An Ecosystem of Carbon Dioxide Removal Reviews – Part 3: Enhanced Weathering

T.J. Suhrhoff[1,2*], C. Dietzen[3], T. Kukla [4], A. Lunstrum[5], T. Reershemius[6], J. Thompson[2], M. Almaraz[1], M. Bertagni[7], L. Bullock[8], R. D'Ascanio[2], I. Falkson[2], P. Frings[9], R. Gregg[2], N. Iff[10], J. Lambert[11], Y. Li[12], B. Rogers[13], J.M. Schneider[13], E. Swanson[2], F. Tao[14], S.S. Tsao[15], R. Van Der Bauwhede[16], S. Zhang[12], S.K. Anand[17], J.S. Campbell[18], I. Chiaravalloti[2], I. Davis[19], M. Dobson[19], X. Dupla[20], S. Foteinis[18], M. Guo[21], K. Harrington[22,18], C. Kent[2], A. Klemme[23], J. Kroeger[15], T. Linke[24], S. Linnekogel[25], S. Moller[1], E. Milliken[2], L. Ndlovu[26], H. Niron[27], S. Patel[26], E. Pihlap[28], K. Rees[29,30], R. Rioux[15], M. Rizzi[3], S. Shaheen[15], L. Steinwidder[27], I. Steeley[29], T. Sweere[31], F. Sun[15], X. Sun[12], W. Tatge[15], L. Tejedal Lemus[13], A. Vienne[27], J. Westphalen[32], B. Woollen[33], C.M. Baum[34], S.L. Brantley[35], S. Calabrese[17], T. Cyronak[32], T. Dorndorf[36,37], C. Fyson[36], M. Hagens[38], J. Hartmann[24], I.O. Holzer[39], B.Z. Houlton[40], R.H. James[19], K. Jensen[41], Y. Kanzaki[42], A. Khan[33], C. Levy[43], S. Lück[36], D.A.C. Manning[6], J.R.H. Manning[44], J. Meyer zu Drewer[45,46], R.B. Neumann[47], C.R. Pearce[48], P.A.E. Pogge von Strandmann[49], I.M. Power[21], P. Raymond[15,1], P. Renforth[18], T. Repke[36], J. Saiers[15], R. Santos[50], J.D. Smolen[51,52], L. Tan[2], S. Vicca[27], M.E. Vorrath[24], R.M. Webb[53], Y. Yao[15], B. Zhang[15], C.T. Reinhard[42], N.J. Planavsky[2,1], S. Fuss[36,54]

[1] Yale Center for Natural Carbon Capture, Yale University, New Haven, CT, 06511, USA
[2] Department of Earth and Planetary Sciences, Yale University, New Haven, CT, 06511, USA
[3] Globe Institute, University of Copenhagen, Copenhagen, Denmark
[4] Carbonplan, San Francisco, CA, USA
[5] School of Engineering and Applied Science, University of Pennsylvania, Philadelphia, PA, 19104, USA
[6] School of Natural and Environmental Sciences, Newcastle University, Newcastle upon Tyne, England, NE1 7RU, UK
[7] Department of Environment, Land and Infrastructure Engineering, Polytechnic University of Turin, Turin, Italy
[8] Department of Geological Resources for the Ecological Transition, Spanish Geological and Mining Institute, Spanish National Research Council, Murcia, Spain
[9] GFZ German Research Centre for Geosciences, Organic and Earth Surface Geochemistry, Telegrafenberg, Potsdam, 14473, Germany
[10] Department of Earth Sciences, Freie University Berlin, Berlin, Germany
[11] NYU Gallatin School of Individualized Study, New York, NY, USA
[12] Department of Oceanography, Texas A&M University, College Station, TX, 77843, USA
[13] Department of Earth System Science, Stanford University, Stanford, CA, USA

[14] Department of Informatics and Intelligent Systems, Institute of Energy and the Environment, The Pennsylvania State University, University Park, PA, USA
[15] School of the Environment, Yale University, New Haven, CT, 06511, USA
[16] Department of Earth and Environmental Sciences, Division of Soil and Water Management, KU Leuven, Leuven, Belgium
[17] Department of Biological and Agricultural Engineering, Texas A&M University, College Station, TX, 77843, USA
[18] Research Centre for Carbon Solutions, Heriot-Watt University, Edinburgh, EH14 4AS, UK
[19] School of Ocean and Earth Science, University of Southampton, Waterfront Campus, National Oceanography Centre, European Way, Southampton, SO14 3ZH, UK
[20] Department of Earth and Planetary Sciences, ETH Zurich, Zurich, Switzerland
[21] Trent School of the Environment, Trent University, Peterborough, Ontario, Canada
[22] Smith School of Enterprise and the Environment, University of Oxford, Oxfordshire, England, UK
[23] Institute of Environmental Physics, University of Bremen, Bremen, Germany
[24] Department of Earth System Sciences, University of Hamburg, Hamburg, 20146, Germany
[25] UK Centre for Ecology and Hydrology, Library Avenue, Bailrigg, Lancaster, LA1 4AP, UK
[26] Department of Chemical and Biomolecular Engineering, University of Pennsylvania, Philadelphia, PA, 19104, USA
[27] Biobased Sustainability Engineering (SUSTAIN), Department of Bioscience Engineering, University of Antwerp, Antwerp, Belgium
[28] Institute of Ecology and Earth Sciences, University of Tartu, Tartu, 50409, Estonia
[29] School of Biosciences, University of Sheffield, Sheffield, UK
[30] Leverhulme Centre for Climate Change Mitigation, University of Sheffield, Sheffield, United Kingdom
[31] Department of Earth and Planetary Sciences, Institute of Geochemistry and Petrology, ETH Zurich, Zurich, Switzerland
[32] School of Earth, Environment and Sustainability, Institute of Coastal Plain Sciences, Georgia Southern University, GA, 31419, USA
[33] Carbon Removal Standards Initiative, San Francisco, CA, 94114, USA
[34] Department of Business Development and Technology, Aarhus University, Aarhus, Denmark
[35] Department of Geosciences and Earth and Environmental Systems Institute, Pennsylvania State University, University Park, PA, USA
[36] Potsdam Institute for Climate Impact Research (PIK), Member of the Leibniz Association, Potsdam, 14482, Germany
[37] Geographic Institute, Humboldt University of Berlin, Berlin, Germany
[38] Soil Chemistry Group, Wageningen University and Research, Wageningen, 6700 AA, The Netherlands
[39] Environmental Studies Program, University of California, Santa Barbara, Santa Barbara, CA 93106, USA
[40] Ecology and Evolutionary Biology and The Ashley School of Global Development and the Environment, Cornell University, Ithaca, NY, USA
[41] Environmental Defense Fund, NY, USA
[42] Georgia Institute of Technology, Atlanta, GA, USA
[43] Science & Innovation, Carbon 180, Washington, DC, USA
[44] School of Chemical Engineering, University of Manchester, Manchester, M13 9PL, UK
[45] Ithaka Institute for Carbon Strategies, Goldbach, 63773, Germany
[46] Institute of Sustainable Energy Systems, Offenburg University, Offenburg, Germany
[47] Department of Civil and Environmental Engineering, University of Washington, Seattle, WA 98195, USA
[48] National Oceanography Centre, European Way, Southampton, SO14 3ZH, UK
[49] Mainz Isotope and Geochemistry Centre (MIGHTY), Institute of Geosciences, Johannes Gutenberg University Mainz, Mainz, 55128, Germany
[50] University of Guelph, Guelph, Canada
[51] Department of Earth and Planetary Sciences, Harvard University, Cambridge, MA, 02138, USA
[52] Department of Human Evolutionary Biology, Harvard University, Cambridge, MA, 02138, USA
[53] Sabin Center for Climate Change Law, Columbia Law School, New York, NY, 10017, USA
[54] Geography Department and IRI THESys, Humboldt University of Berlin, Berlin, Germany

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Table of abbreviations

Abbreviation	Definition
AAEM	Alkali- and alkaline earth metals
ANC	Acid-neutralizing capacity
AR6	Sixth Assessment Report of the IPCC
BECCS	Bioenergy with Carbon Capture and Storage
BET	Brunauer–Emmett–Teller
BGS	British Geological Survey
BiCRS	Biomass Carbon Removal and Storage
BS	Base saturation
CAPEX	Capital expenditure
CCUS	Carbon Capture, Utilization, and Storage
CDR	Carbon Dioxide Removal
CEC	Cation exchange capacity
DACCS	Direct air carbon capture and storage
DIC	Dissolved inorganic carbon
EC	Electrical conductivity
EPPA	Emissions Prediction and Policy Analysis
ERW	Enhanced rock weathering
ESROC	Ecosystem of Systematic Reviews on CDR
EW	Enhanced Weathering
GCAM	Global Change Analysis Model
GDP	Gross domestic product
GHG(s)	Greenhouse gas(es)
GRI	Grantham Research Institute
GSA	Geometric surface area
IAM(s)	Integrated assessment model(s)
IC	Ion chromatography
ICP-MS	Inductively coupled plasma mass spectrometry
ICP-OES	Inductively coupled plasma optical emission spectroscopy
IPCC	Intergovernmental Panel on Climate Change
IQR	Interquartile range
IUGS	International Union of Geological Sciences
LCA(s)	Life cycle assessment(s)
LOI	Loss on ignition
MAOM	Mineral-associated organic matter
MRV	Monitoring, reporting, and verification
NACSOS2	NACSOS2 review management platform
NFZ	Near field zone
NIMBYism	Not-in-my-backyard ism

NUE	Nitrogen use efficiency
OAE	Ocean alkalinity enhancement
OPEX	Operating expenditure
POM	Particulate organic matter
ppmv	Parts per million by volume
PRISMA	Preferred Reporting Items for Systematic Reviews and Meta-Analyses
PTTE	Potentially toxic trace elements
QA/QC	Quality assurance / quality control
REMIND(-MAgPIE)	Regional Model of Investments and Development (linked to the Model of Agricultural Production and its Impact on the Environment)
ROSES	Reporting standards for Systematic Evidence Syntheses
RTM(s)	Reactive transport model(s)
SDG(s)	Sustainable Development Goal(s)
SI	Supplementary Information
SIC	Soil inorganic carbon
SOC	Soil organic carbon
SOM	Soil organic matter
TA	Total alkalinity
TAS	Total alkali–silica
TIMS	Thermal ionization mass spectrometry
TRL(s)	Technology readiness level(s)
UK	United Kingdom
US	United States
USD	United States dollar
USGS	United States Geological Survey
VCM	Voluntary Carbon Market
wt%	Weight percent
XRD	X-ray diffraction
XRF	X-ray fluorescence

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Abstract

Enhanced Weathering (EW) is an emerging Carbon Dioxide Removal (CDR) approach within a growing portfolio of mitigation strategies, offering the potential for durable CDR alongside agronomic co-benefits. As interest in CDR increases, EW is transitioning from a primarily scientific concept toward early-stage implementation, requiring a comprehensive synthesis of the current evidence base. This systematic review focuses primarily on soil-based EW using silicate rock feedstocks. Across empirical and modelling studies, area-normalized CDR fluxes have a median value of 0.84 tonnes of carbon dioxide per hectare per year ($tCO_2\ ha^{-1}\ yr^{-1}$) and span several orders of magnitude. The variance in reported fluxes reflects not only context-dependent differences in EW performance, but also the diversity of quantification approaches, which differ by use case and vary in terms of their system boundaries and treatment of loss processes. When scaled globally, evidence from empirical constraints and/or modelling approaches converges on a maximum technical CDR potential of ~0.2–2 $GtCO2\ yr^{-1}$, though some estimates are higher. Significant uncertainty remains regarding the magnitude, persistence, and timing of loss processes after initial CDR has occurred, including secondary phase formation and cation exchange, and in how these losses depend on local soil and climate conditions and deployment strategies. Interactions with soil organic carbon introduce additional uncertainty but also the potential for increasing total CDR. Beyond CDR, EW is consistently associated with improvements in soil properties and crop productivity. Its deployment within managed land systems creates opportunities for synergies with other land-based CDR approaches, enabling combined inorganic and biological carbon sequestration. At the same time, EW raises important socio-economic and governance considerations, including distributional impacts, environmental risks, and conditional public support. Overall, EW represents a context-dependent and multifunctional CDR pathway, with increasing evidence for its potential to contribute to both climate mitigation and multiple Sustainable Development Goals. Its effective and scalable deployment will depend on improved standardisation, robust monitoring and verification, and alignment with economic, environmental, and societal constraints.

1 Introduction

Carbon Dioxide Removal (CDR) methods are increasingly recognised as an essential complement to rapid and sustained emissions reductions to limit global temperature rise in line with the Paris Agreement and comply with a growing number of sub-national, national and international targets and assessments.[1–5] Across most mitigation pathways consistent with the Paris agreement, CDR is required within the coming decade to compensate for residual emissions from hard-to-abate sectors, limit or reverse temperature overshoot, and provide insurance against climate system uncertainties.[2,4–12]

Existing reviews have provided valuable high-level syntheses of the overall CDR landscape, summarising broad categories of methods, their theoretical potential, and their possible roles in mitigation pathways.[2,3,6–8] These assessments have been instrumental in establishing CDR as a core component of climate stabilisation strategies and in enabling comparison across methods at the global scale. Nevertheless, as CDR research and deployment progress there is increasing demand for complementary assessments that examine individual CDR methods in greater depth, with consistent treatment of empirical evidence, modelling assumptions, and methodological choices.

This systematic review assesses the available evidence on Enhanced Weathering (EW; also called Enhanced Rock Weathering, ERW), a novel land-based geochemical CDR approach that accelerates the natural weathering of rocks through grinding and application to soils, thereby enhancing CO_2 uptake and long-term carbon storage in dissolved inorganic carbon (DIC), secondary carbonate minerals, and marine reservoirs.[13–26] Enhanced Weathering has attracted increasing attention due to its potential scalability, its compatibility with existing agricultural practices, and the possibility of co-benefits for soil chemistry, nutrient supply, and crop productivity.[27,28] At the same time, EW research spans a wide range of disciplines, including geochemistry, agronomy, soil science, hydrology, and life cycle assessment (LCA), resulting in a heterogeneous and fragmented evidence base. Carbonate rock powder additions to soils, commonly referred to as liming, have long been used to manage soil pH and can also generate CDR.[29–31] Carbonate-based EW studies are included where available, but because the current EW literature is dominated by silicate feedstocks, the main synthesis focuses on silicate-based EW. Similarly, the co-benefits assessment incorporates new syntheses of silicate rock-flour soil amendments while drawing on existing meta-analyses to utilize evidence from agricultural liming.

This systematic review of EW is part of a broader ecosystem of coordinated CDR systematic reviews that aims to provide a rigorous, detailed, and methodologically consistent assessment of the state of knowledge across a wide portfolio of CDR approaches and cross-cutting topics. Similar systematic reviews within this ecosystem have been undertaken for other CDR methods, including direct air carbon capture and storage (DACCS),[32] establishing a common foundation for synthesis and comparison across approaches. Within this ecosystem of reviews, all contributions follow a harmonised set of definitions, assessment methods, and review designs. This coordination enables cross-method synthesis, supports cumulative learning across the CDR research community, and enhances the utility of the reviews for scientific assessments, policy processes, and practical decision-making related to the future role of CDR.

This review begins by describing the systematic methods used to identify, screen, and code the EW literature, followed by an overview of the size, growth, and disciplinary composition of the existing evidence base. It then provides a structured synthesis of the fundamental principles of EW, key factors controlling weathering efficiency, transport and long-term carbon storage pathways, as well as the major loss processes, reversals, and limitations that affect net CDR. Building on this process-level understanding, the review evaluates current approaches for monitoring, reporting, and verification (MRV) across solid, aqueous, and gaseous phases, highlighting how different methods capture distinct components of the EW system and providing context for the discussion of observed CDR rates and potentials. We then synthesize reported CDR rates and potentials across spatial scales, examine costs and techno-economic considerations, and assess agronomic, environmental, and socio-economic side effects. The review further situates EW within broader LCAs, interactions with other CDR approaches, policy frameworks, equity and justice considerations, and public perceptions, before concluding with an assessment of upscaling challenges, technology readiness, and the role of EW in integrated assessment models as well as implications for future research and deployment.

2 Methods

We undertook this systematic review following the general protocol for the Ecosystem of Systematic Reviews on CDR (ESROC) and report our methods following the Reporting Standards for Systematic Evidence Syntheses (ROSES) reporting standards.[33] The following sections step through the scope of this review and the methods used to search for EW literature, screen for relevant studies, and extract data, ending with an overview of the EW literature landscape.

2.1 Scope definition and relation to adjacent mineral-based CDR pathways

Enhanced Weathering is one of several mineral-based CDR pathways that rely on the reaction of alkaline rock types with acids to neutralise acidity and generate alkalinity, thereby enabling atmospheric CO_2 uptake and long-term carbon storage. Closely related approaches include in situ mineralisation, in which CO_2 is injected into reactive geological formations and converted into stable carbonate minerals,[34,35] reactor-based or ex situ mineral carbonation, where silicate or carbonate materials are processed in engineered systems to accelerate weathering and carbonation reactions,[36] ocean alkalinity enhancement (OAE), which increases seawater alkalinity through the addition of finely ground minerals or alkalinity generated via industrial or electrochemical processes,[37,38] and EW of mine tailings, which exploits highly reactive waste materials to promote carbonation and alkalinity release within waste stockpiles or ponds.[39] While these pathways share a common geochemical foundation—namely the neutralisation of acidity and the release of alkalinity through mineral–acid reactions—they differ substantially in the environmental setting where reactions occur, the controlling processes that determine process efficiency, timescales, and effective monitoring approaches, and their relevance for land management and agricultural systems.

Within ESROC, these distinctions motivate a clear separation of scope across pathways. This review concentrates specifically on terrestrial EW as implemented through the application of rock powder feedstock to soils, where weathering reactions occur within biologically active, hydrologically complex, and heterogeneous soil environments. Related CDR approaches such as OAE are covered in separate systematic reviews being developed within ESROC. This focus allows for a targeted assessment of EW performance, limitations, monitoring approaches, and side effects under conditions most relevant to agricultural and managed land deployment.

2.2 Literature search

We searched for EW literature using the Web of Science and Scopus literature databases, which are two of the largest bibliographic databases for peer-reviewed literature. Our search query was developed and refined as part of the first global CDR literature mapping exercise and includes a range of relevant terms to capture different EW domains in the context of CDR (see SI).[28] To ensure relevance to the focus of this review, we limited our search to literature focused on EW in soil-based settings (see sections 2.1 and 2.3). The cut-off publication date for this systematic search was July 2024.

One challenge when reviewing EW literature is the large body of research on weathering and the carbon cycle in the geological past, rather than on human activities to remove CO_2. Our search query explicitly excluded specific terms relevant to previous geological eras, though we note this can have unintended consequences if relevant studies mention the geological past as contextual information and are excluded as a result. To ensure key studies were not missed, and to capture recent papers published after our systematic search cut-off date, we also included studies identified using expert knowledge and forward/backward snowballing to supplement our systematic search. The cut-off date for these non-systematic searches was April 2025. Authors of individual text sections were also free to consider studies published after this date where relevant for interpreting recent developments and ensuring that narrative discussions reflected the most up-to-date science. However, data from these later studies were not systematically extracted (see below) and are not included in quantitative assessments or figures presented in this review. In addition, we included non-peer reviewed (e.g., dissertations) and grey literature where we considered the methods to be robust and the results a valuable contribution to the EW evidence base.

Another challenge relates to the boundaries of our review: we are interested in EW in the context of CDR, but we also want to understand the side effects (both negative and positive) of EW deployment in soils. Relying solely on CDR literature could miss some of these side effects. We therefore also included studies (n = 104) focused on the agricultural use of rock flour, identified via expert knowledge and forward/backward snowballing. Only studies published after 1980 were considered, even though a small number of earlier publications existc (e.g., ref. [40]).

2.3 Inclusion and exclusion criteria for the EW literature

The objective of this review is to evaluate the efficacy, constraints, CDR potential, costs, and impacts of EW in soil-based settings, with particular emphasis on agricultural systems. We therefore included literature that explicitly examines terrestrial EW in a soil context, encompassing field trials, laboratory experiments with clear relevance for field-based EW, modelling studies, natural analogues discussed within an EW framework, reviews, policy and governance analyses, and socio-economic, equity, or justice-focused work. In each of these cases we only included literature for which EW constituted a substantive part of the analysis: Laboratory studies determining mineral dissolution, weathering, or carbonation rates were included when they explicitly linked their findings to EW deployment, for example by discussing implications for soil-based applications, surface passivation, reaction kinetics, or transport limitations. Similarly, studies assessing downstream impacts in rivers, catchments, or coastal systems were included only when these impacts were explicitly evaluated as part of a terrestrial EW system.

To maintain relevance to soil-based deployment, we excluded literature focused on reactor-based weathering or carbonation, industrial flue gas treatment, and ex situ mineral carbonation outside of soil environments. Studies on mine tailings or other industrial residues were excluded unless they explicitly assessed or proposed the application of these materials to agricultural soils, recognising that fully concentrated feedstock settings comprised of thick layers of pure rock powder without intermixed soil (i.e., mine tailings) may not be directly informative for weathering dynamics, transport, or monitoring in soils. Similarly, OAE studies were excluded unless they explicitly framed alkalinity generation as originating from terrestrial EW. Papers describing

natural weathering in soils, watersheds, or streams without an explicit "enhanced" component were excluded unless they were clearly discussed as natural analogues for EW.

