 Mass balance). (i.e.

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[60] The change of name happened in 2025.

[61] A site characterisation report must be submitted as an appendix to the Project Design Document (PDD) during project validation, before any rock spreading occurs. The report includes characteristics of the project site, NFZ soil and FFZ.

a solid-phase approach. Changes in soil organic carbon are currently not required for carbon credit issuance under the methodology.

Rainbow permits developers to spread feedstock multiple times during the crediting period. Total CDR is assessed for the entire project, not individual spreading events. Project developers must specify how they intend to manage samples, measurements, and calculations to monitor the CDR from various spreading events and the overlapping weathering signals.

3.3.1 Quantification

Rainbow ERW methodology requires project developers to present that the overall carbon footprint of the project is negative, meaning the project removes more CO_2 from the atmosphere than it emits. The net-negative project carbon footprint is based on preliminary Life Cycle Assessment (LCA) estimates of induced emissions and initial modelled CDR estimates. Processes contributing less than 1% of total induced emissions and removals may be excluded, until exclusions exceed 3% cumulatively.

At the ex-ante project validation stage, project developers must provide estimates for maximum potential CDR and potential CDR over project lifetime, both of which are based on the modified Steinour equation. The methodology allows the use of a model to estimate the expected potential carbon removals, the results of which are used to inform the allocation in the GHG quantification. The model includes, among other inputs, an estimate of carbon loss in the FFZ zones such as rivers and oceans.

Total net CDR is calculated for each reporting period as follows with all units in tCO_2e:

$$NET_{CDR} = Project_{CDR} - Baseline_{CDR} - Project_{emissions} \tag{30}$$

Where:

NET$_{CDR}$ represents the total net carbon removals in the reporting period and is the amount to be issued.

Project$_{CDR}$ represents the net CDR from the project scenario in the treatment plots in the near field zone (**NFZ$_{P,net\ removal}$**) and subtracting the carbon losses in the far field zone (**FFZ$_{P,loss}$**). Project developers must choose either Method 1: Direct measurement of export or Method 2: Mass balance for the quantification of the net carbon removals in the NFZ.

Baseline$_{CDR}$ refers to the net CDR from the baseline scenario. It is calculated by adding the net gain in carbon removal measured in the control plot NFZ (**NFZ$_{B,\ net\ removal}$**) and any removals from the BAU management of the feedstock used in the project (**Feedstock$_{B,\ removal}$**), then subtracting the carbon loss in the baseline FFZ (**FFZ$_{B\ loss}$**). There are two ways of quantifying the **NFZ$_{B,\ net\ removal}$**, by using the same methods of calculations as the ones in the treatment plot or use a negative control plot and assume that agricultural lime fully dissolves and generates CDR at 100% efficiency, with only losses resulting from non-carbonic weathering adjustments to be considered.

Project$_{emissions}$ represents the induced emissions caused by the project activities such as extraction, processing, and transport of feedstock in the project scenario. Emissions from transporting feedstock are calculated based on BICRS methodology. The Ecoinvent database version 3.10 is the main source of emission factors used.

Project emissions shall be allocated across the reporting periods. This allocation should ensure that all upstream emissions are accounted for within the first 50% of potential CDR. The distribution can be proportionate to the amount of CDR completed/credits issued in each reporting period, or it may be done upfront to account for induced emissions early on.

The methodology states that mining and extracting feedstock can be omitted if the feedstock is waste from other mining activities. Feedstock is considered waste if it has no economic value and would otherwise not be used.

3.3.1.1 Method 1: Direct measurement of export, NFZP,net removal

This method is implemented through aqueous-phase measurements in each reporting period. It measures a statistically significant signal for an increase in the export of weathering materials or their concentration in porewaters at NFZ depth or drainage or catchment waters beyond the NFZ between treatment and control plots (i.e. representative baseline). Both project (i.e. treatment plot) and baseline (i.e. control plot) NFZ carbon removals are calculated in tCO_2e following:

$$NFZ_{\text{P, net removal}} = \sum\nolimits_{time} CDR_{export,NFZ} \tag{31}$$

The method aims to measure the total measured alkalinity export $CDR_{\text{export,NFZ}}$ from the NFZ by measuring carbonate system parameters. The project developer must measure at least two, and ideally three, from the following:

- pH
- Total alkalinity
- Dissolved Inorganic Carbon (DIC)
- $[CO_2]$ (Dissolved CO_2 concentration)
- $[HCO_3^-]$ (Bicarbonate concentration)
- $[CO_3^{2-}]$ (Carbonate concentration)

Rainbow states that adjustments to direct measurements must be made in some cases: e.g. when total alkalinity is used, project developers must consider biotic contributions to acidity and alkalinity; if base cation concentration or total alkalinity are used, they must account for carbonic acid system speciation. In addition, project developers must differentiate weathering sources to identify if products come from silicate weathering or carbonate mineral dissolution. Net CDR calculations must be adjusted to exclude CDR from weathering of carbonate minerals.

3.3.1.2 Method 2: Mass balance, *NFZ_{P,net removal}*

This method is implemented through solid-phase based measurements. It measures a statistically significant signal for a decrease of base cations concentration at NFZ depth i.e. measuring the cation release from the feedstock dissolution, considering adjustments to net carbon removals due to changes in soil inorganic carbon stocks (potential carbon losses):

- Cation uptake in biomass
- Cation retention on sorption sites
- Secondary carbonate formation
- Alkalinity that is not charge balanced by bicarbonate

Method 2 is calculated following three steps: 1) measure the release of base cations from feedstock dissolution in the reporting period, 2) use the cation release measurements to calculate the potential CDR associated with the current reporting period, 3) adjust the calculated potential CDR by considering the above mentioned changes in the soil inorganic carbon stocks relative to the previous reporting period. Further carbon losses beyond the NFZ into the FFZ are applied under the $FFZ_{P\ loss}$ calculation.

Both project (i.e. treatment plot) and baseline (i.e. control plot) NFZ carbon removals are calculated in tCO_2e following the equation:

$$NFZ_{P,net\ removal} = CDR_{FD} - CDR_{biomass} - CDR_{Alk\ inefficiency} - CDR_{sorption} - CDR_{carbonate\ precip} - CDR_{silicate\ precip} \tag{32}$$

Where:

$\mathbf{CDR_{FD}}$ is the potential theoretical maximum CDR associated with the base cations release from feedstock dissolution during the reporting period.

$\mathbf{CDR_{biomass}}$ represents the decrease in CDR due to base cations taken up by plants.

$\mathbf{CDR_{Alk\ inefficiency}}$ is the decrease in CDR due to alkalinity not charge-balanced by bicarbonate. This is primarily due to pH-dependent carbonic acid system speciation, non-carbonic acid weathering from sulphuric, nitric, or organic acids, and acid buffering.

$\mathbf{CDR_{sorption}}$ is the temporary decrease in CDR due to sorption of base cations to cation exchange sites or reductions in exchangeable acidity (value can be positive or negative).

$\mathbf{CDR_{carbonate\ precip}}$ represents the change in CDR from the precipitation of secondary carbonate minerals. If a decrease, it is already captured by the feedstock dissolution at the depth of the NFZ under $\mathbf{CDR_{FD}}$. If there is an increase in removals, it can be evidenced via soil inorganic carbon measurements (note these measurements are excluded from Method 1).

$\mathbf{CDR_{silicate\ precip}}$ represents the decrease in CDR due to formation of secondary silicate minerals such as clays, amorphous silica, or Fe/Al oxyhydroxides. The methodology does not require a measurement because it is assumed to be covered in the calculation of the feedstock dissolution at the depth of the NFZ under $\mathbf{CDR_{FD}}$.

3.3.1.3 Carbon losses in the far field zone, FFZP,loss

$FFZ_{P,\ loss}$ represents the loss of carbon removal in the project FFZ and it is part of the wider calculation for quantifying carbon removals from the project scenario.

FFZ losses are anticipated to span a long period (e.g. 1000 years); however, for the purposes of credit issuance, Rainbow requires that these losses are estimated in advance based on the total potential CDR. They are then proportionally allocated across reporting periods according to the amount of carbon removals reported and credited.

$$NFZ_{P,net\ removal} = CDR_{groundwater} + CDR_{surface\ water} + CDR_{surface\ ocean} \tag{33}$$

Where:

$\mathbf{CDR_{groundwater}}$ represents the carbon loss due to permanent alkalinity sinks in groundwater or the lower vadose zone. Rainbow states modelling and monitoring limitations for this parameter therefore measurement is not required.

$\mathbf{CDR_{surface\ water}}$ represents the carbon loss due to permanent alkalinity sinks and CO_2 outgassing in rivers and lakes. This parameter accounts for outgassing from DIC system equilibration (calculated using CO_2 flux equations for water-air gas exchange) and carbonate mineral burial (using models). Project developers must justify qualitatively that the site's hydrology will not cause significant organic carbon destabilisation downstream. This term may be excluded if the project developer demonstrates that the weathering material will bypass surface waters and instead flow directly to groundwater and oceans.