Our review focuses on EW as a CDR strategy, so we identified those studies that explicitly engage with EW in the context of CDR, for example by quantifying CO_2 uptake, alkalinity generation, or net removal, or by framing EW as a climate mitigation strategy. Studies that investigated the use of silicate rock flour solely for agronomic purposes—such as soil fertility, nutrient supply, or yield effects—without reference to CDR were not incorporated into the evidence base for quantifying EW CDR but were instead separately labelled as agronomic rock-powder studies and were used to constrain agronomic responses and side effects.

While the use of carbonates in EW and CDR contexts is beginning to receive increased attention,[30,41] the existing literature remains limited. This review includes carbonate-based rock amendment studies where they explicitly frame amendments as EW for CDR. In contrast, the extensive literature on carbonate amendments for agronomic purposes, i.e., agricultural liming, is excluded from the quantitative assessment where it lacks EW framing. Where relevant, we instead draw on existing meta-analyses to contextualize insights from this broader literature on the co-benefits of rock-powder amendments.[42,43] Accordingly, this review primarily focuses on silicate feedstocks, which form the core of the EW literature and the main basis for current EW CDR assessments.

The full list of inclusion and exclusion criteria, including decision rules applied during abstract and full-text screening, are provided in the Supplementary Information (SI) to ensure transparency and reproducibility. We developed these criteria before screening the literature and iteratively adjusted them through regular reviewer meetings.

We used the review management platform NACSOS2 to screen the titles and abstracts of the records from our literature search using the criteria described above.[44] As a first step, we used the automatic classifier that was trained in Lück et al. (2025)[28] to identify papers that were most likely to be relevant. This reduced the number of records requiring manual screening from 1,151 to 826. Two expert reviewers then applied the screening criteria to identify relevant records, followed by sorting into pre-defined categories to facilitate further analysis (see section 2.4). In cases where the abstract was not sufficient to decide, reviewers consulted the full text and conferred until agreement could be reached. At the end of this process, we had a total of 241 peer-reviewed and 40 non-peer-reviewed records for inclusion in the review (see Figure 1). We have added these to a bibliometric database of EW literature, which is continually updated as new studies are published.[27]

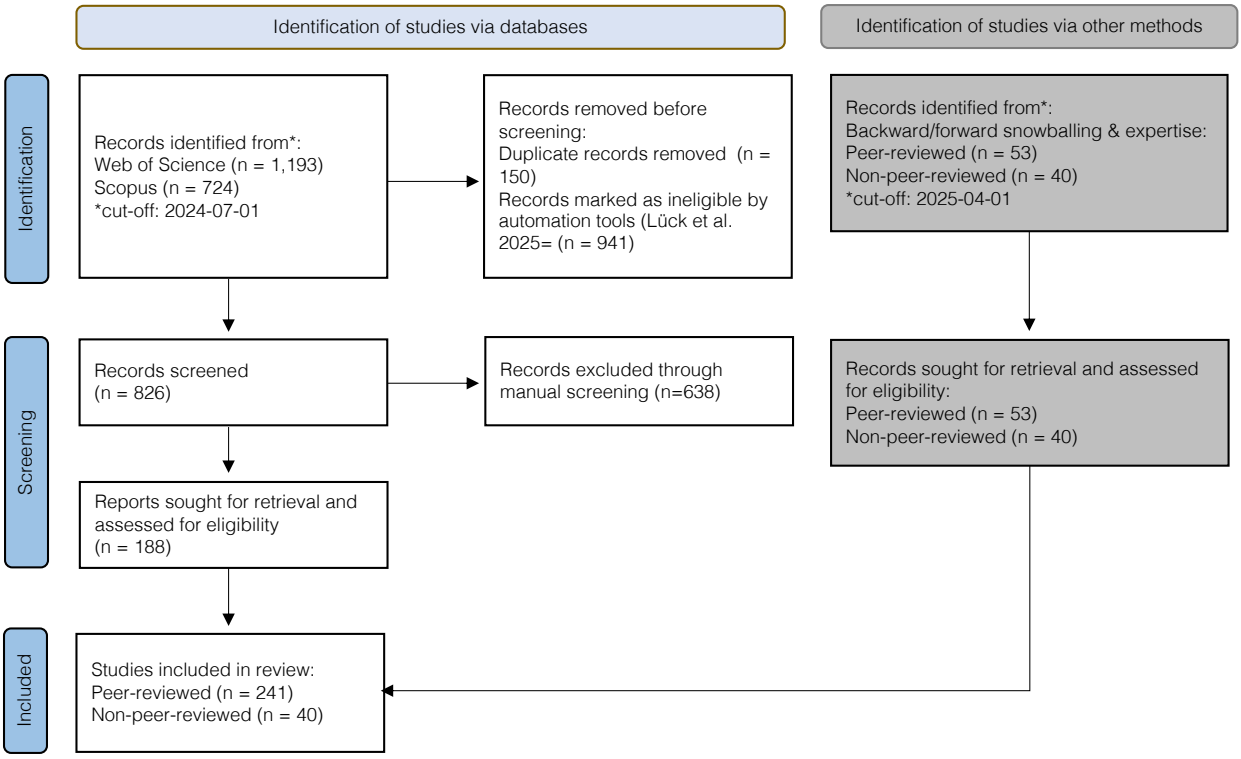

Figure 1: *Literature identification and selection process for the systematic review of soil-based Enhanced Weathering (EW) in the context of Carbon Dioxide Removal (CDR). Note that the number of studies included reflects those focused explicitly on CDR; identified literature on rock flour as a fertiliser is not included here). Figure follows PRISMA guidelines.*[45]

2.4 Data extraction

Data extraction was guided by a purpose-built codebook designed to capture the breadth and heterogeneity of the EW literature while maintaining internal consistency across multiple contributors. The codebook was developed iteratively prior to data extraction to cover reported CDR estimates, underlying geochemical and agronomic measurements, methodological choices, and a wide range of qualitative and quantitative metadata. Because the data extraction effort was embedded within a broader database and meta-analysis project, the codebook was intentionally designed to be more comprehensive than required for this systematic review alone. As a result, only a subset of the extracted variables is used directly in the synthesis presented here, while the full database, intended to support additional meta-analyses and synthesis efforts, will be made publicly available as an open-source resource upon completion.

The codebook structures data extraction into hierarchical sections capturing: (i) basic bibliographic and identifying information, including unique experiment identifiers; (ii) study type, methods, and actors involved; (iii) experimental or model conditions, including spatial scale, duration, soil type, climate, hydrology, and vegetation; (iv) detailed pre- and post-amendment measurements of soils, pore waters, effluent waters, exchangeable pools, and biomass, including elemental concentrations, carbonate system parameters, isotopic data, mineralogy, and physical soil properties; (v) feedstock characteristics and application details, such as mineral type, particle size

distribution, composition, application rate, and co-amendments; (vi) CDR estimates and associated carbon budgets reported at different spatial and temporal scales, together with methodological details on measurement, reporting, and verification (MRV), loss processes considered, and analytical approaches; (vii) quantitative LCA data and cost estimates, including system boundaries, assumptions, and reported cost components; and (viii) qualitative discussion topics, including agronomic effects, environmental and social side effects, policy landscape, and interactions with other CDR approaches.

Data extraction was carried out by a team of 34 contributors, each of whom applied the shared codebook to extract data from up to 15 publications using a standardised Excel-based data entry template. To ensure data quality and consistency across contributors, the extraction process included two organized rounds of quality assurance and quality control (QA/QC). In the first round, a designated reviewer in a supervisory role manually checked all submitted entries against predefined criteria, focusing on completeness, internal consistency, correct use of categories, unit reporting, and adherence to codebook instructions, providing structured feedback to contributors. In the second round, automated checks implemented in Python were used to identify logical inconsistencies and missing dependencies across database fields, for example ensuring that the presence of data in one category (e.g., reported CDR rates) was accompanied by corresponding information in required related categories (e.g., MRV type, system boundaries, etc.). In both rounds, contributors addressed QA/QC queries through documented responses that explicitly justified data entries and any subsequent revisions, following a structured review–response process that was subsequently verified by the designated reviewer. Additional targeted QA/QC was conducted iteratively during data analysis, when ambiguities or inconsistencies became apparent for specific variables or subsets of studies. The full codebook, including definitions, decision rules, and data entry instructions, is provided in the SI to ensure transparency and reproducibility of the data extraction process

2.5 Literature overview

First mentions of the concept of EW can be traced back to the early 1990s.[13,46] However, systematic discussion in the context of CDR only began in the late 2000s and early 2010s, making EW a comparatively recent CDR pathway relative to more established land-based options. Based on the peer-reviewed CDR literature identified through the systematic review process, we identify 241 EW-focused peer-reviewed studies published between 1995 and 2025 (Figure 2a). We note that this is smaller than some other estimates[28] because we applied focused inclusion and exclusion criteria, followed by additional manual screening, to exclude false positives and identify studies most relevant to soil-based EW for CDR. Publication activity remained very limited until the mid-2010s, followed by a marked acceleration after 2017 and particularly rapid growth since 2020. Annual publication counts increased from fewer than 15 studies per year prior to 2021 to 50 studies in 2024, reflecting growing scientific and policy interest in EW as a scalable CDR option. The low count in 2025 reflects the April cut-off date for inclusion.

Geographically, judging by first author location, EW research is currently dominated by institutions based in the Global North (Figure 2b). First authors are most frequently affiliated with institutions in the United Kingdom (27%), the United States (18%), and Germany (10%), followed by Canada (8%) and China (7%). In contrast, large parts of Africa, South America, and Southeast

Asia are sparsely represented in terms of first-author affiliations (Figure 2c). At the same time, the geographic distribution of experimental and trial locations reveals a more nuanced pattern (Figure 2c). While authorship is concentrated in Europe and North America, many field trials—particularly those investigating silicate rock flour as a fertiliser—are conducted in the Global South, including Brazil, parts of Southeast Asia, and Africa. This divergence highlights a structural imbalance between where EW research is led and where on-the-ground experimentation and agronomic application are taking place.

The literature on silicate rock flour used as a fertiliser (i.e., without explicit CDR framing) shows a much stronger representation from countries in the Global South (Brazil 19%, China 9%, Egypt 5%, Indonesia 5%). Brazil, with a strong focus on using rock flour as an agricultural amendment[47,48], accounts for nearly one fifth of fertiliser-focused studies, followed by contributions from the United States, China, the United Kingdom, Australia, Egypt, Indonesia, and several other regions with strong agricultural research programs. These studies form an important complementary evidence base for understanding agronomic responses and potential side effects of silicate amendments, even where CDR is not the primary research objective.

In terms of research approaches, the EW literature spans a wide range of study types (Figure 2d). Experimental studies conducted under laboratory or mesocosm conditions represent the largest single category (23%), closely followed by modelling studies (21%). Reviews and perspective or opinion pieces together account for over a quarter of the literature, reflecting both the rapid expansion of the field and ongoing efforts to synthesize emerging evidence. Field experiments make up ~10% of studies, underscoring that EW remains at an early stage of real-world testing relative to its proposed scale of deployment. Smaller but important fractions of the literature address LCA, integrated assessment modelling, data analysis, natural analogue studies, surveys, and qualitative research, illustrating the growing diversification of EW research beyond purely geochemical questions.

With respect to content, the majority of peer-reviewed EW studies focus primarily on quantifying CDR through EW, including assessments of weathering rates, alkalinity generation, and carbon budgets (65%; Figure 2e). Environmental side effects, technological and logistical considerations, and agronomic impacts each account for smaller but significant shares of the literature. Topics such as public perception, governance and policy, equity and ethics, carbon losses and lag times, and resource availability remain comparatively under-represented, although several of these themes have gained traction in recent years. Overall, the literature reveals a field that is expanding rapidly in volume and disciplinary breadth, but remains unevenly distributed across geographies, research methods, and thematic priorities—highlighting both the progress made and the key gaps that motivate a systematic synthesis of the available evidence.

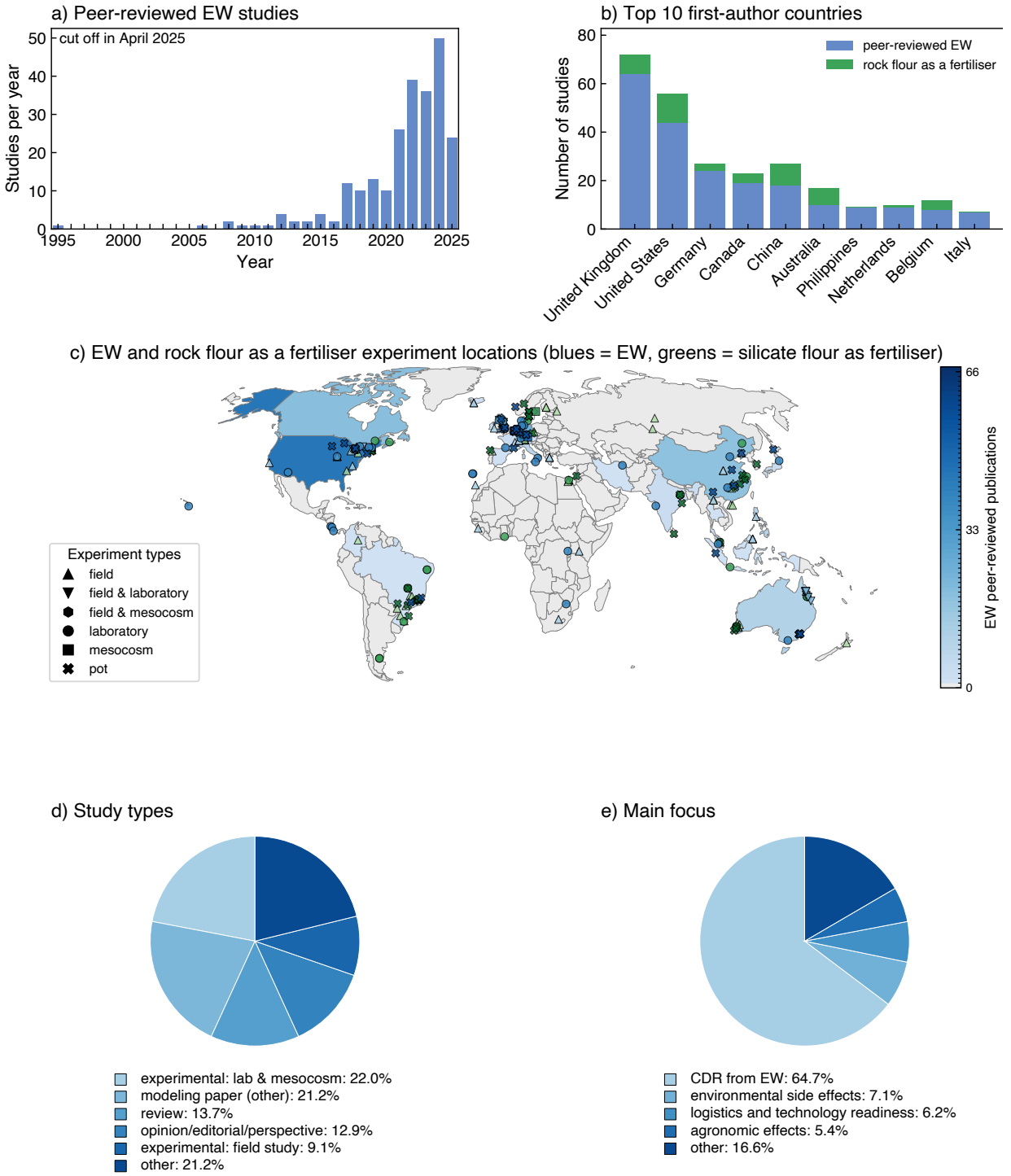

Figure 2: Overview of the Enhanced Weathering (EW) literature. (a) Annual number of peer-reviewed EW publications, showing rapid growth since the late 2010s. (b) Geographic distribution of first-author affiliations, highlighting a strong concentration in Europe and North America. (c) Comparison of first-author affiliations and experimental or trial locations, illustrating a divergence between leadership of EW research and the geographic distribution of field trials. (d) Distribution of EW peer-reviewed studies by primary study type. (e) Distribution of EW peer-reviewed studies by main focus.

2.6 Calculations and data processing

Several of the figures presented in this review summarise treatment effects associated with EW interventions across the literature landscape, expressed as observed changes in geochemical or agronomic parameters relative to appropriate reference conditions. Depending on the variable and study design, treatment effects are reported either as absolute differences (Δ)—for example, changes in soil pH—or as logarithmic response ratios, calculated as $\log_{10}$(treatment/control). Absolute differences are used where direct interpretation of magnitude is meaningful. In contrast, log response ratios are widely applied in systematic reviews and meta-analyses because they provide a dimensionless, scale-independent measure of proportional change, are symmetric around zero, and tend to stabilise variance across heterogeneous studies, facilitating comparison across experiments with differing baselines, units, and magnitudes.[49,50] A positive $\log_{10}$(treatment/control) value indicates an increase relative to the control, while negative values indicate a decrease. A value of 1 (-1) corresponds to an increase (decrease) by a factor of 10.

Where data availability permitted, treatment effects were calculated relative to both concurrent control treatments and baseline conditions, defined as measurements at the treatment site prior to EW application (t = 0). Control-based assessment contrasts differences between treated and untreated systems under otherwise similar conditions, whereas baseline-based assessment reflects temporal changes within treated systems relative to pre-treatment measurements; both types of treatment effects are included in the compilation figures.

3 Enhanced Weathering overview

This section introduces the fundamental principles underlying EW and the factors that regulate its performance across environmental settings. We begin with overviews of co-benefits and broader system impacts compared to other CDR approaches, the mechanistic basis of EW reactions, and distinctions between inorganic and organic carbon pathways (Section 3.1). We then examine the physicochemical controls on EW efficiency, focusing on how feedstock and mineral properties, grain size, soil chemistry, climate, and hydrology interact to govern weathering rates (Section 3.2).

Next, we consider the transport processes linking terrestrial weathering to downstream carbon storage, and the implications for long-term CO_2 sequestration in EW systems (Section 3.3). We then outline the principal pathways that can reduce net CDR, including both carbon losses and limitations on primary weathering (Section 3.4). The section concludes with an overview of environmental and operational constraints (Section 3.5), followed by a discussion of the modelling approaches used to quantify EW performance and scalability (Section 3.6).

3.1 Basic principles of EW

Natural silicate weathering is a fundamental component of the long-term carbon cycle that removes CO_2 from the atmosphere through reactions between CO_2, water, and silicate minerals. These reactions generate dissolved alkalinity and ultimately transfer carbon from the atmosphere into stable inorganic reservoirs, including dissolved bicarbonate in the ocean and solid carbonate minerals.[51,52] As such, this process regulates soil formation, controls the chemical composition of rivers, and governs the transfer of dissolved elements from continents to the oceans, thereby shaping global ocean chemistry and salinity.[52,53] Over geological timescales, this process acts as a stabilising climate feedback because weathering rates increase with temperature[54–59] and runoff[54–57,59], thereby enhancing CO_2 consumption under warmer and wetter conditions. This negative feedback mechanism has long been described as a planetary thermostat regulating global temperatures and atmospheric CO_2 concentrations.[60,61]

Enhanced Weathering seeks to accelerate this naturally slow process by increasing mineral reactivity and optimizing environmental conditions for dissolution.[13–26] This is typically achieved by mechanically grinding, crushing, or pulverising (i.e., comminuting) suitable silicate rocks to increase reactive surface area and applying the resulting material to soils using existing infrastructure for fertiliser or lime (calcium carbonate) application. Here, elevated soil CO_2 concentrations generated by root respiration and microbial activity under often acidic conditions enhance mineral dissolution kinetics.[15,19,62,63] Nevertheless, smaller particle size does not universally translate into faster weathering under field conditions because hydrological transport, solution saturation, and soil physical structure can limit reaction progress even when surface area is high;[63,64] see Section 3.2 for more details. Moreover, because comminution is energy-intensive, optimal grain size reflects a trade-off between dissolution kinetics and life cycle energy demand rather than a simple preference for the finest material.[18,19,23]

Carbon Dioxide Removal through EW occurs through a sequence of coupled geochemical processes, each of which is introduced in more detail in the following text sections. First, CO_2

dissolves in soil water to form carbonic acid, which supplies protons that drive mineral dissolution and release base cations such as Ca^{2+} and Mg^{2+} (see Section 3.1.2). This reaction generates DIC primarily in the form of bicarbonate, representing the main pathway for atmospheric CO_2 consumption during silicate weathering.[15,19] Second, these dissolved weathering products are transported in solution through soils and aquatic systems.[30,65–67] Hydrological pathways and geochemical processes in soils determine whether alkalinity and cations are retained locally, exchanged on soil surfaces, precipitated as secondary minerals, or exported to rivers and groundwater. The efficiency of this transport step strongly influences the fraction of captured carbon that ultimately contributes to durable CDR rather than being re-released locally.[14,19,68] Third, dissolved alkalinity exported to the ocean represents a long-lived carbon reservoir because ocean residence times of bicarbonate and carbonate species span millennia.[37] Alternatively, cations released during weathering may precipitate as solid carbonates in soils or sediments, forming inorganic carbon pools with variable stability. While such mineral precipitation can represent long-term storage, it also causes some of the carbon initially captured by the weathering reaction to be re-released as CO_2.[15]

Most EW studies and deployments have focused on croplands and other managed agricultural systems because these environments already possess the infrastructure, machinery, and logistical networks required for distributing finely ground materials at scale, and because mineral amendments can provide agronomic co-benefits that facilitate adoption.[15,19,22,69] Agricultural soils therefore represent the primary near-term deployment context in much of the literature, as spreading technologies developed for lime or fertiliser application can be readily adapted for silicate materials. Forest ecosystems have also attracted increasing attention as potential application environments, including earlier work motivated by soil amelioration, nutrient replacement for nutrient removal due to harvest and export, and mitigation of soil acidification.[64,70–74] However, implementation in such systems is generally considered more operationally complex and costly due to access constraints and heterogeneous terrain. Marginal or degraded lands have likewise been proposed as candidate sites where mineral amendments could contribute simultaneously to CDR and soil restoration, particularly in contexts where conventional agricultural productivity is limited.[75–77] However, with current technologies, deployment in both forested and marginal landscapes will face logistical challenges related to transport, application efficiency, and monitoring, which may constrain feasibility relative to intensively managed agricultural systems.

3.1.1 Co-benefit profile of EW in comparison to other CDR approaches

As a geochemical CDR strategy that stores carbon predominantly in stable inorganic reservoirs (section 3.5), EW occupies a distinctive position within the portfolio of proposed CDR approaches.[2,6–8,78] In comparison to chemical CDR technologies such as DACCS, EW generally exhibits lower operational energy requirements per unit CO_2 removed because its core reactions are thermodynamically spontaneous and are driven by natural gradients rather than requiring energy-intensive separation processes.[7,18,19] Consequently, EW deployment does not intrinsically depend on large, dedicated renewable-energy inputs, although access to low-carbon energy can substantially improve life cycle emissions associated with material processing and transport (cf., Section 8).[7,18,19]

Relative to most biologically mediated CDR approaches, the storage resulting from silicate weathering is expected to be more durable because it resides primarily in dissolved alkalinity or mineral phases rather than living biomass or SOM pools, which can be rapidly remobilised under changing environmental conditions.[7,15] Biological sinks such as forests and soil carbon are vulnerable to disturbances and climatic feedbacks that can reduce long-term storage efficiency—including fire, decomposition, and warming-induced respiration increases[79–81]—whereas inorganic weathering products are not subject to the same ecological sensitivities. Beyond this high-level comparison, synergies and potential antagonisms between EW and other CDR approaches are explored in detail in Section 9.

Enhanced Weathering also differs from many bio-based CDR strategies in that it does not require dedicated biomass production and therefore does not inherently compete for biomass associated with those approaches, or for land needed for food production or ecosystem conservation.[7,82] Instead, silicate feedstock can be applied directly to already managed lands using existing infrastructure (see also Section 12 on technology readiness), particularly agricultural soils, where they can improve soil chemical conditions through pH buffering, supply essential plant nutrients such as Ca, Mg, K, and Si, and enhance soil fertility.[15,22,47] These agronomic effects can translate into increased crop productivity, improved nutrient-use efficiency, and greater resilience to soil degradation and acidification.[19,20,22] Accordingly, mineral soil amendments of this type have been used in agricultural practice for decades to centuries.[40,47] Rather than competing for biomass, EW may therefore enhance biomass production, creating opportunities for synergies with biomass-based CDR approaches (see Section 9).

Beyond farm-scale benefits, partial substitution or supplementation of conventional fertilisers through mineral amendments may reduce GHG emissions associated with fertiliser manufacture and application. Such additions may also influence soil chemical and microbial processes in ways that suppress non-CO_2 GHG emissions such as N_2O.[22,83] EW can additionally provide broader environmental services, including mitigation of soil acidification, replenishment of weathering-derived nutrients lost through harvest and erosion, and reductions in nutrient runoff via improved nutrient-use efficiency.[15,19,22] Collectively, these characteristics position EW as a multifunctional land-management intervention capable of delivering climate mitigation alongside tangible agricultural, environmental, and societal co-benefits that are aligned with multiple Sustainable Development Goals (SDGs),[19,84] thereby distinguishing it from CDR approaches that provide CDR as their sole primary outcome. These co-benefits are examined in greater detail in Section 7, where the empirical evidence base and the magnitude of reported effects are assessed.

Taken together, this combination of durable carbon storage, comparatively modest energy requirements, limited resource competition, and clear agricultural co-benefits gives EW a distinctive profile within diversified CDR portfolios. These attributes suggest that EW may be deployable not only as a climate mitigation measure but also as a land-management strategy with independent agronomic value, which could facilitate implementation and stakeholder acceptance even outside explicit carbon-removal policy frameworks (cf., Sections 7 and 10).

3.1.2 Geochemical reactions driving CDR from EW

The CDR potential of EW arises from a sequence of aqueous geochemical reactions that govern mineral dissolution, alkalinity generation, and carbon storage; a comprehensive treatment of these reactions can be found elsewhere.[15] Briefly, CO_2 derived from soil respiration reacts with water to form carbonic acid:

$$CO_2 + H_2O \rightleftharpoons H_2CO_3 \rightleftharpoons H^+ + HCO_3^- \quad (1)$$

In terrestrial environments, CO_2 derived from soil respiration is typically present at higher partial pressures than in the atmosphere, further driving this reaction forward. Carbonic acid is a weak acid with a pK_{a1} of 6.35 in aqueous solution.[85] Under most common soil pHs, carbonic acid is partially or completely dissociated, releasing protons and bicarbonate:

$$H_2CO_3 \leftrightharpoons HCO_3^- + H^+ \leftrightharpoons CO_3^{2-} + 2H^+ \quad (2)$$

Protons—from this or other sources—drive mineral dissolution reactions that consume protons and liberate base cations capable of charge-balancing bicarbonate or carbonate anions. These reactions ultimately lead to two principal inorganic carbon storage states discussed in the literature: carbon as dissolved carbonate alkalinity (primarily HCO_3^- and CO_3^{2-}) and carbon in solid carbonate minerals.[14,15,19,24] For example, the dissolution of olivine may lead to alkalinity production due to an increase in HCO_3^- counterbalanced by cations (i.e., Mg^{2+}):

$$Mg_2SiO_4\,(s) + 4\,CO_2 + 4\,H_2O \rightarrow 2\,Mg^{2+} + 4\,HCO_3^- + H_4SiO_4 \quad (3)$$

Here, CO_2 is converted into dissolved bicarbonate, which constitutes the principal pathway for long-term atmospheric CO_2 consumption via silicate weathering.[15] Similarly, dissolution of wollastonite produces alkalinity but may also be followed by secondary carbonate precipitation:

$$CaSiO_3\,(s) + 2\,CO_2 + 3\,H_2O \rightarrow Ca^{2+} + 2\,HCO_3^- + H_4SiO_4 \rightarrow CaCO_3(s) + CO2 + H_2O + H_4SiO_4 \quad (4)$$

In such cases, precipitation of carbonate minerals can release a fraction of the CO_2 initially consumed during dissolution, meaning that alkalinity storage and solid phase carbonate storage differ in their net CDR efficiency depending on reaction pathways and environmental conditions.[15] The dissolved silica produced during silicate dissolution, represented here as silicic acid (H_4SiO_4), often does not fully leave the reaction site because it can reprecipitate as secondary phases or coatings on mineral surfaces,[63,86] limiting downstream silica export.

In contrast to silicates, carbonate mineral weathering does not generally produce net long-term CDR when followed by carbonate reprecipitation. Dissolution of calcite, for example,

$$CaCO_3 + CO_2 + H_2O \rightarrow Ca^{2+} + 2\,HCO_3^- \quad (5)$$

can temporarily increase alkalinity, but if carbonate subsequently precipitates, CO_2 is released again—making carbonate weathering alone a transient rather than permanent atmospheric CO_2 sink.[15] However, if secondary carbonate formation is delayed until after dissolved alkalinity has

been transported to the ocean on long timescales, this pathway can still contribute effectively to CDR,[30,31,87] albeit with a higher likelihood of reduced net uptake in systems where strong-acid inputs drive dissolution without equivalent atmospheric CO_2 consumption.

Beyond its impact on the CDR balance of carbonate weathering, the presence of strong acids (in particular nitric acid derived from fertilizer application and atmospheric deposition of anthropogenic nitric and sulfuric acid) can further modify these EW reactions because protons derived from strong acids can drive mineral dissolution without consuming atmospheric CO_2, thereby reducing net CDR efficiency. The thermodynamic and conceptual framework underlying alkalinity generation and acid-base system behaviour is treated in detail elsewhere,[85,88–90] and the implications of strong-acid weathering for EW carbon balances are discussed in more detail in Section 3.4.3.

3.1.3 Inorganic vs. organic CDR from EW

The reactions described above represent only the inorganic geochemical component of CDR associated with EW. However, mineral amendments interact with biological processes and nutrient cycling, which can modify ecosystem carbon dynamics and thereby influence net carbon balances beyond the purely abiotic weathering signal.[70,75,84,91–111] These coupled soil–plant–microbe interactions can regulate both mineral dissolution and carbon storage, indicating that biological responses are an important component of the overall carbon-cycle impact of EW in terrestrial systems.

Enhanced Weathering has the potential to impact SOM stocks in opposing directions. Increases in soil pH and nutrient availability following mineral additions may stimulate microbial activity and accelerate decomposition of labile organic matter, potentially producing short-term increases in respiration and transient carbon losses or other changes to SOM pools.[112–114] Evidence from the literature on liming shows that such responses can occur shortly after amendment, particularly in acidic soils or systems where microbial activity is nutrient-limited [42,43]. Conversely, mineral inputs can also enhance stabilisation of organic carbon through increased mineral–organic associations and improved plant productivity that raises organic carbon inputs to soils—processes widely recognised as key controls on long-term soil carbon persistence.[75,94] Liming literature indicates that this latter process often dominates in the medium to long term, though there is considerable variability both within and between studies.[42,43]

If EW positively alters soil organic carbon (SOC) stocks, the resulting impact on ecosystem carbon balance could equal or exceed the magnitude of inorganic CDR.[75,94] Such outcomes would represent an important additional climate benefit and highlight the potential of mineral amendments as tools for improving soil carbon management. At the same time, organic-carbon responses differ fundamentally from inorganic EW removal in durability and accounting characteristics. Soil organic carbon typically turns over on decadal to centennial timescales and remains sensitive to disturbance, land-use change, and climatic variability,[81,115] whereas carbon stored through alkalinity generation and subsequent ocean storage can persist for millennia.[15,37] Because persistence, reversibility risk, and monitoring approaches differ so strongly between organic and inorganic processes and carbon pools, combining resulting carbon impacts into a

single CDR estimate would obscure their distinct temporal characteristics and could lead to inconsistent or non-comparable crediting across CDR approaches.

For this reason, CDR during EW in this review refers exclusively to carbon removal arising from inorganic weathering processes. Potential effects of EW on SOM are not disregarded but are instead treated as complementary outcomes whose magnitude, direction, and variability must be quantified empirically and independently. Accordingly, responses of SOC observed across the compiled EW studies are assessed separately in Section 7.4.2.

3.2 Factors affecting EW efficiency

The efficiency with which EW generates alkalinity and climate-relevant CDR depends on a suite of interacting physical, chemical, and biological controls that regulate mineral dissolution and the subsequent fate of weathering products. Across environments, dissolution rates translate to CDR potential and are governed jointly by intrinsic mineral properties (e.g., mineralogy, surface area, and application amount) and extrinsic environmental conditions, including soil chemistry, hydrology, and climate (see Figure 3).[62] These controls are highly coupled and operate across spatial scales—from mineral surfaces to catchments—often interacting nonlinearly such that weathering responses cannot be predicted from any single factor in isolation.[116]

A useful conceptual starting point for interpreting EW is the distinction between supply-limited and weathering-limited behaviour in natural weathering systems. In natural settings with low erosion or mineral replenishment rates, chemical weathering fluxes can be constrained by the supply of fresh reactive minerals to the weathering zone, such that fluxes scale with material supply and cannot exceed it.[59] Where fresh mineral surfaces are abundant, by contrast, weathering fluxes are controlled more strongly by reaction kinetics and transport processes, including the delivery of undersaturated water to mineral surfaces and the removal of dissolved products from porewaters.[59] EW can be understood as an intentional shift along this continuum: by adding finely ground silicates, rock flour applications aim to move systems away from supply limitation toward weathering-limited behaviour, where dissolution rates are governed primarily by kinetics and transport rather than by scarcity of reactive surface area. Once reactive feedstock is present, realised dissolution and CDR may instead be governed by kinetic and transport limitations, including soil hydrology, climate, porewater residence time, saturation state, mineral accessibility, secondary mineral formation, and surface passivation.

The following subsections synthesize current knowledge on the dominant factors controlling EW efficiency and highlight major sources of uncertainty that complicate prediction across sites and timescales. Separated into primary feedstock characteristics, soil geochemistry, and hydrology/climate, this section provides a high-level overview of how these processes regulate EW outcomes while also identifying key conceptual, methodological, and reporting challenges in the current literature. Emphasis is placed on the coupled roles of feedstock characteristics, grain size and surface area, soil chemical environment, and hydroclimatic conditions, as well as on how their interactions shape both dissolution dynamics and downstream alkalinity export.

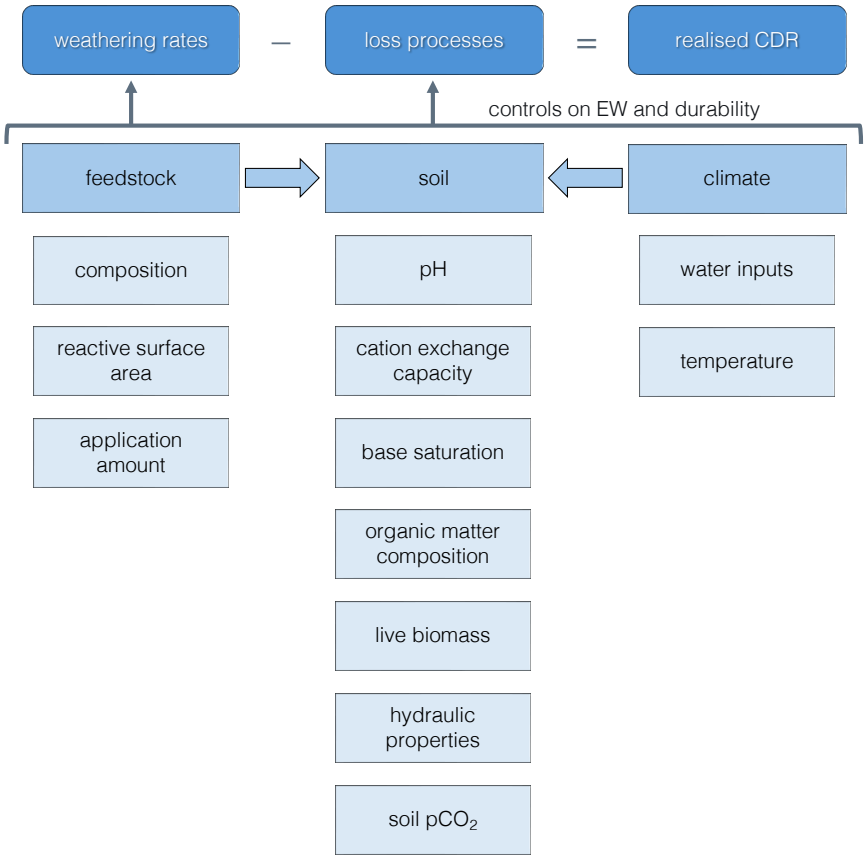

Figure 3: Overview of fundamental parameters that determine both weathering rates and secondary loss processes that determine realised Carbon Dioxide Removal (CDR) through Enhanced Weathering (EW). These parameters—and associated processes—are highly coupled and dynamic, confounding interpretation of their individual effects (see also Figure 22).

3.2.1 Feedstock properties and application

3.2.1.1 Feedstock composition

Feedstock type and composition is a primary determinant of EW performance because mineral composition governs dissolution kinetics, theoretical CDR potential, and the release of both beneficial and potentially harmful elements into soils.[15,62] This section reviews how feedstock properties shape outcomes, covering composition and major feedstock classes, mineralogical controls on weathering rates, compositional controls on CDR potential, materials used in existing studies, commercial availability, pollution risks, and key reporting and comparability challenges in the literature.

644 3.2.1.1.1 Mineralogical control on weathering rates
645 Weathering rates of silicate minerals vary systematically as a function of both thermodynamic
646 stability and dissolution kinetics, making mineralogy a primary control on EW performance.[117–
647 121] This relationship is classically described by the Goldich stability series, in which minerals that
648 crystallize at high temperature (e.g., olivine, pyroxenes, Ca-plagioclase) are typically least stable
649 at Earth's surface and weather most rapidly, whereas low-temperature minerals such as K-feldspar
650 are more resistant. Consistent with this framework, the major base-cation–bearing minerals that
651 dominate mafic and ultramafic rocks—particularly olivine (forsterite Mg_2SiO_4–fayalite Fe_2SiO_4),
652 pyroxenes (e.g., diopside $CaMgSi_2O_6$), and plagioclase feldspars (albite, $NaAlSi_3O_8$–anorthite,
653 $CaAl_2Si_2O_8$)—span orders-of-magnitude differences in dissolution rates under comparable
654 conditions.[119,120,122,123]
655
656 Silicate minerals commonly form solid-solution series in which elemental substitutions produce
657 continuous compositional variation within a fixed crystal structure.[124] These compositions are
658 conventionally expressed as proportions of end members—for example, olivine compositions are
659 reported as Fo_x (percentage forsterite component), while plagioclase compositions are expressed
660 as An_x (percentage anorthite component), reflecting coupled substitutions that preserve charge
661 balance. Such compositional variability may occur even within single grains (zoning), reflecting
662 evolving magma chemistry during crystallization and leading to grain-scale heterogeneity in
663 reactivity.[124] Mineral proportions also vary among rock types: dunite consists of ~90–100%
664 olivine, whereas basalt typically contains mixtures of pyroxene and plagioclase with variable
665 olivine (often zero). These differences translate directly into differences in weathering behaviour
666 and CDR potential among nominally similar lithologies, with measured basalts showing up to six-
667 fold variation in modelled CDR due largely to variations in reactive mineral content such as olivine
668 and augite.[125] Additional variability arises from the proportion of amorphous phases (e.g., glass),
669 which can dissolve as fast as or faster than crystalline silicates depending on composition and pH,
670 producing especially rapid early-stage weathering.[123]
671
672 Accessory minerals may further influence reactivity and environmental behaviour. Mafic and
673 ultramafic rocks commonly contain Fe–Ti oxides, and olivine-rich lithologies such as dunite often
674 contain chromite ($FeCr_2O_4$). Chromite is resistant to chemical weathering and tends to persist as a
675 residual mineral that can be concentrated by sedimentary processes, illustrating that not all mineral
676 phases contribute equally to dissolution reactions. Beyond igneous feedstocks, the Ca-silicate
677 mineral wollastonite ($CaSiO_3$), typically formed by contact metamorphism of impure limestones,
678 is also of interest for EW because of its rapid dissolution rate (log rate = −6.97 mol m^{-2} s^{-1}),
679 comparable to that of forsterite (log rate = −7.16 mol m^{-2} s^{-1}).[118,124]
680
681 3.2.1.1.2 Feedstock CDR potential
682 The maximum theoretical CDR capacity of a feedstock is determined by its bulk elemental
683 composition, because the availability of reactive base cations governs how much CO_2 can
684 potentially be converted into bicarbonate or carbonate during weathering. This potential is
685 commonly estimated using the modified Steinour formulation, which calculates stoichiometric
686 CO_2 uptake directly from oxide concentrations (e.g., CaO, MgO, Na_2O, K_2O) while accounting
687 for species that reduce effective capacity, such as sulfur- or phosphorus-bearing phases.[18,37,126]
688 Because the equation depends explicitly on bulk chemistry, feedstocks richer in Ca- and Mg-
689 bearing silicates exhibit higher theoretical CDR capacities, whereas materials dominated by less

reactive or non-alkalinity-generating components yield lower values. For EW, the Steinour-based formulation to estimate a feedstock's CDR potential can be written as:

$$CDR_{pot} = \frac{M_{CO_2}}{100}\left(\alpha\frac{CaO}{M_{CaO}} + \beta\frac{MgO}{M_{MgO}} + \varepsilon\frac{Na_2O}{M_{Na_2O}} + \theta\frac{K_2O}{M_{K_2O}} + \gamma\frac{SO_3}{M_{SO_3}} + \delta\frac{P_2O_5}{M_{P_2O_5}}\right)10^3\eta \qquad 6$$

where oxide concentrations are expressed in weight percent, M_x denotes molar masses, coefficients (α, β, ε, θ, γ, and δ) represent the relative alkalinity contribution of each oxide (α, β, ε, θ are 1 for typical surface water pH ranges, while γ and δ are negative representing reduction in CDR potential), and η is the molar CO_2-to-divalent-cation ratio. While idealized silicate weathering implies η = 2, buffering in the carbonate system reduces this to ~1.4–1.7 under typical seawater conditions; a value of 1.5 is commonly adopted as a conservative global average that accounts for thermodynamically driven degassing of CO_2 in the oceans.[126] The same formulation shows explicitly that sulfur- and phosphorus-bearing phases (e.g., apatite, $Ca_5(PO_4)_3(OH)$) reduce effective CDR capacity because their dissolution either does not consume CO_2 or may generate acidity that releases it.[126]

Where feedstocks contain carbonate minerals, additional care is required because carbonate dissolution generates alkalinity with different atmospheric CO_2 uptake stoichiometry: dissolution of $CaCO_3$ by carbonic acid produces two equivalents of alkalinity, but only one of the two DIC equivalents is derived from atmospheric CO_2, while the other originates from the carbonate mineral lattice. Consequently, carbonate-derived CaO and MgO should not be treated identically to silicate-derived oxide contributions when estimating theoretical CDR capacity. Carbonate-derived Ca and Mg correspond to only half the atmospheric CO_2 uptake implied by the same divalent cation release from silicate minerals. Where nominally silicate feedstocks contain trace or minor carbonate phases, this contribution to CaO and MgO concentrations should therefore be quantified separately or corrected for.