$\mathbf{CDR}_{\text{surface ocean}}$ represents the carbon loss due to permanent alkalinity sinks and carbonic acid system re-equilibration in the ocean. It is calculated using models or thermodynamic storage efficiency calculations.

Rainbow requires that project developers describe the expected final durable carbon removals reservoir, stating if carbon is stored as bicarbonate ions in the ocean, in groundwater, or in carbonate minerals in the watershed. If significant carbonate minerals are present, project developers must assess the risk of acid weathering; high risks must be deducted from the CDR using a Rainbow risk assessment template.

Rainbow methodology uses models in several key areas including feedstock dissolution for ex-ante calculations for provisional credit volumes and expected timeline of weathering, hydrological models to determine groundwater flow path and residence time and ocean basin determination, FFZ loss models for rivers and oceans. Rainbow states that using models for purposes beyond crediting, such as reactive transport models, is encouraged for scientific advancement and future MRV use but not required for carbon credit issuance under the methodology.

Rainbow requires uncertainty assessment at a project-level and at a methodology-level. The outcome of the assessment determines a discount factor (i.e. a portion (low uncertainty 3%, medium uncertainty 6%, high uncertainty 9% or higher) of verified carbon credits are withheld to guard against uncertainties in GHG reduction and overestimated carbon removals/avoidance). Examples of areas where uncertainty is assessed include assumptions, baseline scenario selection guidance and equations and models. The methodology categorises assumptions with low and high uncertainty. For example, a low-uncertainty assumption is that “organic acids are assumed to degrade after reacting with silicate minerals, producing DIC equivalent to that from carbonic acid weathering”. An assumption with high uncertainty is that there are no net CDR gains in the FFZ, which they suggest is a conservative assumption that would tend to lead to under-crediting. The minimum uncertainty at the methodology level is described as low, leading to an expected discount factor of at least 3%. However, project-specific conditions may increase uncertainty and warrant higher discount factors.

Uncertainty of models and measurements used to calculate carbon removals is required to be assessed for each project. The methodology states “it is recommended that the values selected for CDR quantification represent either the lower bound of a two-sided 80% confidence interval (for frequentist approach), or the 10th percentile of a posterior distribution (for Bayesian models).”.

3.3.2 Indirect emissions and leakage

The project must avoid indirectly transferring GHG emissions. Project Developers need to transparently evaluate leakage risks from activity shifting and upstream and downstream emissions in the project design document. It is suggested that most relevant emissions are usually already included in the LCA-based quantification approach. Any significant sources of leakage that cannot be mitigated must be incorporated into the greenhouse gas reduction calculations or the discount factor. The methodology invites project developers to consider activity shifting risks such as:

- Reallocation of rare earth elements and critical minerals from other uses, such as renewable energy supply chains.
- Possible alternative uses of mineral waste that may result in lower emissions.

- Land use changes resulting from project infrastructure affecting previously undisturbed or high-value land.
- Evaluate the effect of spreading feedstock on crop yields. If significant reductions in yield are observed compared to the baseline, replacement emissions must be accounted for in the project's deductions for this impact.

3.3.3 Additionality and baselining

Project developers are required to demonstrate additionality by performing a regulatory surplus analysis as well as investment or barrier analysis. Rainbow directs to an additionality template, created for further guidance (Riverse, 2024).

- Regulatory surplus analysis requires that project developers must prove there is no law or regulation mandating the project. This includes existing regulations and those confirmed to take effect within 5 years. If regulations partially require the project, only activities exceeding these requirements (the regulatory surplus) are eligible for carbon credits.
- Investment analysis is required to show that carbon finance revenue is essential for making the project financially viable. The carbon revenue must be used to fund project expansion or startup, especially for projects seeking pre-financing through provisional credit sales agreements. Business plans are required for initial investment analysis. Audited financial statements will be used to verify that the estimates in the business plan were reasonable and that carbon finance was utilized as planned. For new sites, additionality is demonstrated if the business plan indicates that carbon finance accounts for at least 80% of the company's revenue. For investments in expansion, only the additional carbon reductions enabled by the expansion will be eligible for carbon credits. The baseline scenario in this context would be the project prior to the expansion.
- Barrier analysis can be employed to demonstrate that the project encounters financial, institutional, or technological obstacles to its continued operations, which can only be addressed through carbon financing. For any barrier analysis, audited financial statements must be provided. These documents should prove financial barriers or show that the project cannot fund solutions to overcome institutional or technological barriers.

A site characterisation report is submitted at a project validation stage, explaining why the project area is suitable for ERW under baseline conditions. The baseline scenario describes the conditions or practices that would exist without the ERW project. It includes only the removals, excluding any induced emissions, to ensure that the project is credited solely for the additional removals it achieves beyond those that would have occurred under typical circumstances. Business-as-usual practices include the use of agricultural lime, cropping and tillage, fertiliser application, irrigation, and weathering in waste feedstock piles. Spreading agricultural lime to fields can lead to increase, decrease on no change in carbon removals. Carbon credits are only awarded for removals achieved beyond the baseline CDR gains. The baseline scenario is determined by either the NFZ in a BAU control plot, quantified using the same quantification Method 1 and Method 2 as in the treatment plots., or by NFZ in a negative control plot when BAU plots are not present. If a negative control plot is used rather than a BAU control plot, it is assumed for calculation purposes that all agricultural lime dissolves and results in carbon removals with 100% efficiency, without accounting for carbon loss terms, apart from losses resulting from non-carbonic weathering adjustment. If the use of agricultural lime in the baseline scenario is determined to be a net source of emissions, its value should be considered 0 tCO_2e, to ensure that avoided emissions from liming are not counted towards project carbon removals. Baseline GHG emissions are calculated following a methodological approach based on a lifecycle assessment in line with ISO 14064-2:2019.

The baseline calculation also accounts for any removals from the BAU management of the feedstock and any carbon losses in the FFZ (calculated in the same way as in the treatment plots).

If the feedstock is waste, the methodology requires project developers to describe its alternative use. This helps in establishing the baseline scenario to decide whether to include baseline carbon removals from feedstock storage. Project developers are required to model carbon removals from baseline weathering for the surface layer of the feedstock considering the environmental conditions of where the feedstock is stored such as rainfall and temperature. If the model results are <1% of emissions, they can be omitted.

The baseline scenario's overall structure for a project is valid throughout the crediting period. It can change if the project developer reports a significant change in operations or baseline conditions, or if the methodology is revised. The actual values in the baseline scenario are updated every reporting period.

3.3.4 Long-term storage and liability

ERW projects under this methodology have a stated permanence of over 1000 years. This is based on geochemistry concepts that suggest that dissolved inorganic carbon (DIC) remains stable in oceans and waterways for thousands of years, eventually forming stable carbonate minerals after 10,000 to 100,000 years. There are various parameters measured in the site characterisation and monitoring section that are used to understand the anticipated pathways of weathering products and their permanence, including the expected final reservoir of DIC. The methodology required project developers to list carbon release risk scenarios for precipitated and dissolved carbon and discuss how they are considered in the project's carbon accounting.

Rainbow also requires from project developers to fill in a methodology risk evaluation template to assess and rank carbon storage reversal risks based on social, economic, natural and delivery risks. Reversal risks that have been assigned a high or very high risk score must be accompanied by a risk mitigation plan. This plan, developed by the project developer, should outline the long-term strategy and investments for preventing, monitoring, reporting, and compensating for carbon reversal. Additionally, each high or very high reversal risk requires extra buffer pool contributions at a rate of 3% of verified removal Rainbow Carbon Credits[62].

Regarding double counting, project developers must demonstrate that the landowner distributing the rock will not issue carbon credits. This is verified through signed contracts.

3.3.5 Sustainability

Eligible feedstocks must consist of silicate rocks containing alkaline materials, including olivine, pyroxene, plagioclase feldspar, basalt, dunite, and peridotite. Carbonate feedstocks are not permitted at this time. The quantity of feedstock used must adhere to jurisdictional regulations regarding nutrient and metal additions to soils, frequency and amounts of rock spreading.

Rainbow ERW methodology requires project developers to provide a detailed qualitative description of the feedstock for each source, including GPS coordinates, name of the quarry, names of the companies involved in extracting the feedstock and any additional processing steps undertaken, general mineralogic composition and heterogeneity.

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[62] The buffer pool acts as a common insurance for all certified projects. Buffer credits may be withdrawn to compensate a project that registers a reversal, or a change in regulation that takes the project out of the voluntary market (Rainbow, 2025b).

Additionally, the methodology requires compliance with Environmental and Social Do No Harm (ESDNH) requirements which entail measuring additional characteristics of feedstock before spreading the rocks[63], including:

- Chemical composition: concentration of elements like antimony (Sb), arsenic (As), barium (Ba), cadmium (Cd), lead (Pb), and others;
- Mineralogical composition: concentration of asbestos, asbestiform minerals, and respirable crystalline silica (RCS);
- Radioactivity: concentration of relevant radionuclides such as Uranium (U-238 series), Thorium (Th-232 series), and others. Proof of adherence to regulations such as BSSD (2013/59/Euratom) in the EU or OSHA 1910 Subpart Z in the US.
- Particle size distribution: concentration of particles with a diameter of 2.5 µm or less.

In accordance with ESDNH requirements, project developers are required to complete a risk evaluation template to assess and prioritise the risks associated with ERW activities. This should be followed by the development of a risk mitigation and compensation plan. The identified risks include:

- Loss of soil organic carbon (SOC) over time.
- Potential asbestos contamination in mineral amendments, and exposure during mining, processing, transport, and feedstock application.
- Air quality impacts from silicate application and dust exposure.
- Health risks to workers and communities due to airborne particulates.
- Hazardous mineral waste generation along the supply chain.
- Heavy metal accumulation in soils and water systems.
- Dissolution of heavy metals, leading to environmental contamination.
- Altered soil chemistry affecting crop yield and agrochemical use.
- Altered soil physical properties affecting e.g. water drainage.
- Water contamination from silicate application, impacting drinking water.
- Harm to soil, water, air quality, biodiversity, and organisms from feedstock use.
- Exceeding national and local pollutant thresholds.
- Presence of radioactive materials or hazardous elements in feedstock.
- Fossil energy use for mining, grinding, and transport of rocks and minerals.

Project developers must demonstrate that their project delivers a minimum of 2 co-benefits from the UN Sustainable Development Goals (SDGs) framework, with a maximum of four. Rainbow lists typical co-benefits for ERW methodology projects but allows developers to add others.

3.3.6 MRV

The methodology is specific about feedstock measurements of relevant characteristics which are used to estimate and track carbon removal potential, following a “Feedstock Sampling Protocol”. The details of the sampling protocol are supplied by the project developer i.e. sampling frequency, representative sampling, number of subsamples, total samples material,

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[63] Additional analyses and monitoring after spreading the rocks is essential to monitor potential environmental risks.

sample collection, and handling methods. Project developers must measure characteristics of feedstock on a pre-agreed composite representative sample once for each feedstock batch, including chemical[64] and mineralogical composition, particle size distribution[65] and morphology, and moisture content. The latter must be measured immediately prior to each spreading event. The methodology assumes that all feedstock within a single batch exhibits consistent characteristics, provided it is stored in covered areas shielded from moisture. When feedstock is exposed to moisture, it may undergo weathering, resulting in altered characteristics. Consequently, the uncovered feedstock is classified as a new batch.

3.3.6.1 Ex-ante and ex-post sampling plan

Project developers must include an ex-ante sampling plan for GHG measurements in the site characterisation report during the project validation stage before any rock is spread. Later after the rock is spread and monitoring has started, project developers are required to document their ex-post sampling approach in the monitoring report. Ongoing monitoring is mandatory throughout the project's crediting period – starting before the spreading of ERW feedstock and extending to a maximum of five years after[66]. It must occur at least once per reporting period, with an annual or more frequent schedule based on site and project design.

Rainbow monitoring requirements differ based on spreading events and the type of the feedstock batch:

- Monitoring requirements for all reporting periods

When there are no subsequent spreading events, and all information related to spreading has been reported and audited in previous reporting periods, project developers are responsible for monitoring the ongoing CDR, ESDNH risks, and any optional co-benefits associated with the continued weathering of the previously spread feedstock. The measured and monitored parameters will vary based on whether Method 1 or Method 2 was used for quantification.

- Monitoring requirements for successive spreading event on the same site, same feedstock batch

These requirements are applicable if the project undertakes new spreading activities on the same sites where feedstock has been previously distributed, using the same batch of feedstock (i.e., if the feedstock was stored and covered at the project operations site between spreading events).

The monitoring scope will be broadened to include the application rate of feedstock, its moisture content, revised maximum potential CDR, and modelled CDR estimates for the new spreading. Additionally, it will encompass updated induced emissions, revised FFZ loss estimates, and refined total project CDR estimates.

- Monitoring requirements for successive spreading event with new site

These monitoring requirements apply when the project spreads feedstock on new sites not listed in the site characterisation report, using the same batch of feedstock stored at the project operations site between spreading events. These requirements include the initial monitoring requirements under the first two bullet points and additional ones for the new site, such as a site characterisation report and extrapolation strata evidence if needed.

- Monitoring requirements for successive spreading event with new feedstock batches

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[64] Concentration of elements such as calcium (Ca), magnesium (Mg), potassium (K), sodium (Na), sulphur (S), phosphorus (P), aluminium (Al), silicon (Si), and iron (Fe).

[65] Measure particles ranging from diameters greater than 2.0 mm to those less than 2.5 µm.

[66] If certifiable removals are continuing beyond this cut-off, the project operator can renew the crediting period.

These requirements pertain to the application of a new feedstock batch on the same sites. They include initial monitoring requirements outlined in the first two bullet points, along with additional requirements for the new feedstock batch such as a feedstock characterisation report, proof of adherence to the ESDNH requirements, and proof of compliance with leakage requirements.

3.4 Carbon Standard International (CSI)

In October 2022, CSI standards released version 0.9 of their “Guidelines for the Certification of Carbon Sinks created by Enhanced Rock Weathering in Croplands” (Ithaka Institute, 2022). This certification methodology was developed by the Ithaka Institute for Carbon Strategies. The methodology is tailored specifically for agricultural lands and does not apply to forests, nature conservation areas, or coastal regions.

The methodology introduces definitions such as C-sink curve and C-sink portfolio:

- A C-sink curve is defined as a function that represents the amount of carbon stored in a carbon sink, expressed in tons of carbon or tons of CO_2e, at any given point in time. Carbon sinks are described as short-term (<100 years) or long-term (>100years).
- A C-sink portfolio integrates the C-sink curves of diverse C-sinks with varying time horizons, locations, and negative emission technologies. For instance, it may combine C-sink curves from biochar and ERW. The C-sink portfolio serves as an evaluation and trading platform that facilitates the assessment and optimisation of both increasing and decreasing, long-term and short-term carbon sinks. CSI states the purpose of the portfolio is to achieve carbon removal of consistent quality in order to offset CO_2 emissions effectively.

The certification methodology specifies that for enhanced rock weathering, a C-sink curve can only be estimated, therefore an expected C-sink curve is introduced for preliminary use, which must be validated through scientifically proven models and/or in-field measurements, if used for certification. The methodology emphasises that it calculates carbon sink potential in a "conservative" manner via proxies, assumptions, and estimates.

The CSI standards do not list admissible rock types but set mineralogy thresholds and elemental composition requirements to qualify under the guidelines. The solubility of a mineral is influenced by the degree of silica polymerization. Minerals with low silica polymerisation such as olivine dissolve faster than those with high silica polymerisation, such as quartz. A rock must have at least 50% highly soluble silicate minerals, such as olivine, plagioclase, pyroxene, and k-feldspar, to be admissible under the CSI guidelines. The methodology requires that the quartz and carbonate concentrations are identified as well. If the quartz or carbonate concentration exceeds 2%, it must be accounted for by deducting the quartz/carbonate minerals from the initial mass balance and elemental composition.

The CSI methodology permits the usage of rock powder with a grain size ranging from 0 to 2000 µm, including an additional tolerance margin of plus 5 wt% (weight percent), for certification purposes. The grain size distribution of each production batch must be determined, as well as the mean moisture content of the rock powder at the factory gate. If moisture content is not provided, a default 10% will be assumed.

The methodology necessitates the application of rock materials to a minimum depth of 5-10 cm, with a recommended depth of 15-30 cm. The methodology can apply to perennial agricultural systems if the weathering material is incorporated regularly.

Regarding maximum application rates, the methodology advises maintaining a nutrient load below 30 kg P_2O_5 per hectare annually. Additionally, the maximum limit is established at 50% of the field-specific P_2O_5 requirement for the upcoming three years (to be calculated before application). The guidelines also recommend applying rock powder with alkaline compounds not exceeding the equivalent of 100% of the liming requirement for the next three years.

The methodology outlines the environmental factors impacting rock weathering rates such as soil temperature, soil moisture, soil pH, and soil CO_2 concentration. Soil temperature and soil moisture can be measured or modelled. Both should be obtained at the same soil depth of 10-30 cm, with a maximum depth difference of 10 cm. The methodology acknowledges that ERW weathering rates are influenced by plants and processes in the rhizosphere such as activities of microbes, earthworms, the production or organic acids and others – factored in the model described in the next section 3.4.1.