Because this formulation is purely based on rock composition, it defines a theoretical upper bound on CDR set by feedstock chemistry alone under the implicit assumption that all weathering occurs with carbonic acid. Differences in CaO and MgO content explain large contrasts in potential between lithologies (e.g., dunite versus basalt)[15,23] and substantial variability among basalts due to differing proportions of reactive minerals.[125,127] Mineral dissolution kinetics, together with negative feedbacks in soils, will then determine how quickly a given feedstock CDR potential is realized via weathering.

3.2.1.1.3 Rocks used in EW studies

The literature shows a clear preference for mafic and ultramafic silicate rocks in CDR-focused EW (Figure 4), reflecting their favourable balance of reactivity, availability, and chemical composition. Basalt is the single most widely used feedstock (n = 86), largely because it is abundant, widely quarried, and contains reactive Ca- and Mg-bearing minerals that support alkalinity generation and nutrient release.[15,128] Nevertheless, basalts vary substantially in weathering rate and CDR potential depending on mineralogy and glass content, meaning that not all basalts perform equally.[125,127]

Olivine-rich ultramafic rocks are often considered among the most effective feedstocks because they combine high theoretical CDR capacity with rapid dissolution kinetics.[23,122] However, their geographic availability is more limited and they can contain elevated trace metal concentrations that require additional risk assessment.[129,130] Wollastonite represents a highly reactive Ca-silicate alternative with fast dissolution and typically low heavy-metal contents, though deposits are comparatively scarce and some sources may contain asbestos phases that must be screened.[131]

Carbonate materials, while abundant, rapidly dissolving, and widely used for soil pH management,[29] differ fundamentally from silicates in their atmospheric CO_2 uptake stoichiometry. Carbonic-acid dissolution of carbonates generates only half as much atmospheric CO_2 uptake per Ca or Mg released to solution as equivalent silicate dissolution, because one carbon equivalent originates from the carbonate mineral lattice. In the presence of strong acids, carbonate dissolution may release carbonate-derived CO_2 before longer-term alkalinity effects are realised. Although carbonate materials can be used in EW contexts, they are not a central focus of this review, reflecting both their distinct carbon-accounting considerations and the limited amount of work that has focused explicitly on carbonate-based EW to date.[30,31,87]

Industrial waste streams, such as mine tailings or concrete-derived materials, were included only where their application was discussed in the context of agricultural soil amendments. Consequently, the counts reported here do not include the broader literature on ambient weathering or carbonation of mine tailings, waste piles, and similar materials outside agricultural application contexts.

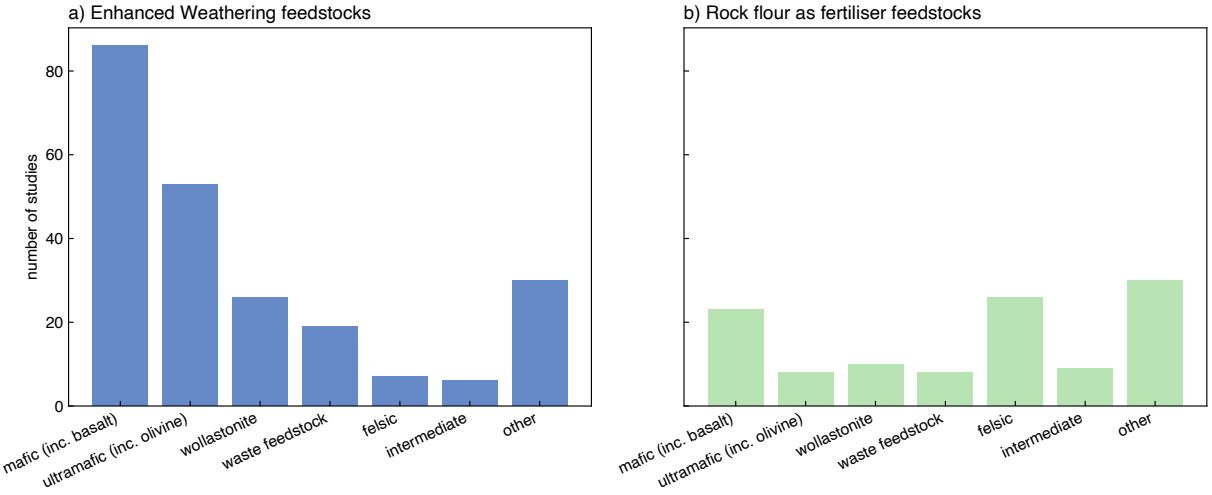

*Figure 4: Distribution of feedstock types used in Enhanced Weathering (EW) and rock flour studies. Counts show the number of unique studies reporting each feedstock class, separated into (a) EW and (b) agronomic rock flour applications. Mafic rocks—primarily basalt—are the most frequently used materials in EW studies (*n = *86), followed by ultramafic rocks such as olivine-rich lithologies (*n = *53), whereas fertiliser-focused studies more often employ felsic materials (*n = *26) alongside mafic rocks (*n = *23). Less commonly reported categories include wollastonite, industrial alkaline wastes, intermediate compositions, and other feedstocks. Counts reflect one entry per feedstock per study.*

Overall, feedstock choice reflects trade-offs among reactivity, theoretical CDR capacity, availability, agronomic co-benefits, and environmental constraints, and these trade-offs explain the dominance of basaltic materials in EW research (Figure 4). In contrast, agronomic rock flour studies show a broader distribution of feedstock types, including more felsic materials, reflecting goals centred on soil amendment and nutrient supply rather than maximising CDR. Note that this categorisation should be viewed as an approximate representation of the literature: it reflects both inconsistencies and misclassifications in the primary studies themselves (see Section 3.2.1.1.6 below),[132] as well as the necessary simplifications introduced here when mapping reported feedstocks into broad categories.

3.2.1.1.4 Commercial availability of main feedstock types

Assessing commercial availability is essential for evaluating the scalability of EW. Even feedstocks with high theoretical CDR potential are largely irrelevant on large scales if extraction volumes, permitting constraints, infrastructure, or competing markets limit supply. Thus, estimates of technical CDR potential must be interpreted in light of real-world production capacity and constraints on capacity expansion, reserve definitions, and market structure.[133]

Determining the availability of suitable rock types is, however, complicated by inconsistent nomenclature and reporting practices. The most accessible production statistics are compiled by the United States Geological Survey (USGS) National Minerals Information Center through the Mineral Commodity Summaries and Minerals Yearbook,[134] and by the British Geological Survey (BGS).[135] These sources provide annual production figures, and in some cases estimates of resources and reserves, but typically combine lithologies into broad industrial categories. Brazil and other countries also publish national statistics, which are consulted by the USGS and the BGS.

Care must be taken in interpreting these statistics, particularly regarding the terms ‘resource’, ‘reserve base’, and ‘reserves’, which have strict technical definitions established by the USGS.[136] A resource refers to a concentration of material for which economic extraction is currently or potentially feasible; the reserve base is the portion meeting defined physical and chemical criteria for mining; and reserves represent the economically extractable part of the reserve base at the time of determination. In investment contexts, reserve estimates are validated under internationally recognised reporting codes (e.g., CRIRSCO, NI-43-101), but such documentation is often not available in the public domain for quarrying operations that supply crushed rock for construction and industrial use. Consequently, data relevant to EW deployments are often inferred from aggregate production statistics rather than from CDR-specific resource assessments.

For basalt in particular, the distinction between geological abundance and economically accessible supply is critical. Basalt constitutes ~3.5% of the global continental crust,[128] but this figure reflects theoretical resource abundance rather than extractable reserves. Reserves are site-specific and constrained by economics and permitting. The concept of permitted reserves—the volume authorised for extraction under regulatory approval[137]—is especially relevant for deployment planning, because annual production cannot exceed permitted quantities. In the absence of explicit permitted-reserve data, annual production figures provide a conservative lower bound.[133] It is also important to recognise that reserve estimates may remain relatively constant over time, as companies continually reclassify proven resources into reserves.[138]

A further challenge arises from the combination of lithologies within industrial reporting categories. The USGS reports production under "crushed stone," combining limestone and dolomite (~70%), granite (~14%), sandstone (~6%), traprock (~6%), and other lithologies.[139] Similarly, the BGS combines igneous, metamorphic, and sedimentary rocks within "crushed rock" categories. Consequently, basalt-specific production is rarely reported directly. Nevertheless, disaggregated data from BGS sources can be interpreted to estimate the availability of basic rocks (basalt, dolerite, gabbro, microgabbro) in the UK,[133] highlighting both the feasibility and uncertainty of such reconstruction efforts. Commercial estimates of global basalt production address the construction market, including manufactured slabs and basalt fibre, and do not consider EW as a potential demand sector; consequently, projected market sizes do not account for additional demand from CDR deployment.[140] The global market value for basalt is reported as 2 billion USD,[140] which—when converted using the USGS average price for crushed rock (17.5 USD t^{-1})—corresponds to an implied production of ~115 Mt yr^{-1}. This back-of-the-envelope estimate is likely conservative, as the UK alone produces 14.8 Mt yr^{-1} for construction.[133]

Table 1: Summary of availability of rocks used for Carbon Dioxide Removal (CDR). United States Geological Survey (USGS) data from Mineral Commodity Summaries.[139,141] Traprock (assumed to include basalt) quantity based on 6% of annual production. Great Britain refers to England, Scotland and Wales; UK refers to the United Kingdom of England, Scotland, Wales and Northern Ireland. Wollastonite price data from ref. [142]. Norway olivine production data from ref. [143]; annual production capacity from Åheim is stated to be 2 million tonnes. Åheim reserves are stated to be 150 years production.

Rock type	Annual Production (Mt)	Nominal price (USD t^{-1})	Resources (Mt)	Reserves (Mt)	Permitted Reserves (Mt)	Source
Crushed Stone	1500	17.50	'plentiful'	-	-	[139]
'Traprock'	90	17.50	-	-	-	[139]
Crushed Rock (Europe)	1255	-	-	-	-	[144]
Crushed Rock (Great Britain)	109.7	-	-	4589	-	[145]
Basalt	114	17.5	-	-	-	Estimated from [140]
Basalt (UK)	14.8	-	-	-	490	[133]
Wollastonite	1.1-1.2	340-380	100	-	-	[141,142,144]
Dunite (olivine)	1.4 (Norway only)	-	-	150 years	-	[143]
Olivine	4-8	-	-	-	-	[146,147]

Table 1 summarises available production, reserve, and price data for key feedstocks relevant to EW. Crushed stone production in the United States alone exceeds 1.5 billion tonnes annually,[139] indicating that large-scale material handling infrastructure already exists. In contrast, wollastonite production is ~1–1.2 Mt yr^{-1} globally,[141,144] reflecting its specialised industrial use and limiting the scalability of the current product type and market for new uses in EW. Dunite/olivine production is similarly limited, with Norway reporting ~1.4 Mt yr^{-1} and long reserve lifetimes at specific

sites.[143] The relatively high nominal price of wollastonite compared to crushed rock reflects processing for high-value applications;[141] lower-grade material not suitable for fillers may be more appropriate for CDR deployment and is likely not reported in published production (or price) statistics.

Overall, these data illustrate that basaltic materials benefit from existing large-scale quarrying and distribution systems that support scalability, whereas high-reactivity endmembers such as olivine-rich dunite and wollastonite are more constrained by geographically limited production and specialised markets. Importantly, current statistical reporting frameworks were not designed with CDR in mind, and improved lithology-specific production reporting would significantly enhance assessments of EW scalability. Although these figures clearly show that human society already mobilises Gt-scale rock fluxes annually,[139] large-scale EW scenarios would need to operate within—or substantially expand—this existing extraction envelope. This implies that deployment trajectories will be governed as much by mining infrastructure, permitting timelines, and land-use constraints as by mineral reactivity or theoretical CDR capacity. Given how fundamental feedstock availability is for scaling EW, it is striking that such basic data as lithology-specific global production (e.g., basalt output) are rarely reported, a gap that complicates near-term deployment estimates and represents a clear priority for future data collection and resource assessments.

3.2.1.1.5 Pollution concerns

Trace elements present in silicate feedstocks—particularly Cr and Ni—are frequently discussed as potential risks for EW because they can accumulate in soils if added repeatedly at large application rates (cf., Section 3.5).[129,131] However, bulk concentrations alone are not sufficient to assess environmental hazard, as risk depends strongly on mineralogical host phases, dissolution kinetics, and resulting bioavailability. For example, chromium in mafic and ultramafic rocks is typically present as trivalent Cr(III) in chromite ($FeCr_2O_4$), whereas the toxic hexavalent form Cr(VI) is not a primary constituent of such minerals. Chromite is also resistant to weathering and commonly persists as a residual phase in soils derived from mechanically weathered parent materials. Ni behaves differently: it commonly substitutes for Mg in olivine and can therefore be released during dissolution of olivine-rich lithologies, a process reflected in the occurrence of metallophyte plant communities adapted to naturally Ni-rich substrates.[148] Both Cr and Ni can function as micronutrients at low concentrations, and Cr deficiency has been reported in human diets,[149] underscoring that toxicity depends on dose and speciation rather than simple presence.

Evaluation of potential contamination must also consider natural background concentrations in soils. Reported baseline levels vary widely depending on parent material.[150] For example, soils in eastern England average ~68.5 mg kg^{-1} Cr and ~22.5 mg kg^{-1} Ni, whereas soils developed on volcanic or metamorphic rocks in northern Britain can contain substantially higher concentrations.[151] By comparison, basalt feedstocks proposed for CDR have been reported to contain roughly 35–350 mg kg^{-1} Cr and 18–195 mg kg^{-1} Ni, values that fall within the lower portion of ranges observed in natural soils.[125,150] Moreover, groundwater drawn from basalt aquifers supplies millions of people in parts of Brazil without documented adverse health effects.[152] These comparisons highlight that natural geochemical variability is large and must be considered when interpreting feedstock composition.

Nonetheless, trace-element accumulation remains a legitimate concern under sustained application scenarios. Modelling indicates that repeated additions of feedstock could exceed regulatory thresholds for certain elements over decadal timescales depending on feedstock composition and soil properties, particularly where weathering releases metals faster than they are removed by leaching or plant uptake.[129] It is also important to recognise that existing regulatory limits for agricultural amendments were largely developed for relatively fast-dissolving materials (e.g., soluble fertilisers or liming products) and are typically based on short-term contaminant release fluxes.[129,131] EW feedstocks, in contrast, often dissolve more slowly and release trace elements over longer time horizons, meaning that both the temporal dynamics of release and cumulative loading need to be explicitly evaluated. In addition, pH increases or secondary phases may temporarily immobilise trace elements, with the potential for later remobilisation if soil conditions change (e.g., acidification following cessation of application), emphasizing the importance of long-term monitoring and risk assessment.

Beyond trace metals, mineralogical contaminants must also be considered. Asbestiform minerals may occur in some mafic and especially ultramafic rocks and pose inhalation hazards if present, as documented for residues from asbestos mining.[153] Screening for such phases is therefore standard practice in industrial mineral supply chains. In contrast, natural radioactivity is generally low in mafic and ultramafic rocks because their concentrations of K, U, and Th are typically much lower than those of felsic lithologies, which are more commonly associated with Rn emissions.[154]

Overall, available evidence suggests that potential contamination risks from EW feedstocks will be highly context-dependent. Risk will vary with mineralogy, weathering rate, soil chemistry, and background geochemistry, meaning that it cannot be inferred from composition alone. A balanced interpretation is therefore that some high-reactivity feedstocks may pose elevated trace-element concerns under certain conditions,[130] but these risks must be evaluated and managed through appropriate feedstock characterization, site selection, application limits, and monitoring strategies.[129,131]

3.2.1.1.6 Challenges in the EW literature

A persistent weakness in the EW literature is the inadequate and inconsistent characterization of feedstocks,[132] particularly mafic rocks described as "basalt," despite mineral composition being a primary control on both theoretical CDR capacity and reaction kinetics.[18,125,126] Classification schemes established by the International Union of Geological Sciences (IUGS) provide clear protocols for naming igneous rocks based on bulk chemistry,[155,156] yet these standards are not consistently applied. In the literature assessed here, only 33 publications use the terms basalt, dolerite (diabase), or gabbro for feedstocks that are texturally distinct but compositionally related mafic rocks. However, correct naming requires more than nominal similarity: it depends on analytically robust major-element data and adherence to established classification boundaries.

Evaluation of 56 published bulk chemical analyses from 33 publications in our database reveals substantial deficiencies in reporting and analytical closure. Only 34 analyses provide an explicit analytical total, and of these, just 16 fall within the acceptable 99–101% range (Figure 5a). Exact totals of 100% strongly suggest post-analytical normalisation without reporting that this has been done, precluding assessment of data quality and potentially biasing subsequent calculations of

weathering potential. When analytical totals are recalculated from reported oxide concentrations, values span 79.4–103.1% (Figure 5c), indicating either analytical problems, incomplete reporting (e.g., omission of volatile components), or mischaracterization of altered materials as pristine igneous rocks. Some of the lowest totals occur in materials described as basalt but known to contain substantial amounts of secondary hydrated minerals such as clinochlore,[157,158] demonstrating how alteration can confound both classification and CDR potential estimates.

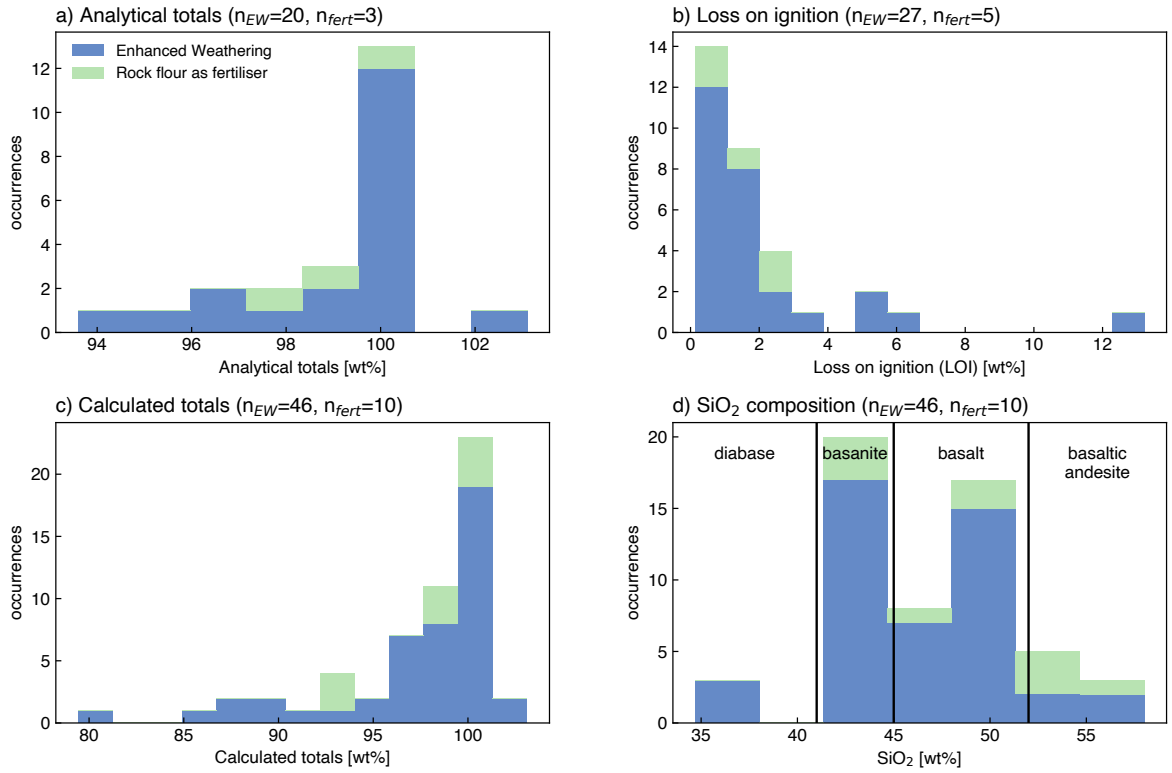

Figure 5: Summary of reported bulk geochemical properties of so-called basaltic feedstocks used in Enhanced Weathering (EW) and rock flour fertiliser studies. (a) Distribution of reported analytical totals from published compositional analyses. (b) Distribution of reported loss on ignition (LOI) values, reflecting volatile content and analytical closure. (c) Distribution of analytical totals recalculated from reported oxide concentrations. (d) Distribution of SiO_2 concentrations for feedstocks reported as basalt, with vertical lines indicating approximate compositional boundaries between diabase, basanite, basalt, and basaltic andesite fields. In all panels, histograms combine data from EW studies and agronomic rock flour applications, shown using consistent colour coding.

Loss on ignition (LOI) is particularly critical in this context because it quantifies volatile components, including structurally bound water and CO_2 in secondary minerals. The IUGS volcanic rock classification requires LOI values below 2 wt% for reliable normalisation and classification.[155] Yet only 32 of 56 analyses report LOI (Figure 5b), and reported values range up to 13.21 wt%, indicating significant alteration in some samples. Among EW studies, 20 analyses report LOI <2 wt%, but many do not, and in several cases LOI is entirely omitted. Elevated LOI values imply that Ca and Mg may be partly hosted in hydrated or carbonated phases rather than

primary silicates, necessitating correction before applying stoichiometric approaches such as the Steinour equation. Without such correction, CDR capacity can be systematically overestimated.