3.4.1 Quantification

The methodology outlines the calculation of the theoretical maximum amount of CDR capacity i.e. CDR_{max}, which depends on the metal cation flux releases from the weathering process. It is calculated by multiplying the absolute metal content of the rock with the cations' specific valence, present in ionic form in the solution. A safety margin of 10% is deducted from CDR_{max} to account for secondary mineral precipitation.

The CSI methodology offers a model to calculate weathering rates but notes the CDR potential may be "underestimated" due to assumptions and safety margins. Based on laboratory experiments of silicate rock dissolution under non-equilibrium conditions, the methodology highlights that applying this model in real field conditions involves assumptions, limitations, and uncertainty. CSI states that the method has not been validated yet and that it requires further optimisation to capture the intricate dynamics of enhanced weathering of multi-mineral rocks within a dynamic cropland field.

The model is a silicon normalised weathering model instead of a multi-mineral one. Minerals in rocks suitable for ERW contain silicon. When these minerals dissolve, silicon is released into the solution or forms secondary clay particles. This dissolved silicon can indicate the multi-mineral weathering rate. By standardising silicate rock composition to a silicon-normalised formula, the amount of rock weathered can be estimated from the silicon release, regardless of its fate. The model is based on assumptions that span a 100-year time horizon.

Empirical data from 26 silicate rock dissolution experiments was used to construct a generic silicon release model. These experiments used crystalline Icelandic basalt, which qualifies as a model silicate rock due to its elemental and mineral composition. Steady state dissolution experiments were performed in a mixed-flow reactor under conditions without water limitations, spanning a temperature range of 5°C to 75°C and a pH range of 2 to 11. The surface area normalised silicon release rate was calculated as a robust mean from multiple measurements across the 26 experiments. Experimental data from acidic to neutral conditions were analysed using non-linear regression to create a model estimating silicon release rate (rSi) in mol Si $cm^{-2}s^{-1}$ based on temperature and pH.

The base silicon release rate is adjusted for environmental factors affecting the weathering process, such as CO_2 concentration in the rhizosphere, biogenic weathering agents like microbes and earthworms, and months with sufficient soil moisture. This adjusted silicon release rate is then used in a modified shrinking particle model (SPM) model (Hangx & Spiers, 2009; Rinder & von Hagke, 2021). The base silicon release rate is adjusted for environmental factors affecting the weathering process, such as CO_2 concentration in the rhizosphere, biogenic weathering agents like microbes and earthworms, and months with sufficient soil moisture. This adjusted silicon release rate is then used in a modified shrinking particle model (SPM) model (Hangx & Spiers, 2009; Rinder & von Hagke, 2021).

The SPM model translates the mass loss from a rock particle due to weathering into a proportional decrease in reactive surface area. This calculation is made using the known material density and assuming the particles are spherical. A surface roughness factor is incorporated to consider fractures, elevations, and porosities of rock particles. This factor is multiplied by the silicon release rate (rSi), which is then utilised in the SPM to determine the fraction of rock grain weathered over time, measured in years since application. This is calculated for each particle size class and converted into mass of dissolved rock powder (rock powder mass weathered (t)).

The final calculation of the SPM model involves determining the gross rock C-sink. This is done by dividing the mass of weathered rock powder by the initial rock mass and then multiplying by CRD_{max}:

$$Gross\ Rock\ C\ Sink\ (t) = CDR_{max} \times \frac{Rock\ powder\ mass\ weathered\ (t)}{Initial\ rock\ mass} \tag{34}$$

The rock C-sink potential refers to the calculated carbon sink potential resulting from ERW, representing the amount of carbon removals in tCO_2e:

$$Rock\ C\ Sink\ Potential = CDR_{max} * m_{rock} * sf_{soil} * sf_{aquatic} * sf_{marine} \tag{35}$$

Where:

CDR_{max} is the theoretical carbon removals capacity, including a safety factor for metals retained in clay minerals.

m_{rock} is the mass of rock material applied per field in tonnes dry weight.

sf_{soil} is a safety factor of 0.9 to cover calcium carbonate precipitation.

$sf_{aquatic}$ is a safety factor of 1.

sf_{marine} is a safety factor of 0.86, based on Lewis et al. (2021).

Safety factors are used to account for potential losses from paedogenic carbonate precipitation and ocean losses. The methodology states that significant losses from groundwater and riverine systems are not anticipated. CSI notes that if suitable process-based hydrological models become available, the safety factors can be lowered or replaced.

The net C-sink potential is the carbon removals achieved after accounting for potential carbon losses and emissions from the project value chain, also referred to as “carbon expenditures”.

Direct and indirect greenhouse gas emissions must be quantified and compensated to ensure net carbon removals are achieved. Only after the compensation of carbon expenditures and their retirement in an equivalent carbon sink is completed, the Rock C-sink as per this methodology can be entered in a carbon registry and is admitted for sale as a climate service.

The methodology recognises two types of sources for ERW feedstocks which defines the calculation of the project emissions:

- **ERW rock material from mine tailings: a by-product of rock mining operations**, which occurs in the baseline scenario as business as usual. For a transition period of three years from publication of the CSI methodology[67], emissions from mine tailings covered only material transport and emissions from application of material to the project field. After the transition period, an additional share of the company's scope 1-3 emissions is attributed to mine tailings based on the revenue generated from rock powder sales.

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[67] Approval date is 06/04/2023 and publication date is 31/10/2022.

- **ERW rock material exclusively produced for enhanced rock weathering project activity**. Emissions from mining and milling operations, material transport, and application to the field need to be fully covered and quantified. Emissions from material production are verified by the certification body q.inspecta AG, while downstream emissions from transportation and application operations must be quantified through a tracking service accredited by CSI.

The total project emissions (i.e. carbon expenditures) are the sum of production, transport, and application emissions of weathering material. Carbon expenditures are expressed in tCO_2e per tonne of rock powder:

$$Carbon\ expenditures = Ex_{production} + Ex_{transport} + Ex_{application} \qquad (33)$$

CSI methodology requires that emission factors used in the calculation must be revised annually (e.g. the calculation of $Ex_{production}$ which includes emissions from mining and milling and a safety margin of 10% for scope 3 emissions) or after any changes in mining and milling technologies.

The methodology briefly outlines alternative approaches for calculating carbon removals via enhanced rock weathering. These approaches include process-based models such as SCEPTER, PhreeqC, and the Rock Chip Project, in addition to direct in-field measurements from liquid, solid, and gas phases. It is noted that the current CSI model may be revised or replaced should validated methodologies and assessment methods become available.

Due to inherent uncertainties involved in the modelling of the weathering process under this standard, CSI states that the carbon sink potential must not be traded within the framework of CO_2 compensation schemes[68], instead it is offered as a climate service. Under the CSI methodology, a climate service entails the creation of a carbon sink within a timeframe that is relevant for mitigating anthropogenic climate change. The methodology calculates the CDR and carbon sink potential; however, the exact size of the carbon sink at any given moment between the application and year 100 remains insufficiently precise. The latter is what the methodology requires as an essential part for creating carbon sink portfolios to compensate for CO_2 emissions. CSI highlights that the option of using Rock S-sink carbon sinks as a CO_2 emissions compensation is available only as part of a C-sink portfolio and when they are validated.

3.4.2 Indirect emissions and leakage

The methodology differentiates sourcing feedstocks from mine tailings as a by-product or weathering material produced specifically for ERW project activity. Emission attribution is based on direct and indirect GHG emissions following scope 1, 2, and 3 concepts per the GHG protocol. After a three-year transition period post-2022, emission attribution to mine tailings is equivalent to the share of annual revenue generated from the sale of the rock material. For example, if 5% of the company revenue is generated from the sale of rock powder, then 5% of the company's scope 1-3 emissions must be attributed to rock powder material production (no emissions are attributed in the first year).

The methodology does not include any details about leakage emissions in the context of activity-shifting, market leakage, and ecological leakage.

3.4.3 Additionality and baselining

The CSI methodology does not require the demonstration of additionality in comparison to other certification schemes for carbon removals via enhanced rock weathering. The

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[68] CO_2 compensations i.e. carbon offsetting can be used to achieve net zero emissions targets.

methodology allows the use of mine tailings, a by-product of rock mining, noting that unused mine tailings do not undergo significant weathering due to the chemical conditions within a pile of mine tailings. However, the methodology does not clearly provide a calculation of emissions and removals from a baseline scenario, also known as counterfactual analysis in other methodologies.

3.4.4 Long-term storage and liability

The methodology requires that the carbon sequestered be stored for an extended period, generally over 100 years, to ensure that the carbon removal is permanent. The methodology estimates that most of the rock minerals applied to lands with optimal characteristics will weather within 70-100 years. However, the exact timing of carbon removals is still uncertain according to the methodology. If the certified carbon sink portfolio is used for CO_2 emissions compensation as part of a compensation scheme, the carbon removals must be guaranteed for at least 100 years. Additionally, the SPM model employs assumptions around a 100-year time horizon.

3.4.5 Sustainability

The methodology specifies that the weathering material used for application in agricultural lands must comply with national and European regulations pertaining to fertilization and soil and water protection. The ERW material must be analysed to verify that nutrients and trace elements comply with safety standards and specified thresholds. The process requires examining the materials applied for ERW activity for nitrogen, phosphate, potassium, magnesium, sulphur, boron, copper, zinc, cobalt, alkaline components, selenium, and chlorine. Additionally, the analysis should include checking for trace elements such as arsenic, lead, cadmium, chromium, nickel, mercury, and thallium. An analysis for organic contaminants (e.g. PFT and PCB) is not required under the CSI guidelines.

The CSI methodology discusses the appropriateness of feedstocks in combination with lands that have a set of minimum characteristics, e.g. favourable agroclimatic conditions such as soil temperature and soil moisture, for effective enhanced rock weathering projects achieving permanent carbon removals. ERW projects outside the EU are required to apply to CSI to create a country-specific Annex document. This document will address ERW applications under national legislation and evaluate the quality of agroclimatic data.

To address future climate change, the methodology uses an extrapolation method grounded in historical climate data. The baseline year is determined by selecting the year with the lowest average national or sub-national precipitation during the decade preceding enhanced rock material application. This baseline year's climate data is then used to estimate future conditions.

The certification outlines the requirements for optimal characteristics of lands suitable for ERW as follows:

- Rock materials must comply with national and European fertilizer regulations and adhere to the specified precautionary values for inorganic soil contaminants.
- Maximum soil pH: the baseline soil must have a maximum pH of 7.3 to be permissible under the CSI guidelines.
- ERW projects are only permitted on land classified as agricultural land and it is not converted to land for urban use. In the event that a surface is sealed after the application of the weathering powder but before the estimated carbon sink potential is realised, this must be reported to the certification body. There must be a deduction from

the C-sink potential or compensation through the retirement of other C-sinks, proportional to the unrealised portion of the certified C-sink potential.

- Decadal aridity index (Ai) is the ratio of precipitation (potential soil water infiltration in mm) to evapotranspiration (potential soil water loss in mm) in a region, indicating that a net downward movement of soil water occurs. (Ai) is calculated using datasets covering the last 10 years prior to rock application. If (Ai)>1, the precipitation is larger than the potential evapotranspiration. To qualify under the CSI guidelines, the decadal aridity index of a location must be ≥ 0.75.

The methodology requires fire and dust protection to meet local and national regulations during the production, transportation, and application of ERW material. Workers must be informed in writing of potential risks and sign a document. Those involved in rock powder production and transport must wear appropriate personal protective equipment, including respiratory protection when operating material mills.

3.4.6 MRV

The CSI standard specifies that the expected C-sink curve, which represents the amount of carbon stored in a carbon sink at any given time, is "only indicative, but not certified" until the model is validated or replaced. It does not specify any monitoring requirements for in-field sampling intended for model validation after the application of the weathering material.

The mining company and rock powder product are certified by q.inspecta GmbH, approved by Carbon Standards International and accredited by the Swiss Accreditation Service SAS. The certification body is required to check relevant environmental protection, work safety, process emissions, and rock material sampling requirements and data. The methodology's pilot phase occurred in 2022-2023, followed by required on-site inspection and verification. Information regarding the on-site inspection is limited. According to current CSI guidelines, a C-sink trader, accredited by CSI, can also be responsible for monitoring downstream emissions from transportation and application operations via a tracking service, which is also accredited by CSI.

The methodology requires a routine analysis of the characteristics of the rock material batch every six months or after changes in mining and milling technologies. An accredited laboratory by Carbon Standards International conducts the routine analysis which includes particle size distribution, mineral composition, and presence of trace elements. Once each batch is certified, it is assigned a unique identification number that can be referenced for project-specific deliveries (delivery unit ID) at a later stage. The delivery note (with both IDs) includes a QR code that connects to a digital database containing details about the rock and field characterisation. Finally, an image of the delivery note and a geo-tagged image of the rock material are uploaded in an online registry system.

3.5 Verra

Verra notes that a methodology ‘idea note’ for a Methodology for Atmospheric Carbon Removal through use of Volcanic Basalt Soil Treatments (Verra, 2023) has been on hold since September 2023, and Verra does not provide a copy of the proposed methodology document on its website. Verra states that the methodology was proposed by ‘Carbon International’, but it is not clear to us whether this legal entity is still operational. Verra has announced that they will revisit the proposed methodology in Q3 2025.

3.6 Gold standard

The Gold Standard has identified enhanced weathering activities, alongside ocean-based carbon removal approaches, as being eligible in principle (Gold Standard, 2023), but to the best of our knowledge there is no active process to develop a methodology at this time.

4 Issues

This chapter highlights technical questions and challenges that may require further consideration in the context of the CRCF. These are grouped under the QU.A.L.ITY headings.

4.1 Quantification

It is possible to estimate the carbon removal associated with an ERW application, but a rational and cautious approach to the uncertainties of such estimates should be taken. Challenges remain in turning measurements into reliable indications of carbon removal in the field, quantifying downstream losses, and monitoring impacts on the organic carbon cycle. Moreover, if baseline carbon removals are not carefully accounted for it may lead to over-crediting. It is outside the scope of this report to produce a quantitative analysis or review of these uncertainties, but here we emphasise the key elements.

4.1.1 Measurement techniques

In-field measurements seek to quantify the amount of dissolved inorganic carbon that can be linked to an ERW project – for instance by tracking the concentration of metal cations that derive from dissolving ERW feedstock, or by tracking the concentration of carbon-containing ions in soil water. Different measurement techniques reveal different pieces of information but none is able to give the whole picture. Metal cations may be absorbed by the roots of growing plants, which limits their participation in the inorganic carbon system; DIC anions are formed in a broader geochemical system, and it may not be trivial to connect them to ERW feedstock application. Moreover, time lags in ions' movements between zones of the soil and the wider coupled system mean that different measurement approaches aiming to quantify the CO_2 removed from the atmosphere may deliver very different results (see Section 4.1.3).

At present, soil models may give useful insight when used in conjunction with sample measurements, but are not yet reliable enough to be used on their own. Certain types of soil sampling may be laborious and costly to perform regularly; all measurements may suffer from spatial variability, which raises the question of the sampling density needed to obtain meaningful results – particularly considering the diversity of project locations and conditions. Voluntary certification schemes allude to a threshold level of statistical significance (placing the onus on project developers to interpret and provide evidence), or to a minimum sampling density (possibly including flexibilities depending on the homogeneity of the land). Among these, and in the literature beyond, we have identified no implementable protocol that definitively constrains the sampling approach.

Both aqueous- and solid-phase measurement approaches to identify the rate of weathering are available and are considered viable in existing standards. The two approaches entail very different sampling practices, calculations, uncertainties, and levels of modelling effort; and, importantly, they measure fundamentally different things about what is going on in the soil. Certain project operators have been instrumental in driving the development of the techniques and in linking the results to underlying carbon removal processes and quantification. Nevertheless, considerable uncertainty remains in how to interpret the results into a unified picture; past attempts to do so have generally affirmed the complexity of the soil and biotic processes at play (and by implication, the need for ongoing scientific research).

Existing voluntary certification schemes / registries have established measurement protocols for both approaches, allowing applicants to choose. This recognises that there is as yet no clear consensus on which approach gives the most operationally accurate results. An CRCF methodology could similarly allow operators to choose an approach, potentially with a requirement to perform both kinds of test for validation and data gathering. This flexibility may be subject to review and additional guidance as knowledge and experience builds.

Existing voluntary schemes have set out sampling requirements for the two measurement frameworks, though the detail of the protocols to be used rests with the operator. This is informed by the recognition that over-prescriptive protocols risk failing to capture the complexity and diversity of ERW deployment conditions, and may be rendered obsolete as field technology and understanding evolves. In developing a CRCF methodology for ERW, it may be advisable for the Commission to follow this lead: i.e. to set broad principles and minimum requirements that engender uniformity and rigour, but defer to the certification bodies for more detail, and ultimately to the practitioners on the ground to develop and justify their favoured approach. Additional technical review of monitoring techniques/tools/devices, their technology readiness levels, accuracies, and sampling frequencies, and their complementarity with other types of measurements, is outside the scope of the current report. But it will be needed to inform any development of certification requirements now and in the future.

4.1.2 Non-carbonic-acid weathering

Existing certification methodologies treat weathering by acids other than carbonic acid as reducing the CDR potential of the ERW activity, and this framing is also widespread in the academic and grey literature (see Section 2.7.2. While this picture has its merits as a short-hand, it obscures the geochemical processes that actually deliver carbon draw-down, and risks misconstruing ERW carbon sequestration as working solely through the increase of DIC in the field where feedstock is applied. In reality, DIC may be produced in the field and/or anywhere downstream of it: under a ‘full system’ accounting perspective, neutralising any terrestrial acidity – whether carbonic or not – with an external source of alkalinity will eventually result in an increased net DIC concentration (though in some cases ‘eventually’ could be a relatively long timeframe, see Section 4.1.3 below).