A particularly important case is the presence of calcite or other carbonate minerals in nominally basaltic feedstocks. Even trace carbonate phases can dissolve much faster than primary basaltic silicates and may dominate early Ca release in experiments or field deployments. Without explicit carbonate quantification, rapid initial Ca mobilisation could therefore be misinterpreted as fast basalt weathering, biasing estimates of silicate dissolution rates and theoretical CDR potential (cf., also section 3.2.1.1.1). Carbonate contents should therefore be quantified separately, for example through LOI, X-ray diffraction, or selective dissolution approaches, before applying stoichiometric CDR calculations to basaltic feedstocks.

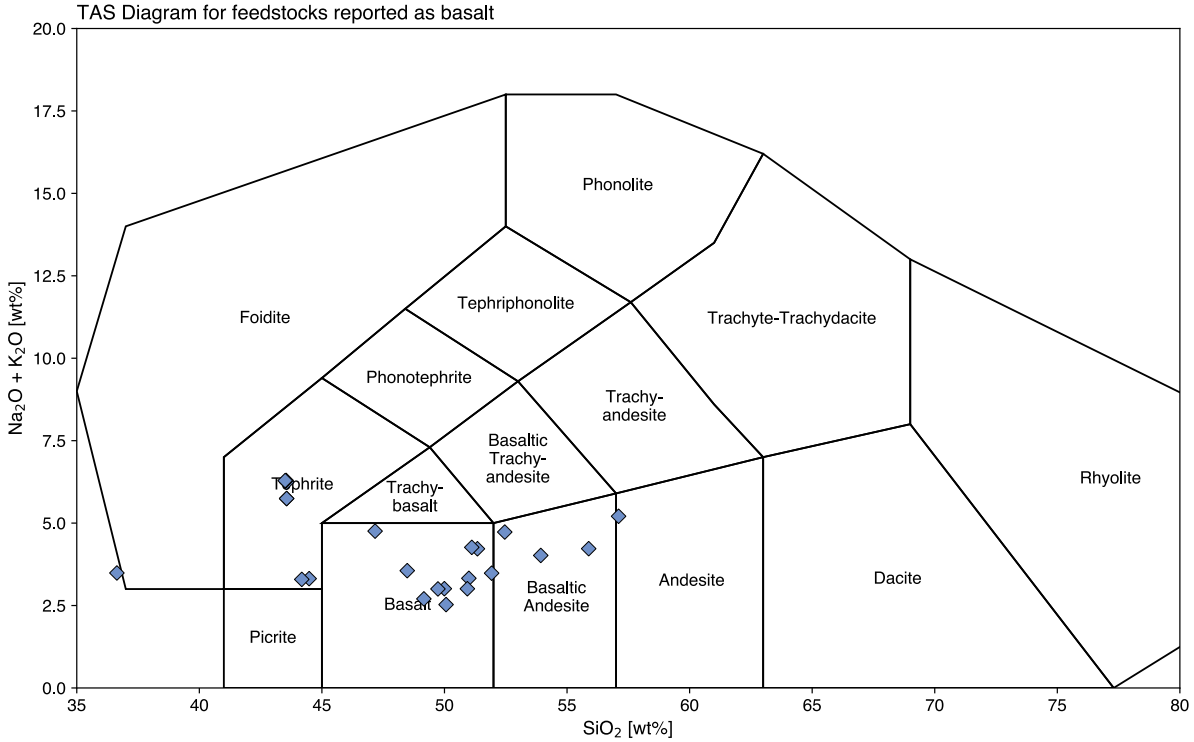

Figure 6: Total alkali–silica (TAS) diagram showing the compositions of samples reported as basalt with loss on ignition (LOI) < 2 wt%. Major-element data are plotted as $Na_2O + K_2O$ versus SiO_2 (wt%), and volcanic rock-type fields and boundaries follow the combined Le Maitre classification scheme as implemented in the pyrolite Python plotting template. Points denote individual analyses, and field labels indicate standard TAS rock names.

When analyses with LOI <2 wt% are normalised and plotted on a Total Alkali–Silica (TAS) diagram, most samples fall within basalt or basanite fields (Figure 5, Figure 6), indicating that classification is generally straightforward and appears to be often correct where adequate data exist. However, only 23 samples meet the minimum requirements for such classification and are plotted in Figure 6. For the remaining majority, available data preclude rigorous application of IUGS criteria, and the rocks cannot be named based on the information provided by the authors. Even if one considers SiO_2 content alone, as shown in Figure 5d, reported compositions cluster

across basaltic and basanitic ranges, with roughly equal numbers corresponding to basalt and basanite. Despite this, only a small fraction of studies explicitly identify their materials as basanite (e.g., ref. [159]), suggesting that formal classification boundaries are rarely considered. This inconsistency hampers comparability across studies and obscures potentially important differences in alkalinity content and weathering behaviour.

These issues extend beyond terminology. Recent critiques have highlighted that EW experiments frequently rely on a narrow set of commercially available materials, often selected for convenience and lacking sufficient petrographic characterisation.[132] Mineralogical information is commonly limited to bulk oxide chemistry or qualitative phase identification, whereas dissolution rates vary by orders of magnitude among minerals within the same compositional group.[123,160] Without detailed mineralogical and microtextural characterization, including identification of secondary phases and alteration features, experimental results cannot be reliably generalized or used to constrain reactive transport models (RTMs).[161]

Similar concerns apply to ultramafic feedstocks described as dunite. By definition, dunite should contain ≥90% olivine and minimal volatile content; however, reported LOI values as high as 6.48 wt% indicate substantial serpentinization or other secondary alteration.[162] From a petrological perspective, such materials cannot be considered pristine dunite, and their weathering behaviour will differ accordingly. Where artificial or industrial by-products are used, strict petrological nomenclature is less relevant, but rigorous chemical and mineralogical characterization remains essential to support quantitative modelling of CDR.

Taken together, Figure 5 and Figure 6 demonstrate that more than half of the published compositional datasets lack sufficient information for reliable classification under established IUGS standards. The required analytical approaches—complete major-element analyses, reporting of analytical totals and LOI, and formal classification using TAS criteria—are long established and straightforward. Without their consistent application, subsequent modelling, upscaling, and MRV efforts rest on uncertain geochemical foundations. Greater integration of petrological expertise into EW research is therefore essential to ensure that feedstocks are accurately described and that estimates of CDR potential are robust and comparable across studies. Beyond characterization, limited reporting on feedstock production pathways, accessible reserves, expansion constraints, permitting requirements, and realistic development timelines represents an additional potential bottleneck for assessing EW scalability.

3.2.1.2 Feedstock application amount

Feedstock application amount in EW sets the inventory of reactive mineral (and thus potential reactive surface area) available to dissolve and generate alkalinity, release base cations, and shift soil chemistry—often increasing shifts in pH, cation exchange capacity (CEC), nutrient availability, and CDR yields as dose increases.[75,163–167] However, these responses are frequently non-linear: (i) agronomic and soil physical benefits can show optima or diminishing returns (e.g., lower yields at very high doses; plateauing CEC/base cation responses beyond ~10–25 t ha^{-1}),[164,168–170] (ii) dissolution per unit applied can decline at higher doses—consistent with negative feedbacks in soils (e.g., saturation effects and/or secondary phase formation reducing weathering efficiency),[75,161,171,172] and (iii) higher doses can impair soil physical/hydrological function in some

contexts (e.g., pore clogging, reduced gas transport), while showing mixed or minimal effects in others.[173] Application rate also affects detectability and transfer of the weathering signal beyond soils (e.g., alkalinity transfer) and can influence whether field/mesocosm studies observe a measurable chemical weathering signal at all.[174,175]

Additionally, dose interacts with material and environment: experiments explicitly comparing low vs. high application rates (or multiple doses) show rate-dependent changes in weathering/nutrient uptake, soil inorganic carbon (SIC) accumulation, and sometimes phytotoxicity or contaminant mobilisation risks,[170,176–180] while models similarly suggest application rate can independently change both total CDR and CDR efficiency (CDR per mass of feedstock applied).[75] Finally, application amount also determines the total load of contaminants (e.g., heavy metals) introduced to soils, such that high cumulative inputs can approach or exceed regulatory thresholds and increase risks of ecological or agronomic harm.[129,131]

3.2.1.2.1 Implementation and challenges in primary literature

Across the compiled literature (98 unique experimental studies; 234 unique application-rate datapoints restricted to field and laboratory work; Figure 7), two distinct practice patterns emerge depending on whether studies are framed as EW or as rock flour used for agronomic fertilisation. In EW-focused studies (44 studies; 79 datapoints), application amounts are generally high and often substantially exceed typical agronomic fertilisation rates. For EW, the median application amount is 74 t ha^{-1} (Interquartile Range, IQR, 20–153 t ha^{-1}; total range 0.05–125,000 t ha^{-1}). Field studies tend to apply lower rates than laboratory experiments: field medians are 20 t ha^{-1} (IQR ~6–100 t ha^{-1}; total range 1.24–125,000 t ha^{-1}), whereas laboratory studies cluster around 100 t ha^{-1} (IQR 50–200 t ha^{-1}; total range 0.05–500 t ha^{-1}). In other words, EW laboratory experiments typically test substantially higher loadings than field trials, and the upper end of the EW distribution includes very large, sometimes extreme experimental doses. In contrast, studies treating rock flour primarily as a fertiliser or soil amendment (54 studies; 155 datapoints) apply markedly lower amounts: the median is 5 t ha^{-1} (IQR ~1–20 t ha^{-1}; total range 0.001–300 t ha^{-1}). Field-based fertiliser studies use the lowest rates, with a median of 2 t ha^{-1} (IQR ~0.2–5 t ha^{-1}; total range 0.001–50 t ha^{-1}), while laboratory fertiliser experiments apply higher amounts (median 10 t ha^{-1}, IQR ~3–28 t ha^{-1}; total range 0.01–300 t ha^{-1}). In both cases, these remain far below typical loadings in EW laboratory studies.

A recurring challenge in the literature is the inconsistent and sometimes incomplete reporting of application amounts, which complicates cross-study comparison and synthesis. In laboratory and mesocosm experiments in particular, feedstock additions are often reported only as mass per treatment (e.g., grams per pot) without sufficient information on surface area, soil depth, bulk density, or experimental footprint, making it difficult to derive area-normalised application rates (t ha^{-1}). Some studies report applied rock volumes without providing material densities, preventing conversion to mass-based metrics, while others describe repeated applications without clearly specifying frequency, duration, or cumulative totals. These reporting gaps hinder quantitative comparison and meta-analysis. To improve comparability and reproducibility, studies should always report (i) area-normalised application amounts (t ha^{-1}) alongside mass per treatment, (ii) material density, (iii) soil depth and experimental dimensions, and (iv) clearly stated application frequency and cumulative total applied mass over the full study period. Consistent reporting of

these parameters would substantially strengthen synthesis efforts and the interpretation of dose–response relationships in EW research.

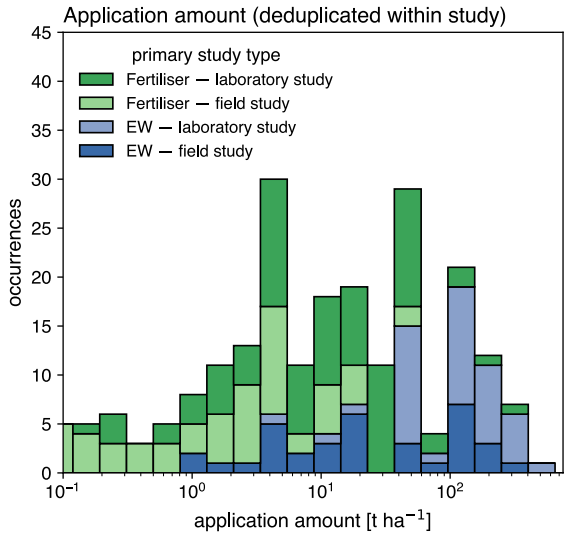

Figure 7: Distribution of total (cumulative) feedstock application amounts reported in field and laboratory studies, separated by study framing: Enhanced Weathering (EW) vs. agronomic fertiliser use. Each occurrence represents one unique value per unique study, where some were calculated from given application amounts and experimental dimensions. Enhanced Weathering studies (blue tones) generally apply substantially higher amounts than fertiliser-focused studies (green tones). In both research domains, laboratory experiments tend to use higher application rates than field trials, with medians of ~100 vs. 20 t ha^{-1} for EW and ~10 vs. 2 t ha^{-1} for fertiliser studies. These patterns highlight systematic differences in experimental design and objectives between CDR-focused and agronomic research traditions.

3.2.1.3 Grain size and surface area

One of the most important factors influencing EW outcomes—and a defining feature distinguishing it from natural weathering—is the fine particle size of applied rock materials. Particle (or grain) size and the associated reactive surface area exert a fundamental control on mineral dissolution and therefore represent key parameters in efforts to accelerate mineral reactivity and CDR. Finer particles generally have higher specific surface area (SSA, in $m^2\ g^{-1}$), promoting fluid–solid interaction and faster dissolution kinetics, a relationship widely documented across laboratory experiments, natural weathering systems, and theoretical rate formulations.[181–183] Thus, grain size is primarily a proxy for SSA in EW studies. Because of this mechanistic link, surface area is commonly used to normalise dissolution rates and evaluate the reactivity of candidate feedstocks in EW studies (e.g., refs. [125,184,185]).

Surface area can be determined either experimentally or by geometric estimation. The most commonly reported experimental measure is SSA determined using the Brunauer–Emmett–Teller (BET) method, which derives SSA from gas adsorption isotherms and accounts for both external particle surfaces and internal features such as surface roughness, pores and microfractures.[186–188] N_2 is most frequently used as the adsorbate due to its availability and suitability for a wide range

of solids, although alternative gases such as krypton or argon may be employed depending on surface characteristics.[189–191] BET measurements provide a comprehensive representation of surface complexity, but results may vary substantially due to differences in adsorbate gas, sample preparation, analytical conditions and inter-laboratory procedures.[188,192,193]

In contrast, geometric surface area (GSA) is calculated from particle size data assuming idealized particle shapes, typically spherical or cubic, and smooth surfaces.[194–196] Particle size information is commonly derived from laser diffraction or sieving techniques and may be reported in diverse formats, including mean or median diameters, minimum-maximum ranges or percentile-based size distributions. This heterogeneity in reported grain size metrics complicates direct comparison across studies and requires pragmatic assumptions to derive representative particle diameters.[197–199] Representative diameters can be estimated using geometric means of size bounds or percentile bins following established particle technology guidelines,[200–202] allowing GSA to be calculated consistently across datasets when particle density and shape are assumed or known.

While GSA offers a reproducible and transparent approach that is less sensitive to analytical variability and sample ageing,[193,203] it systematically underestimates total SSA because it neglects surface roughness, porosity and microfracturing.[181,194] Consequently, GSA values are typically one to two orders of magnitude lower than BET-derived SSA, a discrepancy widely reported in dissolution studies (e.g., refs. [183,184,188,204]). Conversely, BET measurements may overestimate the truly reactive surface by including mineralogically non-reactive surface area (i.e., co-mingled phases not susceptible to acidic dissolution) and micropores not accessible to fluids.[193,194,205] Neither approach is universally optimal, and the choice of surface area normalisation depends on data availability, study objectives, and the balance between physical realism and methodological consistency.

The divergence between BET and GSA surface areas is often conceptualized using a surface roughness factor, defined as the ratio between measured BET surface area and geometrically estimated surface area.[125,206,207] This provides a first-order indication of the extent to which real mineral surfaces depart from idealized particle geometries. In practice, however, studies report grain size and surface area using a wide range of metrics, measurement approaches, and derived quantities that are not directly comparable and may reflect fundamentally different physical properties of the material. This heterogeneity—in both what is reported and what those quantities represent mechanistically—poses a substantial challenge for synthesizing EW datasets and interpreting cross-study trends. The implications of this variability for evaluating reported CDR rates are examined in Section 5.1.2.

Importantly, neither particle size nor measured surface area alone uniquely determines dissolution behaviour. Experimental and modelling studies show that the sensitivity of weathering rates to grain size depends strongly on environmental context, including solution chemistry, pCO_2, hydrologic conditions, and secondary mineral formation.[65,184,208] This reflects the fact that reactive surface area interacts with transport processes and evolving porewater chemistry, such that grain size acts as a first-order control on intrinsic reactivity but does not independently predict realised weathering fluxes across natural systems.

Finally, because specific surface area is fundamentally determined by the degree of comminution, particle size also links mineral reactivity to energetic and logistical constraints. Producing finer materials generally requires greater comminution energy, increasing costs and potentially reducing net CDR under some system boundaries.[23,209,210] As a result, deployment strategies must balance the kinetic advantages of fine particles against the practical considerations of processing intensity and scalability. The implications of grain size for observed CDR rates across experimental, field, and modelling studies are examined in detail in Section 5.1.2.

3.2.1.3.1 Implementation and challenges in primary literature

Across the literature compiled here, primary studies implement EW using a broad range of particle sizes and reporting conventions (Figure 8). Mean and median diameters in laboratory and modelling studies typically fall in the tens of micrometres, while higher percentiles (p80, p90, p97) often extend into the hundreds of micrometres and, in some fertiliser-derived field datasets, into the millimetre range. Surface area reporting shows similarly wide variability (Figure 9). Brunauer–Emmett–Teller SSA most commonly cluster between ~1 and 15 $m^2\ g^{-1}$ but span several orders of magnitude, while GSA values are reported far less frequently and are typically one to two orders of magnitude lower. Laboratory and field studies dominate BET reporting, whereas GSA is most often used in modelling contexts, reflecting the practicality of deriving GSA from assumed particle diameters.

We also note that particle-size distributions are reported using an exceptionally wide range of metrics across the EW literature, substantially complicating cross-study comparison. Across the compiled studies, grain size is described using >80 distinct measures or reporting conventions spanning several fundamentally different representation types. These include (i) continuous distribution descriptors, such as mean diameter and percentile diameters (e.g., p10–p97), represented by 8 distinct metrics; (ii) cumulative mass fractions, reporting the proportion of material finer or coarser than specified size thresholds, represented by 20 distinct cutoff definitions; and (iii) discrete size-class mass fractions, where material is partitioned into explicit size intervals (e.g., sieve or laser-diffraction classes), comprising 43 distinct bin definitions and representing the most heterogeneous reporting category. Additional reporting modes include aggregated textural classes (sand, silt, clay, and gravel), size-range or extrema descriptors (minimum and maximum grain sizes and reported size ranges), and hybrid or method-specific composite classes that reflect analytical or application-driven grouping choices rather than standardized particle-size conventions.

This diversity in grain-size reporting makes systematic comparison across studies challenging, particularly because particle size is ultimately used as a proxy for reactive surface area, rather than a direct controlling variable. Compounding this issue, many studies do not report measured BET surface area or even calculated GSA, limiting the ability to directly relate reported grain size to weathering kinetics or CDR outcomes. Together, these factors highlight the need for more standardised grain-size reporting, alongside more routine measurement or reporting of surface area, to improve comparability, interpretability, and mechanistic understanding across EW experiments and models.

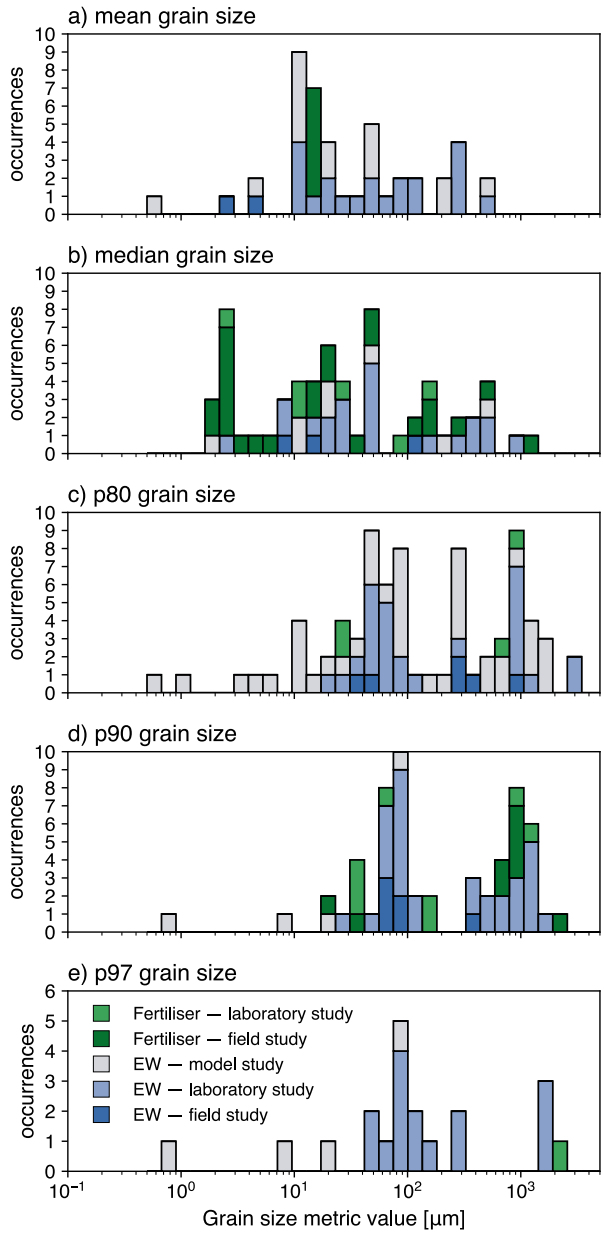

Figure 8: Grain-size metrics reported in Enhanced Weathering (EW) and rock flour as fertiliser studies: mean (a), median (b), p80 (c), p90 (d), and p97 (e). Each occurrence represents one unique value per unique study. Stacks separate EW vs fertiliser studies and are further stratified by primary study type (field, laboratory, and modelling).