Exclusion of non-carbonic weathering would systematically diminish crediting in acidified soils where there is the greatest agronomic need for it, and where weathering reactions progress fastest: in other words, it directs ERW projects to areas where ERW is less economically, agriculturally, and climatically attractive. There is an argument to be made that an inadequately thoughtful application of the principle of conservatism could unnecessarily undermine the deployment of the ERW activity (though of course alternative protocols would need careful justification and to be reactive to the developing scientific understanding).

An ERW certification methodology in the CRCF could follow the precedents in existing certification schemes, or could differ from them by not treating non-carbonic acid weathering as a loss term (though potentially treating it as a lag).

4.1.3 Carbon removal delays and credit timing

The timing of the physical carbon removal delivered by ERW is subject to a number of factors, and may be difficult to precisely identify. While the countdown to carbon removal begins at the point that feedstock is applied to the field, there can be a substantial delay between the spreading of rock powder and the draw-down of atmospheric CO_2: low and suppressed rates of mineral dissolution; sorption of ions by soil complexes; secondary clay formation; cation uptake by plants and other biota[69]; non-carbonic-acid weathering (discussed in the previous

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[69] Cf. Neumann et al. (2025).

sub-section); and other phenomena all have the potential to dilate the time between ERW feedstock application and carbon removal[70].

Some of these time-lags are better understood than others. Most existing certification methodologies issue certificates on the basis of in-field sampling, i.e. measuring either the progress of feedstock dissolution in the solid phase (which will include some of the material in a delayed pool), or the production of DIC in the aqueous phase (i.e. carbon that has entered the soil water once weathering products have presumptively exited the delay pools). Both of these approaches allow certificate issue to be delayed in the case that, for whichever reason, feedstock dissolution proceeds slowly (e.g. if applied in excessively large particles); and the methodologies typically specify the depth range at which soil should be sampled, which reduces variability in terms of measuring the weathering progress. But there are still lags, potentially on the order of decades, between feedstock dissolution and the final equilibration of DIC storage (Kanzaki et al., 2025). And since the solid-phase and aqueous-phase approaches measure different stages in the weathering process and must be linked using estimates and approximate models, they may yield very different results (te Pas et al., 2025; Vienne et al., 2025, preprint). For example, in soils that are predisposed to clay formation, weathering products may become quickly incorporated into soil particles, preventing them from neutralising acidity and hence preventing carbon sequestration from taking place.

It is worth pointing out that some phenomena which could be considered a delay to carbon removal are treated in certification methodologies as a carbon loss. As discussed in Section 4.1.2, some delays may be treated as losses because of the primacy of local in-field measurements rather than downstream modelling as a basis for estimating carbon removals; but the observation also applies to parameters that could in principle be measured locally. For example, plants that absorb ions may eventually re-release them back to the soil after they die (a delay), but this factor is sometimes treated as a negative term in the carbon balance (a loss). The delay-loss dichotomy could arguably be applied to non-carbonic-acid weathering too: a net carbon removal happens when the acid that would otherwise have flowed out of the field and caused CO_2 outgassing downstream doesn't do so due to its prior neutralisation by ERW feedstock. Precisely when this occurs depends on the time it would take acids to leach from the soil and into the watershed (a delay); but again, it is treated as a loss by certification methodologies.

Beyond the project boundary, it becomes infeasible for project operators to routinely measure lags from re-releases and re-absorptions of CO_2. This has implications for aqueous-phase measurements of DIC concentrations as well as solid-phase measurements. Though aqueous-phase DIC measurements can be said to capture the most direct evidence of carbon storage in surface water, it's important to recognise that a given unit of DIC flowing from the field into the watershed may exchange many times between the water and atmosphere until it can be considered durably sequestered in its final reservoir (the sea).

A certification methodology for ERW would have to decide when to issue carbon removal units: at the point of feedstock application; after the identification of feedstock dissolution or a similar measurable outcome; or at some point after the identification of dissolution based on modelling. There is also the possibility of staggering the release of credits to take account of the longer-term modelled evolution before reaching the final reservoir.

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[70] As an example, when metal cations adsorb onto soil particles, they may eject acidic hydrogen or ammonium ions that stimulate CO_2 outgassing – at least until the metal cations leach back out and the effect reverses. The diversity of soil properties means that such delays are poorly understood and the subject of ongoing research (Paessler et al., 2025; Pogge von Strandmann et al., 2021).

4.1.4 Modelling losses

Following on from the previous sub-section, some delays will be so long that they can most appropriately be treated as losses. These could include, for example, enhanced formation of clays at the project site and precipitation of solid carbonates downstream of it: in the first instance, careful measurements in the field can be used to detect and quantify the loss term; but it is not feasible for project operators to track the flow of physical molecules arbitrarily far from the project site (in the catchment, groundwater, and deep soil). Estimating these carbon loss pathways necessitates the use of models.

These models encompass, with varying levels of detail, chemical transport and reactions in soils, the hydrology and chemical signatures of specific watersheds, interactions with crops and biota in the soil and water, and the residence time in various pools during transport out to sea. While they are growing in scope, complexity, and sophistication, the field could be said to still be in early stages of development, with a number of modelling frameworks available and a lack of convergence or consensus on what the most accurate and appropriate choices are for a given project location and loss pathway.

Existing voluntary standards leave the door open for project developers to propose their own assessments of downstream carbon loss. As with the lags discussed in the previous sub-section, additional research to (i) elucidate the key factors that could contribute to carbon loss, and (ii) develop expected percentage loss ranges as they depend on these factors, would be a useful contribution to the space.

4.1.5 Indirect emissions

Some potential feedstocks have existing uses, e.g. as fine aggregate for concrete production. Where demand for these feedstocks for use in ERW projects diverts the material from these existing uses, this could lead to indirect emissions. For example, the Puro ERW methodology says, “In the context of enhanced weathering, economic leakage may occur, primarily, if the weathering material (whether a primary product or a burden-free co-product) was already used to deliver another product or service, and thereby possibly entail the extraction of additional primary material, if demand persists. In that case, the LCA shall include primary material extraction.”.

4.1.6 Organic carbon and non-CO_2 gases

Emissions of methane and nitrous oxide may be augmented or suppressed by changes to soil chemical make-up following an ERW amendment. Both gases are produced and metabolised by soil microbes (the latter especially in nitrogen-rich environments), and changes to the physiological balance and/or composition of the microbial population would affect the net greenhouse gas impact of ERW.

It is also possible that ERW activities may stimulate biomass and SOC changes. Promoters of ERW emphasise its potential to increase agricultural productivity, which would tend to amplify carbon benefits (Garrett Boudinot et al., 2023), but the potential net impact on soil organic carbon is not yet fully established (Lei et al., 2025b). Given that organic carbon fluxes tend to be higher than inorganic carbon fluxes, it is conceivable that the magnitude of these changes could eclipse the formation of DIC. Transparent reporting by ERW project operators of shifts in organic carbon flows would augment our understanding of the extent to which ERW can promote additional nature-based (i.e. temporary) sources and sinks of carbon. Any persistent detriment to SOC stocks would reduce the net climate benefit of an ERW activity, but SOC is difficult to monitor and subject to variability. Ideally ERW activities would be targeted to contexts where there was high confidence of either no net detriment to, or an increase in, SOC and biomass carbon following application. If this confidence is not possible then consideration should be given to whether some form of SOC monitoring requirement would be appropriate as part of a certification approach. The Isometric standard requires that a representative SOC measurement should be taken during baseline characterisation and then again after about 5 years, but at this time there is no requirement to include any SOC loss that is estimated based on these measurements in the quantification of associated emissions.

The Commission could decide to disqualify ERW projects on high-carbon-stock soils (e.g. with SOC > 5%) or on soils that are particularly vulnerable to carbon loss: this would preclude cases with the risk of the largest SOC emissions and would give prospective developers some clarity in which areas are off-limits. This would not, however, be a substitute for an approach that either integrated robust SOC monitoring, or was more able to better distinguish cases where SOC loss was unlikely.

4.2 Additionality and baselining

4.2.1 Financial additionality

Applying ERW materials to soils adds alkalinity and may disperse additional nutrients into soils that have agricultural value and allow increases in agricultural yield. Similarly to biochar application, there is therefore a dual value available from ERW practices – the agricultural value plus the value of the delivered carbon removals. If it could be demonstrated that ERW practices were financially viable (in at least some cases and for some application rates) without the value from carbon removal units, this could affect the assessment of the baseline for the activity.