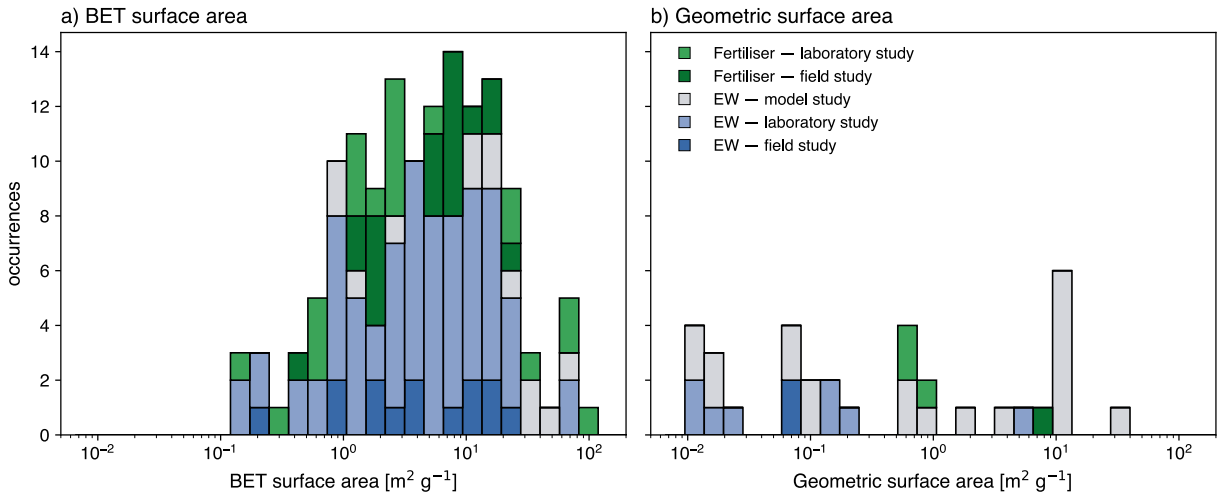

Figure 9: Reported feedstock surface areas, harmonised to $m^2\ g^{-1}$ from (a) Brunauer–Emmett–Teller (BET) surface area and (b) geometric surface area (GSA) methods. Each occurrence represents one unique value per unique study. Histograms are stacked by study focus (Enhanced Weathering (EW) vs. rock flour as fertiliser studies) and primary study type (field, laboratory, modelling).

3.2.2 Soil chemistry

The efficiency of EW in soils is strongly regulated by environmental conditions that govern mineral dissolution kinetics and CDR efficiency. As extensively reviewed elsewhere,[62] pH, temperature, and saturation state of the soil solution exert first-order controls on both weathering rates and the fraction of weathering that results in direct CDR.[62,211] Their combined effects define a constrained set of conditions under which EW can be most effective.

Soil pH is arguably the most critical geochemical variable controlling EW outcomes. Silicate mineral dissolution rates typically increase under acidic conditions (pH <5.5) and tend to reach a minimum near circumneutral pH, with some minerals also exhibiting increased dissolution under highly alkaline conditions (pH >9; Figure 10).[123,212] However, highly alkaline soils are rare in agricultural systems,[213] making acidic to circumneutral conditions most relevant for EW. Importantly, rapid silicate dissolution at low pH does not necessarily translate into direct CO_2 sequestration, because additional alkalinity—expressed in its explicit conservative form as a cation charge surplus[89]—will not necessarily directly translate into more DIC.[214] This concept, known as alkalinisation carbon-capture efficiency,[214] reflects the extent to which added alkalinity increases the storage of DIC, and depends on the partitioning of DIC between dissolved CO_2 (including H_2CO_3), HCO_3^- and CO_3^{2-}. In soils, this relationship also depends on buffering capacity: where added alkalinity is consumed by exchange reactions, proton buffering, or dissolution/precipitation reactions without substantially increasing pH, near-term conversion of weathering-derived alkalinity into bicarbonate may remain limited; where buffering capacity is lower or becomes exhausted, weathering-induced pH increases can shift DIC speciation toward bicarbonate and enhance CO_2 uptake.[215]

At very acidic pH conditions (pH <5), inorganic carbon exists almost exclusively as dissolved CO_2. Under such strongly acidic conditions, silicate dissolution may initially neutralise acidity without directly producing substantial bicarbonate, such that near-term CO_2 capture per unit alkalinity generated is low,[161] as seen for instance in an Australian trial on acidic tropical soil.[216] Conversely, at circumneutral pH, inorganic carbon exists mainly in the form of HCO_3^-. The same silicate weathering reaction—generating the same amount of alkalinity—would then convert a much larger fraction of added alkalinity into bicarbonate ions:[214]

$$CaSiO_3 + H_2O + 2\,H^+ \leftrightarrow Ca^{2+} + H_4SiO_4 \quad (7)$$

$$CaSiO_3 + H_2O + 2\,CO_2 \leftrightarrow Ca^{2+} + H_4SiO_4 + 2\,HCO_3^- \quad (8)$$

Within this framework, these contrasting tendencies suggest the existence of an optimal pH range in which moderately acidic conditions combine sufficiently high weathering rates with sufficient buffering capacity and carbonate-system conditions to enable substantial CO_2 sequestration.

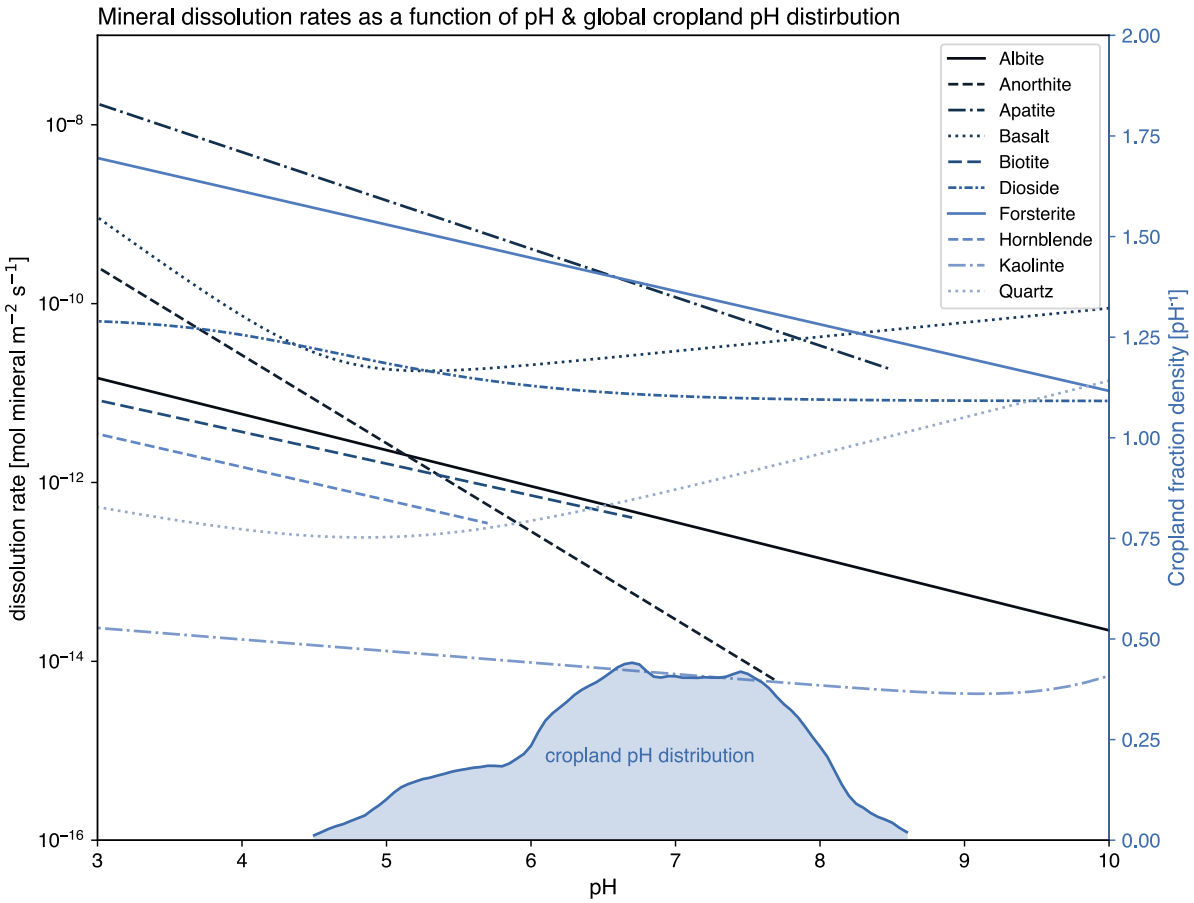

Figure 10: Mineral dissolution kinetics and global cropland soil distribution as a function of pH. Mineral dissolution rate laws (refs. [119,217]) for selected silicate and accessory minerals are shown as a function of solution pH, illustrating the strong sensitivity of weathering kinetics to soil pH. The shaded curve (right axis) shows the distribution density of global cropland soil pH derived from gridded global soil datasets.[218,219] The area under the distribution between any two pH values corresponds to the fraction of cropland soils within that range.

However, in highly acidic soils where alkalinisation carbon-capture efficiency is close to zero, EW could still contribute to CDR downstream from the field and potentially much later in time. This can be conceptualized as a lag time in CDR realisation: weathering in the soil decreases acid export and releases cations, but atmospheric CO_2 uptake is only realised once exported weathering products encounter higher-pH conditions in rivers or the ocean and carbonate equilibria re-adjust. As with bicarbonate and cation export more generally, retention of these weathering products within the field through plant uptake, incorporation into secondary minerals, or adsorption reactions on SOM, clays, and/or oxides would limit or prevent this downstream CDR.[68,92,220]

Temperature exerts a fundamental kinetic control on silicate dissolution, with rates generally increasing exponentially according to Arrhenius-type behaviour when water and mineral supply are not limiting (see also Section 3.2.3).[58,221] Laboratory and soil experiments demonstrate substantially lower dissolution rates at low temperatures, with differences of up to several orders of magnitude between <10 °C and >20 °C conditions.[222,223] In soils, this kinetic enhancement is coupled to indirect biological effects: warming increases soil respiration and CO_2 production, potentially enhancing weathering via higher pCO_2 and organic acid exudation, but may simultaneously stimulate organic carbon losses that complicate net carbon accounting.[100,224]

The saturation state of the soil solution further constrains EW efficiency. As silicate minerals dissolve, reaction products accumulate in solution, progressively reducing dissolution rates as thermodynamic equilibrium is approached. Many natural soil solutions are already near saturation with respect to primary silicate minerals, helping to explain why field weathering rates are often orders of magnitude lower than laboratory-derived rates.[225] Soil texture and structure play a central role in regulating this process. Poorly drained, compacted, or clay-rich soils restrict solute flushing and promote saturation,[226] while excessively sandy soils limit water retention and mineral contact time.[227] Optimal EW performance might therefore be found in intermediate soil textures that ensure both adequate water availability for dissolution and sufficient permeability to export solutes and maintain far-from-equilibrium conditions.

Taken together, these factors underscore that EW efficiency is highly site-specific and will be contingent upon the alignment of multiple environmental and geochemical parameters. In addition to regional climatic drivers such as temperature and precipitation, local soil properties—including pH, texture, structure, and hydrological regime—critically influence weathering dynamics and CDR potential. Maximizing EW efficiency will therefore require careful site selection and integrated consideration of climate, soil chemistry, and hydrology, rather than uniform application across landscapes.

3.2.2.1 Implementation and challenges in the literature

Across the compiled EW literature, baseline pH conditions span predominantly acidic to mildly acidic soils, with medians in the range of ~4.8–5.5 depending on extractant (Figure 11). Soil pH measured in water clusters around moderately acidic values (median ≈ 5.5), while $CaCl_2$-based measurements are systematically lower (median ≈ 4.8), reflecting differences in ionic strength and exchange equilibria inherent to the extractant chemistry. Porewater and effluent measurements

exhibit broader variability (median ≈ 5.4) with values extending into near-neutral and mildly alkaline conditions.

These distributions indicate that most EW experiments to date have been conducted in acidic to weakly acidic agricultural soils, often within the pH range where silicate dissolution kinetics are enhanced and agronomic lime recommendations are common. This focus is consistent with two practical considerations. First, acidic soils are widespread in intensively managed croplands,[29,228–230] particularly in humid tropical regions as well as regions with high legacy acidity inputs. Second, from a geochemical perspective, lower pH conditions increase proton-promoted dissolution of silicate minerals[119,120]—but also risk adverse impacts on immediately realised CDR due to strong acid effects (see also Section 3.4.3).[214,231]

The comparatively higher porewater and effluent pH data further suggest that EW deployment frequently occurs in systems where solution chemistry is responsive to mineral inputs and soil buffering processes. This pattern is consistent with model–experiment comparisons showing that aqueous pH can increase rapidly following basalt addition, even where bulk soil pH responds more gradually due to exchange buffering.[232] In this context, porewater pH may more directly reflect short-term reaction progress and alkalinity export potential, whereas bulk soil pH integrates exchange reactions and longer-term shifts in soil acidity status.

Methodological heterogeneity remains a central limitation. Soil pH depends strongly on extractant type, molarity, and soil-to-solution ratio, and different measurement definitions (e.g., water pH, $CaCl_2$ pH, buffer pH) can yield systematically different values for the same soil.[232] Because agronomic liming recommendations are often based on buffer pH rather than water pH, translating reported baseline values into amendment requirements requires careful attention to measurement protocol. In the EW literature, such methodological details are not always reported consistently, complicating cross-study comparison and synthesis. A further source of ambiguity is that soil pH and aqueous pH values are distinct, even though they are sometimes used interchangeably. For example, some studies do not clearly distinguish between changes in bulk soil pH and changes in porewater or effluent pH, nor do they consistently separate soil matrix measurements from drainage chemistry. Conflating these variables—which reflect distinct biogeochemical domains—can obscure mechanistic interpretation, particularly when assessing EW-induced alkalinisation, nutrient availability, or trace metal mobility.

Overall, the baseline pH distributions compiled here indicate that EW research has primarily targeted acidic agricultural soils where traditional agronomic benefits (acidity remediation) and enhanced dissolution kinetics are expected. However, this concentration of studies in acidic systems also limits the empirical basis for extrapolation to neutral or alkaline soils, where weathering kinetics, buffering capacity, and biogeochemical feedbacks may differ substantially. The implications of EW-induced pH modification for soil chemistry, nutrient cycling, and long-term CDR efficiency are examined further in Section 7.

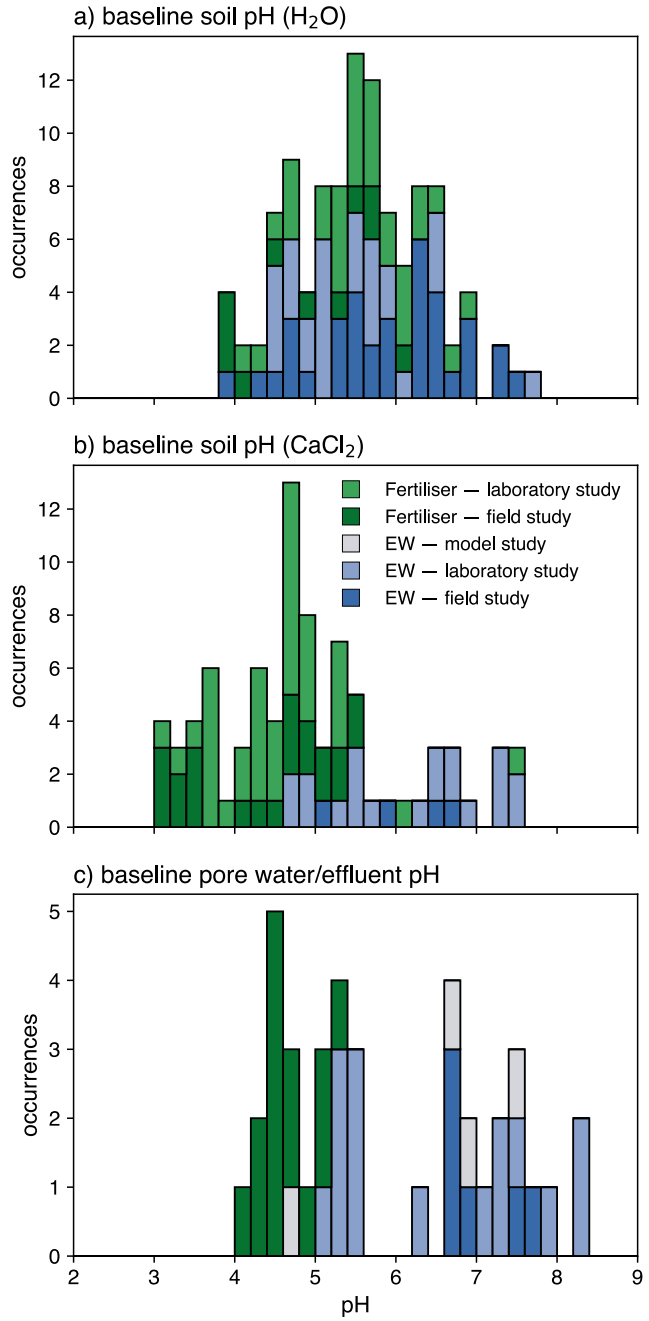

Figure 11: Distribution of reported baseline soil and solution pH values in Enhanced Weathering (EW) and rock flour as fertiliser studies. (a) Soil pH measured in H_2O extracts. (b) Soil pH measured in $CaCl_2$ extracts. (c) Pore water and effluent pH values, combining measurements from soil solution and drainage waters (stream, river, as well as column experiment effluent). Stacked histograms are further disaggregated by primary study type (field, laboratory, model). Each occurrence represents one unique value per unique study.

3.2.3 Hydroclimatic control on EW

Hydroclimate is a fundamental regulator of CDR through EW, as it controls the availability of water and temperature—two key factors strongly influencing weathering reaction kinetics.[23,172,233,234]

In humid regions, elevated soil moisture increases the wetted surface area of mineral amendments, enhancing their exposure to weathering agents.[235] Elevated soil moisture also promotes soil drainage[236] and the efflux of weathering products.[237] Drainage, and lower residence times, is key to maintain non-equilibrium chemical conditions that sustain higher weathering rates,[238,239] compared to low-drainage conditions that favour saturation of soil solution.[240] High temperatures complement moisture availability by accelerating mineral dissolution kinetics.[56,221] At regional scales, the co-occurrence of warm and wet conditions maximizes both reaction rates and solute export, particularly where temperature and precipitation seasonality are in phase.[236] Rainfall and temperature also regulate ecosystem photosynthesis and respiration, with higher values generally increasing *p*CO2 and the release of acidifying root exudates,[241,242] further enhancing weathering rates.

Consequently, warm and humid regions—especially the tropics—consistently emerge as hotspots of modelled and experimental CDR potential.[19,24,243–245] Laboratory experiments frequently demonstrate higher weathering rates under wetter and warmer conditions.[237,246] Substantial CDR potential has also been reported for example in Costa Rica.[244] A global RTM confirms the high efficiency of warm and humid regions (cf., Section 5.2.3), which often overlap with economically developing regions.[19,24] However, global warming is expected to have a minor effect on CDR rates[24]—with even potential negative impacts (at least for carbonates) in some tropical regions.[87]

Despite these insights, quantifying the hydroclimatic impacts on soil chemistry remains challenging. Laboratory conditions often struggle to replicate the natural hydroclimatic variability.[180] In the field, CDR rates are influenced by complex and difficult to disentangle interactions with soil type and biogeochemical conditions:[247,248] moisture and drainage are also controlled by soil texture,[97,105] land management and use of other amendments,[249] as well as the hydrologic transport of weathering products through the vadose zone, aquifers, and riverine systems.[66] Moreover, the efficiency of CDR depends on water chemistry and climate[214] and may be constrained by silicic-acid saturation or ecological impacts associated with elevated riverine pH and alkalinity.[17] Ultimately, site selection for EW must balance favourable hydroclimatic conditions with proximity to mineral resources and associated transport requirements.[133,250,251] In drier regions, irrigation is unlikely to fully offset rainfall deficits. In fact, while irrigation can supplement moisture, modern irrigation methods aim to maximize efficiency and minimize drainage—contrary to the high-moisture, high-flow conditions required for optimal EW.

Lastly, the temporal structure of water inputs is also critical,[66,172,252] as hydrological forcing in soils is inherently intermittent. For example, a given total water input delivered during a single event may have very different geochemical consequences than the same amount distributed across multiple smaller events, due to differences in infiltration dynamics, residence time, and soil solution chemistry. Reporting time-resolved hydrological data (e.g., event frequency, intensity, or time-series precipitation records) alongside cumulative water inputs would therefore improve interpretation of weathering responses. Such reporting would enable more robust meta-analysis,

reduce ambiguity in interpreting hydrological controls on weathering, and strengthen the empirical basis for linking climate and water availability to realised CDR from EW.

3.2.3.1 Implementation and limitations in the literature

Even though climatic conditions strongly influence mineral dissolution kinetics and the transport of weathering products, these parameters are reported with varying levels of detail and consistency across the compiled literature.

Reported average temperatures span a broad range across the compiled studies (Figure 12). Values extend from ~3–35°C, with a median near 20°C, reflecting the diversity of climatic and experimental settings represented in the literature. Field experiments generally occur in temperate to warm climates, with median temperatures of roughly 11°C for EW field studies and 16°C for fertiliser field studies, while laboratory and modelling studies commonly assume or simulate warmer conditions (medians ~22–24°C). This wide distribution indicates that EW experiments are conducted under a broad range of thermal environments, though temperature itself is typically not a controlled variable in field studies and is therefore often reported only as contextual site information.

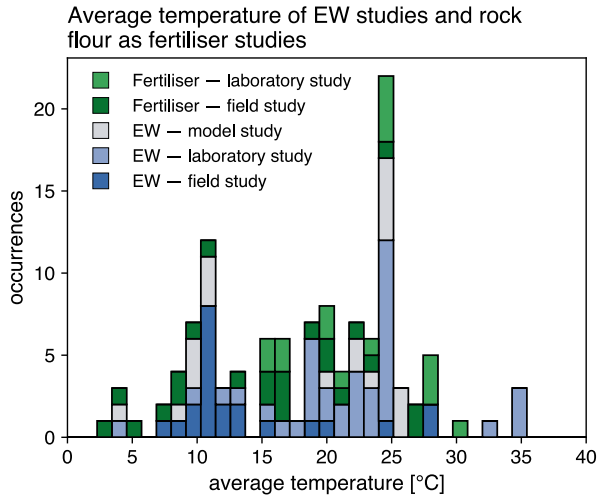

Figure 12: Average temperature distribution from Enhanced Weathering (EW) and rock flour as fertiliser studies. Occurrences are further stratified by primary study type (field, laboratory, model). Each occurrence represents one unique value per unique study.

Despite the critical control of water availability and hydrological flux on EW, the reporting of water inputs in the literature is uneven and fragmented (Figure 13). Few studies report the full set of water balance components required to constrain hydrological conditions. While precipitation is relatively well-documented (Figure 13a), the reporting of other hydrological fluxes is much less consistent. Irrigation (and precipitation + irrigation) is reported in fewer studies (Figure 13b,c), but this likely stems from the fact that not all studies add irrigation water. Additional variables that would help characterize the broader site water balance—such as evapotranspiration (Figure 13d)

and infiltration (Figure 13e)—are rarely reported. As a result, only a limited subset of studies provides sufficient information to reconstruct a simplified water balance.

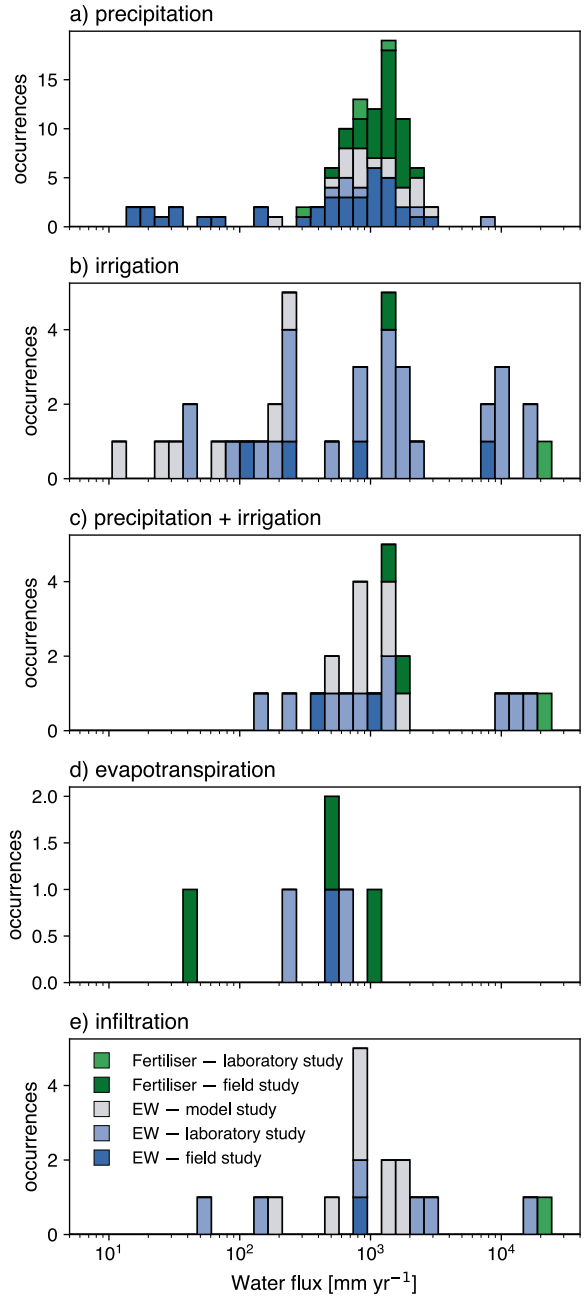

Figure 13: Distribution of water inputs reported in Enhanced Weathering (EW) and rock flour as fertiliser studies: (a) precipitation, (b) irrigation, (c) precipitation + irrigation, (d) evapotranspiration, and (e) infiltration. All values were harmonised to mm yr^{-1}. Data are stratified by study type (field, laboratory, model) and by study focus (EW vs. rock flour fertiliser studies). Where studies reported total water fluxes over an experimental period, these totals were converted to annualized fluxes by dividing by the reported study duration. Each occurrence represents one unique value per unique study.

Where precipitation is reported, values span a wide climatic range, with a median near 1000 mm yr^{-1} and an IQR of roughly 600–1470 mm yr^{-1} (Figure 13a), consistent with typical temperate and humid agricultural environments. Irrigation rates exhibit substantially greater variability (Figure 13b), with median values near 800 mm yr^{-1} (IQR 128–2000 mm yr^{-1}) but extreme values can reach 17000-24000 mm yr^{-1} in some laboratory studies. Similarly high values also occur for combined water inputs (Figure 13c) and infiltration (Figure 13e). While these extreme fluxes (unrealistic in field conditions) have been used in controlled laboratory or column experiments to eliminate water and thermodynamic limitations and provide strong signals for estimating potential rates on laboratory timescales, EW efficiency in field conditions is often influenced by hydrological constraints, making it difficult to extrapolate these results to more realistic hydrologic regimes. Some EW field studies exhibit extremely low water inputs which may significantly limit the amount of weathering one would expect to occur.

Closely related to this issue, water input and moisture conditions are reported using a wide array of metrics across EW studies, substantially complicating synthesis and comparison. Reported measures include total water volumes applied per treatment, irrigation rates expressed per plot or experiment, and qualitative descriptions of watering intended to maintain a target soil moisture. In many cases, incomplete metadata—such as missing experimental dimensions or soil water balance information—make it difficult to convert these values into a common, comparable metric. This problem is particularly pronounced in studies that aim to maintain constant soil moisture, which frequently report the moisture target but not the actual water input required to achieve it. Reporting water depth-equivalent water flux (or simply precipitation depth, e.g., mm or mm yr^{-1}) would substantially improve cross-study comparability by providing a scale-independent representation of hydrological forcing. These inconsistencies mirror challenges identified for grain-size reporting and highlight a broader need for methodological standardization.

3.3 Transport and long-term storage of CDR from EW

For EW to deliver durable CDR, weathering products must also be transported through soils, groundwater, rivers, and ultimately into long-lived carbon reservoirs. This section examines how hydrological and biogeochemical transport processes govern the movement of alkalinity, cations, and DIC from land to downstream environments, and how these pathways influence the timing, location, and durability of storage. We first consider transport through soils and rivers and then discuss the principal long-term storage reservoirs for EW-derived carbon, including ocean alkalinity and carbonate formation in soils.

It is important to note that the same transport pathways and storage domains considered here are also where losses, reversals, and delays in CDR can occur. While this section introduces these processes in the context of carbon transport and storage at a system level, Section 3.4 examines them explicitly as loss mechanisms, providing a more detailed, process-level assessment.

3.3.1 Transport of weathering signals in soils

The products of EW are mobilised within soils through a combination of hydrological transport and biogeochemical processing. As weathering-derived solutes move through porewaters, they are not transported conservatively but interact with mineral surfaces, organic matter, and soil exchange complexes—undergoing processes such as adsorption, ion exchange, secondary mineral formation, and biological uptake—that can modify their composition, timing, and eventual export to aquatic systems (cf., Sections 3.4.1 – 3.4.6). Together, these processes determine how, when, and in what form the weathering signal is transmitted from soils to downstream environments.

Precipitation, irrigation, or snowmelt that interacts with feedstocks and carries dissolution products can move to streams along various surface and subsurface pathways. Shallow flow paths activate during rainfall events as the regional water table rises or as a perched water table forms and saturates permeable near-surface soils.[253–257] While ephemeral, these shallow flow paths may contribute substantially to streamflow, especially during events with high antecedent water storage, and can transmit dissolved solutes rapidly to the stream. During dry periods, deeper and slower-moving groundwater represents the main contributor to flow in perennial streams.[258,259] Depending on climate and watershed characteristics, baseflow may provide from one-third to more than one-half of total annual streamflow.[260,261]

Because various pathways deliver water to the stream and these precipitation-runoff pathways originate across the watershed at various distances from the stream, the transit times of catchment waters that convey solutes—including the EW products—will be distributed. The nature of the transit-time distribution is influenced by topography, watershed size, prevailing geology, precipitation characteristics, human modifications (e.g., tile drains), and various other factors. Although *a priori* determination is not yet possible, transit-time distributions can be estimated from analysis of natural tracers in precipitation and streamflow and from simulations with watershed hydrologic models. Findings from this research reveal that mean water transit times on the order of years are not uncommon and that the distribution in travel times may span from days to years within small watersheds and from weeks to millennia in larger watersheds [262–266].

Compared to water transit times, those of EW-derived solutes may be somewhat longer due to storage of solutes, for example due to sorption to cation exchange sites, conversion to secondary phases, or plant uptake (cf., Sections 3.4.1–3.4.5). Considering the EW-relevant geochemical processes together with precipitation-runoff processes, it is reasonable to expect that EW solutes could appear in surface waters soon after application of a mineral amendment, but a substantial fraction of alkalinity generated from initial weathering reactions will be stored within the terrestrial reservoir for extended time periods and released slowly—over a period of years—to surrounding streams and rivers (e.g., ref. [68]; Section 3.4.6).

To date, there are exceedingly few EW field trials suitable for sharpening these expectations.[70,243] Extrapolations from soil-column and mesocosm experiments have limited utility for constraining how EW products are exported from watersheds because these experiments poorly represent complex watershed flow paths shaped by geologic structure, topography, and natural precipitation regimes. In contrast, widespread liming across the Mississippi River basin provides an inadvertent large-scale analogue, demonstrating that weathering products from CO_2-capturing soil amendments can be exported to surface waters.[267–269] A >100-year record from the terminus of the

Mississippi River suggests efficient transport of liming-derived weathering products to the river network, albeit with a substantial time lag after carbonate rock application.[270] Watershed-scale RTMs capable of simulating these hydrologic characteristics are being advanced and may be suitable for exploring the dynamics of EW solute delivery to streams and rivers.[271–273] Building confidence in these model-based assessments will require evaluation against streamwater chemistry measurements from EW field trials.

3.3.2 Transport of weathering signals in rivers

Once EW products are exported from soils into the aquatic continuum, rivers do not merely function as passive conduits but operate as active biogeochemical reactors—transforming and redistributing weathering through coupled physical, chemical, and biological processes.[274]

On a global scale, recent modelling suggests that riverine carrying capacity is generally not a major limiting factor for EW deployment.[275] However, this conclusion is sensitive to the assumed saturation state threshold for carbonate precipitation, which releases CO_2 and reduces alkalinity loads. This precipitation threshold remains poorly constrained in river systems and is rarely evaluated empirically in EW studies.[65,276,277] If conservative thermodynamic thresholds are applied—such as assuming precipitation occurs immediately upon reaching saturation—rivers can potentially become a bottleneck for durable CDR, significantly reducing the efficiency of carbon transfer to the oceans.[277] At the regional scale, carbon transport limitations arising from carbonate precipitation also warrant attention. Recent catchment-scale assessments in the UK arable croplands indicate that carbonate precipitation could reduce the CDR potential of EW by 16–27% depending on EW deployment amounts.[65] However, for both global and regional studies, it is critical to distinguish between temporary and net carbon loss from carbon precipitation; precipitated carbonates may redissolve once in undersaturated deep waters, surface bedload or coastal marine sediments, effectively neutralising the net carbon loss.[31]

Even in the absence of carbonate precipitation, shifts in the carbonate chemistry itself from thermodynamic equilibria can compromise net CDR efficiency through carbon degassing (cf., Section 3.4.7). The addition of alkalinity alters the aquatic carbonate system, potentially increasing the partial pressure of CO_2 and driving "carbon leakage" via degassing to the atmosphere. In a recent study that used a dynamic river network model to track downstream carbon loss in rivers following EW in North America, most analysed river reaches exhibited low leakage rates (<5%), but specific hydrological regimes experienced leakage exceeding 15% of initially captured carbon.[66] This high variability complicates the quantification of net carbon credits and underscores the necessity for high-resolution, region-specific analyses rather than relying on continental or global averages.

Beyond inorganic chemistry, EW products may induce complex indirect effects on riverine biogeochemical cycles. Increases in pH and alkalinity have been argued to accelerate the decomposition of dissolved organic carbon, potentially releasing additional CO_2 and offsetting the intended sequestration benefits.[111] On the other hand, alterations in river chemistry can also influence primary production; stimulated photosynthesis may temporarily increase uptake of DIC,[67] yet the long-term fate of this biomass remains a significant uncertainty. These biological

fluxes are further complicated by exchange with the hyporheic zone, where intense respiration within the stream corridor contributes substantially to riverine CO_2 evasion.[278] The effects of EW on hyporheic zone processes remain largely unexplored. Other downstream processes, including reverse weathering and the fate of alkalinity as rivers enter the coastal ocean, also remain largely unconstrained in the context of EW.[279]

Despite these complexities, rivers represent a potentially critical leverage point for MRV of CDR from EW.[280] By spatially integrating geochemical signals across watersheds, river monitoring offers a potentially cost-effective and scalable method for verifying net carbon capture. However, a primary challenge for this approach is the "signal-to-noise" issue: detecting the specific alkalinity signal from EW against the high background variability of natural river dynamics remains a key hurdle for confident attribution. River network models coupled with field sampling as the use of control catchments could help address this question and test the feasibility of this river aggregation approach.[281]

3.3.3 Carbon storage from EW

For an effective climate intervention, it is imperative that the captured and converted CO_2 remains sequestered from the atmosphere such that our climate goals are met. Alcalde et al., (2018)[282] postulate that storage on the order of 10,000 years (a loss rate of 0.01% yr^{-1}) is sufficient for these ends. This framing implicitly assumes a net-zero emissions context, in which storage must persist until natural carbon cycle processes restore equilibrium.[283] However, under net-negative emissions scenarios, shorter storage durations may still provide climate benefits by reducing peak atmospheric CO_2 concentrations and delaying warming, even if storage is not strictly permanent.[5,284,285]

There are two long-lived products of EW that form the storage repositories for atmospheric CO_2, namely bicarbonate/carbonate ions and solid carbonate minerals. In all processes, CO_2 will be initially transformed into bicarbonate ions dissolved in soil porewaters. These can be transported through run-off or shallow groundwater to rivers and eventually the ocean, where they may remain for thousands of years slowly being transformed into solid carbonate minerals (Section 3.3.1). Alternatively, providing sufficiently over-saturated solutions, solid carbonate minerals may form within soils or drainage waters enroute to the ocean (Section 3.3.2). These may constitute a transient carbon pool, dissolving back into bicarbonate ions if drainage waters become under-saturated. While both storage mechanisms are durable, the precipitation of solid carbonate minerals reduces the efficiency of EW by nearly half due to the re-release of CO_2 during the precipitation reaction:

$$Ca^{2+} + 2\,HCO_3^- \rightarrow CaCO_3(s) + CO_2 + H_2O \qquad (9)$$

3.3.3.1 Storage in soils and along transport pathways

Globally, the upper metre of soils contains an estimated ~2,500–2,700 $GtCO_2$-eq as carbonate minerals, with estimates for the upper 2 m being substantially larger.[286,287] Paedogenic carbonates—those formed in soil—are particularly abundant in arid and semi-arid regions where leaching is limited. These carbonates may form from the dissolution and precipitation of

carbonate-rich bedrock and thus may not represent a net increase in carbon storage. Alternatively, paedogenic carbonates may form in soils atop lithologies that lack carbonates, either via silicate weathering or via external inputs, especially aeolian deposition.[288–293] Similarly, paedogenic carbonates may be a storage reservoir for carbon dioxide removed by EW.[294] Hence, changes in SIC pools have also been used to quantify CDR from EW (cf., Section 4.1.2).

Paedogenic carbonates can be long-lived within soils,[295] with > 250 million years recorded for some materials.[296,297] At the same time, more typical residence times for paedogenic carbonates are on the order of 10^3–10^4 years.[233,298] However, these estimates largely derive from arid and semi-arid environments where limited leaching and persistently alkaline conditions favour carbonate stability and may not be representative of many agricultural systems targeted for EW. In more humid or actively managed soils, or where EW application ceases and soil conditions revert toward more acidic states, paedogenic carbonates may be susceptible to dissolution and remobilisation, reducing their long-term stability as a carbon store while increasing the overall efficiency of carbon removal through EW. Dissolution and reprecipitation of paedogenic carbonates is thought to be important within their formation,[298] which is a feature of some EW RTM simulations (Figure 14; ref. [20]). Paedogenic carbonates are also reported in soils heavily modified with artificial alkaline minerals[277] and predicted using geochemical modelling in EW field trials using returned crushed concrete.[299]

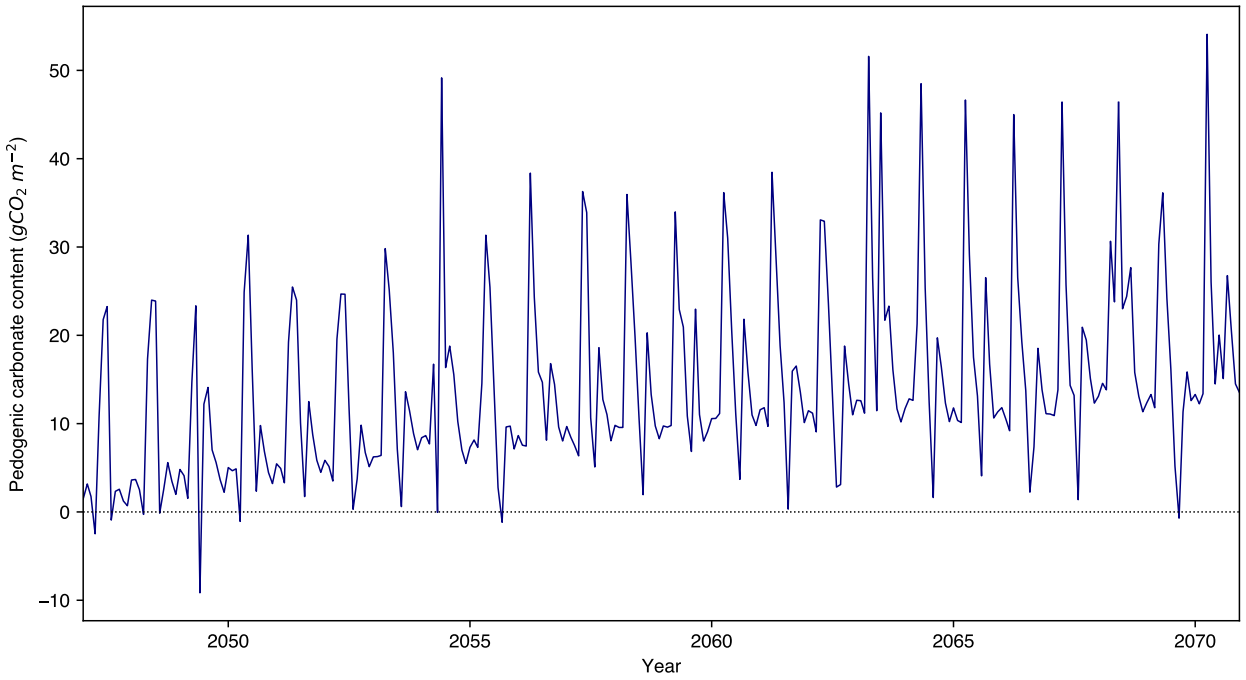

Figure 14: Accumulation of paedogenic carbonate using an Enhanced Weathering (EW) reactive transport model (RTM) that simulates basalt dissolution in U.K. soils under winter barley. Adapted from Kantzas et al., (2022)[20] Extended Figure 7. Simulations were undertaken for Cragmill basalt mineralogy with p80 particle diameter of 100 µm, applied to a winter barley crop rotation field in the UK midlands. The plot shows seasonally driven dissolution/precipitation of paedogenic carbonate with net accumulation over decades.