4.2.2 Baselining for by-product feedstocks

Some candidate ERW feedstocks are by-products of other industries: quarry dust, fly ash, steel slag, crushed concrete, etc. These have the advantage of lower lifecycle emissions, but may be associated with non-zero carbon removal in their existing disposal (Stubbs et al., 2025). Candidate feedstocks which are likely to have significantly weathered under typical conditions (i.e. those which would be exposed to water and CO_2 anyway) may experience a comparable amount of weathering in the baseline to that which would be delivered by application on agricultural land.

The counterfactual weathering rate of ERW feedstock could be assessed on a case-by-case basis, with periodic review to adjust to changes in standard disposal practices. Fine material exposed to air and water for extended periods may weather quickly without intervention. However, certain conditions can dramatically slow the natural weathering progress. Formation of mineral crusts on the surface of compacted material hinders penetration of CO_2 (as discussed in another technical report, Sandford & Malins (2025a)); and cold and basic conditions hinder feedstock dissolution (as was found for a rock flour deposit in Greenland (Ray Sarkar, 2023)).

Feedstocks with a high potential rate of natural weathering could be excluded from eligibility under the CRCF by using either a negative or positive list – such a list could be revised if the understanding of likely weathering rates for a given case improves. Alternatively, blanket exclusion could be avoided by adopting robust requirements to calculate material specific baseline removals estimates[71], but this would be subject to the development of an adequate evidence base for such estimates.

These options could potentially be operationalised through the certification schemes, by setting a more generic requirement that the approach to baselining should be appropriately conservative – thereby allowing individual certification schemes to either adopt a list approach or to propose baseline estimation approaches.

4.2.3 Baseline farming activity

As discussed above, ERW activities could fulfil a comparable agricultural function to conventional liming. At the national inventory level, liming is considered an emissions source (e.g. by the IPCC); but under a range of conditions has carbon removal potential (Hamilton et al., 2007). This represents a baseline issue if an ERW practice fully or partially replaces a liming practice.

Baseline rates of liming could be assessed based on farm records for the project site in question, possibly in combination with an assessment of common practice among farms in the locality. Isometric, for example, requires that a control plot is maintained following business as usual farming practices including liming, stating that this, “ensures that any removals associated with business as usual agricultural liming are not attributed to ERW in agriculture projects.”.

On the other hand, reducing demand for lime products would avoid emissions from limestone grinding and potentially from calcination to produce quicklime. It should be considered whether, if liming is to be included in the baseline, it would be appropriate to offset the avoided lime production emissions against $GHG_{associated}$.

4.2.4 Natural alkalinity flux

In a scenario where soil pH remains low (e.g. there is no ERW amendment), acids filtering down through the soil may participate in natural weathering of bedrock material (Wan et al., 2021). This can be seen as representing a natural geochemical carbon sink, as it increases soil water pH and enables bicarbonate formation. The rate of alkalinity release depends on the mineral composition of the bedrock in question – e.g. limestone would be expected to weather more quickly – but can be significant (Wittmann et al., 2024).

Adding alkalinity to the top layers of soil through ERW or liming activity may disrupt this natural weathering, and this would have implications for calculating the net carbon impact of the alkalinity enhancement activity. We expect that the effect would be more significant for shallow, well-drained, calcareous soils (Renforth et al., 2015). There may also be an effect where enhanced biological activity owing to the ERW amendment accelerates mineral dissolution, compensating for the reduction in natural weathering. Incorporating natural weathering into the baseline may be appropriate in situations where this background is likely to be significant, and is considered by Isometric (2025a, 2025b), for example.

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[71] This could include empirical measurements of natural weathering progress (at the same location and depth as feedstock extraction), and where appropriate, explanation of the key parameters that are limiting natural recarbonation.

4.3 Long-term storage and liability

4.3.1 Loss or reversal due to improved understanding or changed physical conditions

When it comes to downstream losses, existing ERW certification methodologies are framed as being conservative, i.e. erring on the side of understating the effectiveness of projects. This is done through adopting conservative factors and imposing strict measurement protocols to reduce the risk of over-estimating removals.

Even so, there are two eventualities that are not directly covered by these conservative accounting practices. First, unforeseen exogenous developments outside the scope of certified projects could lead to loss or reversal after feedstock dissolution – for example, a drought might dry up a river, causing some of the carbon in dissolved bicarbonate to be released to the atmosphere before reaching the ocean. Second, improvements in our understanding of the fundamental geochemistry could potentially lead to reductions (though also potentially to increases) in the best estimate of the CDR delivered by previous projects.

Consider an activity that generated carbon removals for five years based on the calculation rules in place at the time, but where later revisions to the best available modelling suggested that only three-quarters of the initial estimate would have been durably delivered. A decision would need to be taken about whether this should be treated as a reversal, and, if so, how to assign liability for the reversal. Some existing carbon standards include buffer pools as a form of insurance against reversals. Both Isometric and Rainbow require ERW projects to contribute a portion of their credits into a buffer pool. Isometric specifically highlights the evolving scientific understanding of DIC stability as a risk factor.

It would need to be decided whether a certification scheme under the CRCF should introduce a monitoring and liability system for reversals of this sort, or would treat these risks as adequately dealt with through a conservative quantification methodology.

4.4 Sustainability

4.4.1 Eligible feedstock types

ERW feedstock can be divided into naturally-occurring rocks and industrial wastes. Within naturally-occurring rocks we can also differentiate between ‘virgin’ by-products (i.e. by-products requiring no further processing such as crushing), processed by-products, and purpose-mined material; within industrial wastes, there is a wide range of source industries and material compositions. A CRCF certification methodology could consider whether any specific feedstocks should be ruled out as a precaution.

While this report has occasionally used magnesium silicate from basalt and calcium carbonate from limestone to exemplify aspects of the geochemistry, it is important to remember that a range of rock and mineral types could be used for ERW. The lesser overall abundance of other silicates like nepheline and other carbonates like dolomite should not preclude their use where they are locally available (unless of course there are sustainability issues).

4.4.2 Eligible sites

The focus in the embryonic ERW industry is on application of material to agricultural soils, but as noted in section 2.4.1 there is also the possibility of application in non-agricultural contexts such as forestry, urban soils, and industrial sites. The evaluation of associated benefits and of sustainability risks will differ between types of site. It would need to be decided whether an initial certification methodology under the CRCF would restrict eligibility to agricultural contexts or offer certification for a wider range of systems.

Even within a single land type, areas can be considered more or less favourable for ERW projects. For example, agricultural soils with high organic content might be seen to pose a higher SOC depletion risk; soils with high clay-forming potential are more likely to lead secondary mineral formation; and sites where water runoff flows into a calcium-saturated catchment will have a greater risk of secondary carbonate precipitation.

A certification framework may opt to restrict eligibility to land areas which are known to satisfy certain (clearly specified) thresholds; or simply require applicants to flag where their project sites may exceed these thresholds and commit to ongoing risk monitoring.

4.4.3 Soil contamination

The regular application of ERW feedstocks in large quantities could potentially lead to soil contamination with heavy metals like nickel and chromium, or other toxicants. When rocks like basalt weather in the soil, these metals can be released, potentially accumulating to levels that have long-lasting harmful effects on plants, soil microorganisms, and water quality, or that could affect food safety.

Different soils will have different background concentrations of heavy metals and other contaminants, depending on local geology, land use practices over time, and what other activities are in the watershed. In some instances, contamination may already be close to or exceed acceptable levels. Soil testing would indicate which potential sites could be unsuitable for ERW activity.

Existing voluntary schemes have made testing of feedstock materials and/or routine monitoring of soil constituents a prominent feature of their criteria. Isometric requires that contaminant concentrations do not contravene applicable local regulations. Puro further notes that the EU’s Fertilising Products Regulation would place limits on the content of chemical contaminants for ERW feedstock (in units of mg of contaminant per kg of feedstock); but it indicates that these may in some cases be too stringent for ERW feedstock, noting that the thresholds were developed for fast-acting soil amendments, whereas ERW feedstock tends to dissolve more slowly. Depending on the application rate, a given contaminant content might lead to lower realised soil contamination in the ERW case than for another type of fertilising product. The same is true of industrial waste disposal, where toxicological effects are sensitive to the type of material under consideration.

If a CRCF methodology is developed for ERW, it would be appropriate to explore whether ERW-specific feedstock contaminant thresholds could be adopted. These would depend on the feedstock type, the application schedule, and environmental conditions. Given that ERW projects will be subject to oversight by certification bodies and schemes, there is also a potential opportunity for knowledge development and sharing to inform future methodological development, for example by monitoring the bio-availability of contaminants present in different feedstocks.

4.4.4 Ecologically sensitive areas

ERW amendments may influence the chemical and pH conditions of nearby or downstream ecosystems, with potentially beneficial or adverse consequences. For example, peat bogs and wetlands have been identified as susceptible to ecological disruption, as well as degradation of organic carbon stores. Proposed ERW projects could be subject to evaluation of their impacts on sensitive or fragile land types, areas of scientific interest, or habitats of conservation concern – both in the local area and downstream (for large watersheds, this may require consideration of cross-border impacts). Prospective project operators would benefit from demarcated soil-type maps, and/or pH thresholds to compare outflow readings against.

Should the ERW industry begin to scale, there may also be a role for EU institutions to establish watershed-level monitoring, ensuring that the cumulative effect of distributed deployments remains within safe ecological limits.

4.4.5 Over-application

The development of the carbon removal credits market may lead farmers and land-owners to undertake ERW application even when there is no clear agricultural imperative. To avoid a situation where this degrades the productivity of the land, it may be appropriate to require certification applicants to provide evidence that the ERW measures to be taken are consistent with sound agronomic planning.

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Annex 1 Variables of interest for measurement and modelling

The table below summarises which variables may be measured for an ERW project. Variables are split by monitoring stage (Section 2.6 of the main text). More detail and references to the literature can be found in the original source (Holzer et al., 2023).

Stage	Variable	Measurement Method	Monitoring Strategy	Description
Feedstock characterisation	Project emissions	LCA	Other	Project emissions must be accounted for when estimating net carbon removal from an enhanced weathering deployment.
	Rock application	Record keeping, weighing, monitoring	Measurement, Records	Records of material application rates, dates, and rock moisture content at the time of application for determining the mass of applied material and its weathering potential.
	Rock elemental composition	XRF, complete digestion plus ICP-MS or ICP-OES	Measurement	Elemental composition identifies the abundance and distribution of chemical elements within the rock (silica content, cation ratios, trace metals, ...).
	Rock mineralogy	QEMSCAN, XRD	Measurement	Mineralogy identifies the relative proportions of minerals within a rock (places an upper bound on theoretical carbon removal and helps reveal what fraction of the rock is likely to weather quickly).
	Rock particle size distribution	Particle size analyser or sieve method	Measurement	Particle size distribution is valuable for estimating weathering potential and suitability of rock material.
	Rock surface area	BET	Measurement	Specific surface area of the applied weathering material can provide additional information to model weathering potential.
	Upstream emissions	LCA	Other	Upstream emissions must be accounted for when estimating net carbon removal from an enhanced weathering deployment.
Field weathering	Bulk density	Various (e.g., Geoprobe sampling, core method, clod method, excavation method)	Measurement	Bulk density refers to the mass of dry soil per unit volume and is influenced by factors including soil type and field management.
	Cation exchange capacity (CEC) and exchange complex cations	NH_4OAc, $BaCl_2$, or appropriate extraction for given soil	Measurement	CEC refers to the negatively charged sites on soil solid surfaces (e.g., clay minerals, organic matter) where cations are attracted, held, and exchanged with soil pore water. Characterising the CEC can be important for interpreting other weathering signals.
	Cations or anions	ICP-AES, ICP-MS, AAS, IC	Measurement	Direct measurements of ions in soil water or downstream export can provide evidence of weathering at the field or watershed scale.

Stage	Variable	Measurement Method	Monitoring Strategy	Description
	Crop yield	Record keeping, weighing	Measurement, Records	Changes in yield due to rock application.
	Dissolved inorganic carbon (DIC)	DIC analyser	Measurement	Increases in total dissolved inorganic carbon (DIC) can provide evidence of weathering and be used to estimate CO_2 removal at the field or watershed scale
	Electrical conductivity	EC probe, sensors	Measurement	Electrical conductivity (EC) is typically correlated with total alkalinity.
	Field management	Record keeping	Records	Includes application of agricultural lime and fertilizer, tillage patterns, and crop type and rotation -- these inform estimates of initial weathering and other field-scale processes.
	Field outgassing	Default discount factor	Other	Default factors may be used to account for leakage from field processes that result in CO_2 outgassing (e.g., carbonate precipitation or secondary mineral formation).
	Greenhouse gas fluxes	Various (e.g., chamber, Eddy covariance flux tower, remote sensing)	Measurement	Gas flux measurements inform understanding of total carbon budget of the ecosystem, or indicate changes to other gas fluxes such as N2O emissions.
	Irrigation	Flow meter, record keeping	Measurement, Records	Determining a water balance for the field is important for converting measurements expressed as concentrations (mass / volume) into weathering or carbon removal fluxes (mass / time).
	Isotopic tracers	Various (e.g., MC-ICP-MS)	Measurement	Changes in the concentrations of isotopic tracers (e.g., strontium or neodymium) could be used to understand rock dissolution and weathering dynamics.
	Mobile / immobile element ratios	Geochemical mass balance	Measurement	Comparing concentrations of immobile elements like titanium (Ti) or zirconium (Zr) which originate in the weathering material and remain in the soil as weathering progresses to concentrations of mobile elements, like silicon (Si), calcium (Ca), sodium (Na), or magnesium (Mg), which are progressively lost.
	pCO_2	Sensors, infrared gas analysers	Measurement	The partial pressure of CO_2 in the soil or watershed influences initial weathering rates and provides information about carbonate equilibria in a system.
	pH	pH meter, sensors	Measurement	An increase in pH could signal that weathering has occurred, and depends on the weathering rate, the starting pH of the system, and the buffering capacity of the soil.
	Plant cation uptake	ICP-AES, ICP-MS, AAS, record keeping	Measurement	Increased consumption of cations at the field scale following rock application

Stage	Variable	Measurement Method	Monitoring Strategy	Description
				reduces the potential CO_2 removal of the associated enhanced weathering.
	Precipitation	Local weather station, record keeping, or other	Measurement, Records	Determining a water balance for the field is important for converting measurements of weathering products or carbon, which are expressed as concentrations (mass / volume), into weathering or carbon removal fluxes (mass / time).
	Secondary mineral formation	Various (e.g., SEM-EDS, TEM, XANES, NanoSIMS, XRD)	Measurement	Secondary mineral formation can bind up cations released during initial rock weathering and therefore reduce net carbon removal.
	Soil and air temperature	Sensors, local weather station, record keeping	Measurement	Soil and air temperature measurements could potentially be used to constrain weathering rates and equilibria dynamics.
	Soil biology	Various (e.g., PLFA, DNA sequencing)	Measurement	Changing pH or micronutrient availability will have impacts on soil biology, including microbial diversity and functioning.
	Soil inorganic carbon	Various (e.g., TGA, pressure calcimeter, acid dissolution)	Measurement	Inorganic carbon content of soil solids to characterise in-field carbonate precipitation.
	Soil moisture	Sensors, soil samples	Measurement	Water balance and moisture impacts on weathering rates.
	Soil organic carbon	Various (e.g., EA, LOI; can be combined with methods for isolating SOC fractions)	Measurement	Adding minerals to soil has the potential to induce changes to soil organic carbon (SOC), for example by affecting pH, impacting plant or microbial activity, or changing secondary mineral content.
	Soil physical properties (water infiltration, aggregate stability)	Various (e.g., infiltrometers, wet sieving, ultrasonic dispersion, immersion, SLAKES)	Measurement	Soil physical properties, like water infiltration or aggregate stability, could inform an understanding of initial weathering processes as well as changes to the soil following rock application.
	Soil texture	Particle size analyser, texture by hydrometer	Measurement	Soil texture measurements are important for characterizing the soil system, including its suitability for enhanced weathering, field processes that may influence net carbon removal, and likely impacts of enhanced weathering for soil health.
	Soil total carbon	Elemental analyser (EA)	Measurement	Soil total carbon measurements characterize the combined organic and inorganic carbon content of soil samples.
	Total alkalinity (TA)	Titrations	Measurement	Changes in total alkalinity can provide evidence of weathering, as bicarbonate ions are generally the dominant contributor to alkalinity.

Stage	Variable	Measurement Method	Monitoring Strategy	Description
	Trace metals	ICP-AES, ICP-MS, AAS	Measurement	Trace metal concentrations may inform estimates of initial rock weathering.
	Weathering rate and field outgassing	Process or empirical models	Model	Process models like reactive transport models can be used to characterize initial rock dissolution and subsequent reactions as weathering products move within the soil column.
Downstream transport	Ocean outgassing	Default discount factor	Other	Default factors may be used to account for leakage from ocean processes that result in CO_2 outgassing (e.g., carbonate precipitation or secondary mineral formation).
	Ocean outgassing	Process or empirical models	Model	Process models of ocean circulation and ocean equilibration with the atmosphere can be used to estimate additional ocean outgassing of CO_2, that occurs as a result of enhanced rock weathering.
	River outgassing	Default discount factor	Other	Default factors may be used to account for leakage from watershed processes that result in CO_2 outgassing (e.g., carbonate precipitation or secondary mineral formation).
	River outgassing	Process or empirical models	Model	Process or empirical models of river outgassing may be used to characterize watershed processes that affect net carbon removal, such as carbonate precipitation or evasion, and their impact on carbon removal uncertainty.

Source: Adapted from Holzer et al. (2023)