As rivers outgas CO_2 (cf., Section 3.4.7), many become saturated with respect to calcium carbonate minerals, which could result in their precipitation in the water course—often termed 'tufa' for non-geothermal or 'travertine' for geothermally produced carbonate minerals.[300] This impacts the natural cation budget of rivers[301] and could possibly constrain the scale of bicarbonate flux from EW to 3 to >20 $GtCO_2$ yr^{-1} (Section 3.3.2).[30,275] If carbonate were to precipitate within drainage waters, this storage will be transient if saturation states are not sustained and the solid carbonate remained submerged in river water. Nevertheless, natural tufas and travertines can be long-lived, with the oldest known records exceeding 100,000 years.[302]

Taken together, these observations indicate that carbonate precipitation can constitute a stable carbon storage pathway under specific environmental conditions, particularly in arid and alkaline soils. However, this stability is strongly context-dependent and may not extend to many managed or humid systems targeted for EW deployment. Importantly, the limited durability of paedogenic carbonates compared to oceanic bicarbonate storage does not necessarily represent a net loss pathway. Redissolution of SIC is equivalent to carbonate weathering, with the same potential impact on CDR.[31] Where dissolution proceeds via carbonic acid—or where acid neutralisation suppresses CO_2 outgassing along the soil–river continuum—this process can partially or fully recover the initial CDR penalty associated with carbonate formation.

3.3.3.2 Storage in the oceans

Ocean inorganic carbon is the largest reservoir of carbon at the Earth's surface—equivalent to 150,000 $GtCO_2$—with overall residence times on the order of 100,000 years.[37,90] Inorganic carbon in the ocean is predominately (99%) bicarbonate or carbonate ions,[303] and additional bicarbonate flux from EW offers the possibility of scalable durable storage. However, >95% of the DIC that is introduced into to the ocean from EW would remain within the upper 1 km of the water column. The majority of this would reside within the upper 500 m, or somewhat deeper if the alkalinity is introduced in bottom water formation regions.[304] The mixed surface ocean (<200 m) contains ~3,400 $GtCO_2$-eq of DIC with a residence time of <1,000 years.[37,305] While representing a scalable and durable reservoir of carbon, surface ocean values are substantially smaller/shorter than the potential of the total ocean.

Alkalinity removal from the surface ocean through carbonate mineral precipitation is controlled by calcifying organisms, which collectively remove ~7 $GtCO_2$-eq per year, exported through sinking particles to deeper waters (ref. [306], and references therein). Alkalinity is also removed through secondary silicate mineral formation ('reverse weathering') but this makes a relatively small contribution to the modern ocean.[307] Assessing natural chemical gradients in the surface ocean, Lehmann and Bach (2025)[308] suggest that under 'moderate' (<50 $GtCO_2$-eq yr^{-1}) increases in OAE the loss from microorganism calcification is <3% of the added alkalinity.

There is no evidence of significant alkalinity loss in month-long mesocosm experiments,[309] at least within the resolution of conventional routine measurement techniques for total alkalinity (TA; 0.1–0.5%).[310] Spontaneous carbonate precipitation in seawater requires substantial carbonate supersaturation, and riverine inputs to the ocean are generally not expected to exceed these critical conditions.[311] It therefore appears unlikely that land-based EW would commonly trigger spontaneous carbonate precipitation in the ocean. Thus, CDR from EW is primarily stored as

dissolved bicarbonate. Taken together, these constraints indicate that oceanic losses should be limited (~10–20%; Section 3.4.7) and that most EW-derived carbon will be retained as DIC.[37,279] In practice, storage can be considered durable on climatically relevant timescales once carbon is transferred into the ocean bicarbonate pool.

3.4 Loss processes, reversals, and limitations

The CDR potential of EW is not determined by primary mineral dissolution alone. A range of coupled geochemical, biological, and hydrological processes can reduce, delay, or in some cases reverse realised CDR by retaining weathering products in soils, re-releasing CO_2 in soils, slowing further dissolution, or promoting losses during downstream transport. In this section, we examine the main processes that constrain net EW performance, including secondary mineral formation, cation exchange processes, the presence and interaction with strong acids, biomass uptake, surface passivation, lag times between weathering and CDR occurrence, and CO_2 degassing along the soil–river–ocean continuum.

3.4.1 Formation of secondary phases

During chemical weathering, surface waters are undersaturated with regard to primary silicate minerals (olivine, pyroxene, feldspar, etc.), which therefore dissolve. This can drive these waters to supersaturation with regard to secondary minerals, which include carbonates, (oxy)hydroxides, zeolites and clay minerals. The subsequent precipitation of these secondary minerals (known as neoformation of authigenic minerals) can act to keep waters undersaturated with regard to primary minerals, supporting continued dissolution (e.g., refs. [312–314]). In flow-through soil and watershed systems, continued primary mineral dissolution can be sustained by solute export and/or by secondary mineral precipitation in situ or downstream; secondary mineral formation is therefore not required at every reaction site, but remains a common and important component of weathering systems. If infiltrating water is derived from rainwater, which is initially strongly undersaturated with respect to most secondary minerals, it will take longer for porewaters to reach supersaturation than systems supplied by irrigation groundwater, which may already contain higher dissolved solute concentrations. Secondary minerals (with the potential exception of carbonates) have low chemical activity and are therefore difficult to redissolve.

The formation of secondary minerals has several implications for both EW and natural weathering reactions, and the sequestration of atmospheric CO_2:

1. The formation of soil carbonates affects CDR similarly to marine carbonate precipitation: as discussed above, carbonate precipitation removes alkalinity from solution and releases approximately half of the previously weathering-derived DIC as CO_2, although this CO_2 is not necessarily emitted to the atmosphere. Approximately ~2,500–2,700 $GtCO_2$-eq are stored in the top m of soil in this way, with millennial-scale turnover.[312,315] Changing rainfall conditions or fertiliser use could redissolve this carbonate, reforming the alkalinity in the surface waters.

2. The coating of feedstock silicate grains with secondary minerals will cover the reactive silicates with unreactive silicate material, thus significantly slowing their dissolution (known as "surface passivation" or "soil shielding") (e.g., refs. [19,172,316]).
3. Over repeated years of amendment, it is possible that sufficient clay minerals form to change the drainage patterns of fields, due to alteration of permeability and porosity. This can also change weathering processes via alteration of the hydraulic conductivity of soils.[313]
4. Increased amounts of clay minerals will change the CEC of the soil, with implications for monitoring EW (e.g., ref. [220]) and fertility of the soil, which could be affected positively or negatively (depending on local conditions).
5. Secondary minerals retain cations that would otherwise be used to charge-balance alkalinity. In other words, the formation of secondary minerals—especially clays—hinders the efficiency of EW, such that less CDR is realised per unit of feedstock applied [317,318]. This is a slightly different process from "reverse weathering", which occurs in marine settings when clay formation re-releases stored CO_2 (e.g., ref. [319]), and the effects of reverse weathering in continental settings remains largely unknown.[320]
6. Secondary minerals in the form of oxides (particularly iron-rich oxides and oxyhydroxides, known as the "rusty carbon sink") and clays can help preserve organic carbon during burial both in soils and in the marine realm.[101,321–324] Hence, secondary mineral formation that hinders inorganic carbon sequestration may enhance organic carbon sequestration. This behaviour is largely unconstrained in EW systems but may be more significant than inorganic carbon sequestration.

One of the key issues in measuring secondary mineral formation is that the most important phases, especially clays and oxides, will often have no fixed chemical formulae but instead act as a "sponge" for available cations.[124] This means that there are no definitive diagnostic elements, and in any case each has a different mobility during weathering,[312] making secondary mineral formation difficult to extrapolate across settings. Further, using detection methods such as X-ray diffraction (XRD) only detects the presence of secondary minerals, rather than their neoformation due to feedstock weathering. Novel "non-traditional" metal isotopes may be able to resolve this problem (e.g., refs. [325,326]; Section 4.4). In field settings, however, newly formed secondary phases are likely to represent small changes relative to pre-existing soil mineral pools, making them difficult to resolve analytically against natural background variability.

The amount of secondary mineral neoformation occurring during EW is largely unconstrained (e.g., ref. [325]). Natural basaltic weathering studies indicate that substantial fractions of Ca and Mg can be removed from solution by multiple processes, including cation exchange and secondary mineral formation, with clay formation representing an important sink in some settings.[327] Clay formation relative to primary mineral dissolution during basalt weathering has been reported across climatic gradients,[220,328–331] and weathering rates, and hence potential secondary mineral formation rates, generally increase at higher temperatures, although temperature also affects secondary mineral solubility and cation exchange equilibria.[57,184,222,223,332–334] These observations demonstrate that secondary mineral formation is a potentially important loss pathway. Nevertheless, natural basaltic weathering terrains are imperfect analogues for agricultural EW systems, where basalt is typically added as a relatively small fraction of the soil mass.[335]

Conditions that can promote secondary mineral formation include long water-rock contact times[327,336,337] and pH ranges favouring oxyhydroxides (<6), clay minerals (>6), or carbonates (>10).[333] Laboratory and groundwater studies further show that secondary phases can form over weeks to months under favourable conditions, including substantial Mg uptake into clays in basaltic groundwater and reprecipitation of dissolved basalt as clay in highly reactive laboratory systems.[338]. However, similar to natural analogues from basaltic catchments, these systems may overstate the extent or rate of secondary phase formation in field deployments, where feedstock additions are commonly only on the order of a few percent of soil mass,[335] hydrological flow-through can limit saturation buildup. Constraining the extent of these processes in real-world EW deployments—and identifying deployment strategies that minimise secondary clay formation through site, feedstock, and application-rate selection—remains a central uncertainty in assessing EW efficacy and a major focus of current research.

3.4.2 Cation exchange processes

Cation exchange processes exert a first-order control on whether base cations released during EW contribute to alkalinity export and realised CDR or are retained within soils. Upon silicate dissolution, cations such as Ca^{2+} and Mg^{2+} released to soil solution may be reversibly sorbed onto negatively charged mineral and organic exchange sites,[339–342] displacing exchangeable acidity (H^+ and Al^{3+}) as well as native cations. This process alters both the magnitude and composition of the exported solute flux relative to the initial dissolution stoichiometry and can suppress or delay net CDR through charge-balancing reactions that regenerate CO_2 if there is a net increase in BS.[68]

From a charge-balance perspective, net uptake of base cations by the soil exchange complex must be balanced by release of acidity. Where this acidity reacts with bicarbonate produced during silicate weathering, CO_2 is regenerated according to Eq. 1, effectively offsetting part of the initially generated alkalinity and CDR. This concept (see also Figure 15a) has been demonstrated in model simulations showing that transient cation retention at exchange sites can drive rapid conversion of bicarbonate to CO_2 until base saturation (BS) increases and exchangeable acidity is depleted.[68] Experimental and modelling studies consistently identify this exchangeable acidity pool as a dominant control on EW efficiency in acidic soils.[68,172,216]

Field and laboratory evidence indicates that the magnitude of losses associated with CEC can be substantial, particularly in highly weathered soils with large reserves of surface-bound acidity. For example, in a five-year tropical field experiment, 98% of the alkalinity lost to secondary processes was attributable to acids stronger than carbonic acid, with surface-bound protons constituting the dominant acidity source [216]. Despite sustained basalt dissolution and increases in soil pH and exchangeable Mg^{2+}, bicarbonate export in deep drainage remained minimal, demonstrating how large CEC-associated acidity pools can in some cases temporarily suppress realised CDR. Shorter-term experiments provide complementary evidence that cation exchange modifies not only the magnitude but also the composition of exported cation pools. For example, Mg and Li isotope tracers in olivine-amended soil columns reveal that a substantial fraction of weathering-derived Mg^{2+} is transiently retained on exchange sites, leading to isotopic signatures in pore waters that differ from those expected based on mineral dissolution alone [326]. These results demonstrate that

CEC processes can decouple composition of cations released via weathering from solute export fluxes, complicating inference of CDR from cation or isotope measurements alone.

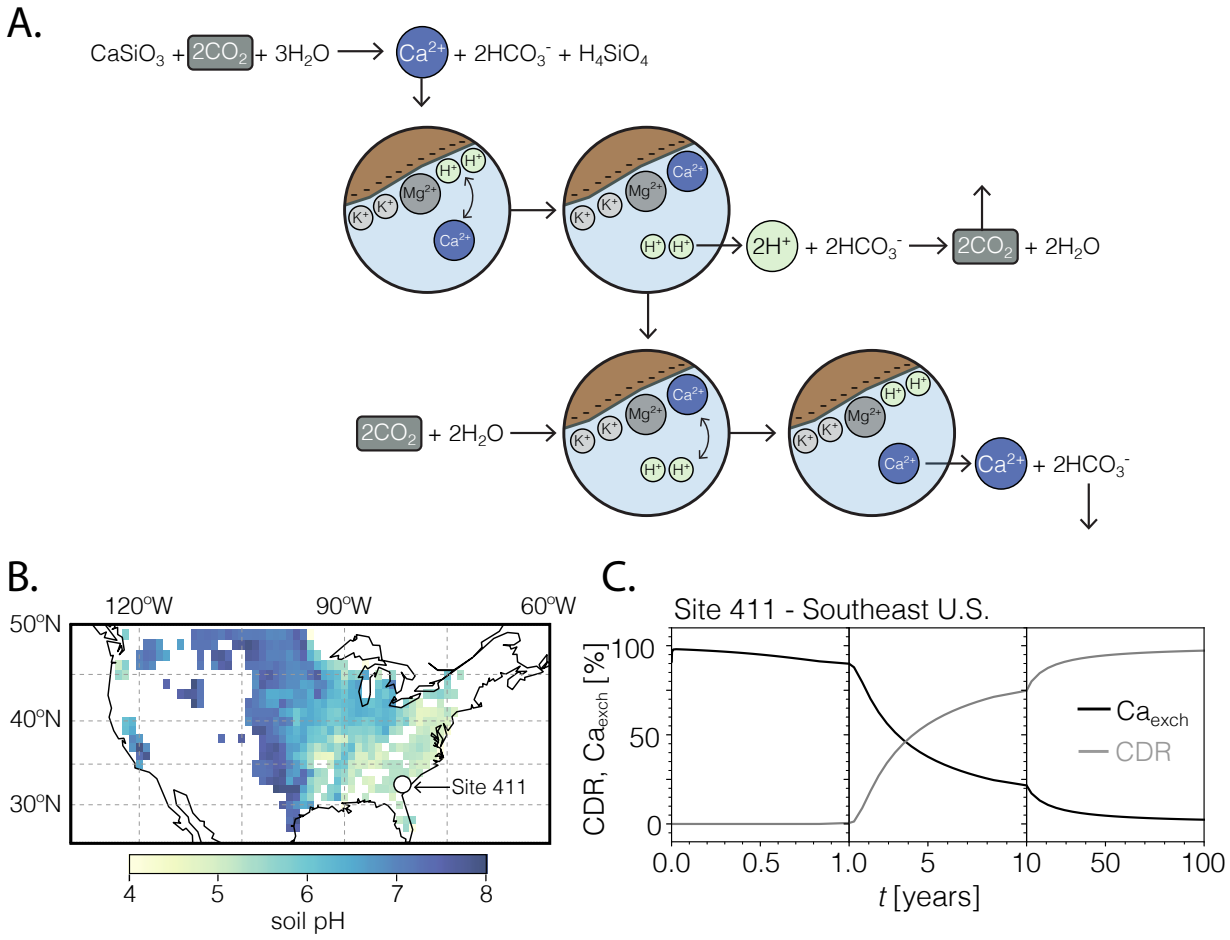

Figure 15: Influence of soil cation exchange on the temporal evolution of Carbon Dioxide Removal (CDR). A) When silicate minerals dissolve, divalent cations (e.g., Ca^{2+}) are released and initially balanced by bicarbonate formation. A portion of these cations can instead displace exchangeable acidity (H^+) from soil exchange sites, which consumes bicarbonate and temporarily offsets some of the initial CDR. Over time, these cations may be released again from exchange sites, allowing part of the removal to be realised later. B) Spatial distribution of soil pH across the United States for cropland grid cells at a resolution of 1°, illustrating background variability relevant for exchange reactions and weathering outcomes. C) Reactive transport simulations for a southeastern U.S. site (Site 411) showing (i) the proportion of calcium derived from basalt that resides on the exchange complex (Ca_{exch}) and (ii) CDR expressed relative to total potential through time. Exchange processes introduce delays in realised CR (feedstock., Section 3.4.6), and the magnitude of these delays differs among locations depending on soil chemistry, hydrologic fluxes, and mineralogical properties. Figure modified from ref. [68].

Whether CEC-associated retention represents a short (month to year) delay or a longer-term (decade scale) loss depends on soil chemical trajectories and acid inputs. For example, reactive transport simulations show that transient storage of Ca^{2+} on exchange sites can delay CDR realization by years to decades in US croplands,[68] with lag times varying by orders of magnitude across soils as a function of CEC, BS, and hydrology (Figure 15c, see also Section 3.4.7).

Experimental studies further show that Ca^{2+} and Mg^{2+} can remain sequestered on exchange sites over agronomically relevant timescales, delaying alkalinity and DIC export even when initial silicate dissolution is rapid (e.g., refs. [237,343]).

An additional complexity is that CEC is not necessarily a fixed soil property. In variable-charge soils and organic-rich horizons, exchange capacity can increase with pH as deprotonation of mineral and organic surfaces generates additional negative charge.[215] EW-induced alkalinisation may therefore expand the effective exchange pool, allowing a larger fraction of newly released Ca^{2+} and Mg^{2+} to be retained on exchange sites. This creates a potential negative feedback on cation export: greater weathering can drive larger pH and CEC responses, increasing cation retention and thereby amplifying CEC-associated delays or losses in realised CDR.

Cation exchange capacities and related properties vary widely across soil types, horizons, and management regimes**,** leading to strong spatial heterogeneity in EW outcomes. Highly weathered tropical soils typically exhibit large pools of reserve acidity relative to exchangeable base cations, increasing the potential for CEC-mediated neutralisation of alkalinity.[215,216] In contrast, soils with low CEC or high initial BS are more likely to translate cation release into rapid alkalinity export and realised CDR.[68] Where bicarbonate generated near the surface is neutralised at depth by acidity released from exchange sites, CO_2 may be emitted to soil gas or aquatic systems rather than transported to long-term marine reservoirs.

By controlling both the timing and magnitude of alkalinity export, CEC processes directly influence downstream CDR along the soil–river–ocean continuum. These processes provide a mechanistic basis for soil CO_2 leakage and lagged CDR frequently observed in EW studies and underscore the need to explicitly account for cation exchange dynamics when estimating net CDR, designing MRV frameworks, and interpreting field-trial data.

3.4.3 Strong-acid weathering

Carbon Dioxide Removal via EW is often quantified by linking mineral dissolution or base-cation release to stoichiometric uptake of CO_2 through carbonic-acid weathering. However, silicate minerals can dissolve via reactions that do not involve carbonic acid. Strong acids such as nitric or sulfuric acid can drive mineral weathering while producing no local CO_2 uptake, despite releasing cations that may otherwise be interpreted as evidence of CDR.[14,15,30,62,68,214,216,231,313,344] In addition, strong acids can react with bicarbonate produced during carbonic-acid weathering. Protonation of bicarbonate shifts DIC back toward dissolved or gaseous CO_2, effectively reversing previously realised CDR. Hence, when cation fluxes are used as a proxy for CDR without explicitly accounting for strong-acid inputs, these processes can lead to systematic overestimation of net CDR at the field scale.[325] However, as outlined above, strong-acid-driven dissolution may still contribute to CDR at a larger system scale if it reduces acid export from soils and the resulting weathering products are transported to more alkaline downstream waters.[345] In this case, CDR is not realised locally at the point of weathering, but may occur later as riverine or marine carbonate equilibria re-adjust,[346] with the timing and efficiency depending on local hydrology, solute retention, and mixing conditions.

3.4.3.1 Sources of acidity and the role of soil initial conditions

Strong acidity influencing EW systems arises from both ongoing inputs and legacy processes. In agricultural settings, nitric acid generated during fertiliser nitrification represents a major source of strong acidity, particularly where nitrogen inputs exceed crop uptake or nitrogen use efficiency (NUE) is low.[228,347,348] Sulfuric acid inputs can originate from atmospheric deposition or sulfur-containing fertilisers and can also contribute substantially to mineral dissolution without associated carbonic-acid consumption.[349–351] Some feedstocks, particularly when waste-derived, may also contain sulfur-bearing phases that produce acidity upon oxidation.[179]

In addition to active acid inputs, many soils contain substantial pools of accumulated acidity stored on exchangeable sites. Acidic soils with low BS and high cation-exchange capacity contain large proton pools, delaying alkalinity export and suppressing measurable CO_2 uptake via the release of acidity from exchangeable sites (cf., Section 3.4.2). In such systems, EW feedstocks may initially neutralise legacy acidity before contributing to net carbonic-acid weathering, making realised CDR strongly dependent on initial soil pH, BS, cation-exchange capacity, and land-use history.[68,172,216] Over time, depending on land-use history and ongoing acid inputs, many agricultural soils may reacidify as exchangeable cations equilibrate with soil waters. This can remobilise cations that were initially retained on sorption sites, contributing to delayed alkalinity export and CDR rather than permanent loss.

3.4.3.2 Approaches to assessing strong-acid impacts

Several complementary approaches are used to assess the impact of strong-acid weathering on EW-derived CDR. One method estimates fertiliser-derived acidity under the assumption of complete nitrification, sometimes combined with assumptions about NUE, and attributes this acidity to mineral dissolution that does not contribute to CDR.[352] Related approaches infer strong-acid contributions from measured nitrate and sulphate fluxes in soils, leachates, or catchment waters, allowing acid-driven weathering to be quantified directly from observed anion exports.[243,353]

Other approaches infer the presence and magnitude of strong-acid weathering from internal geochemical constraints rather than from external acid inputs. Deviations between observed soil pH and pH predicted from soil pCO_2 provide an estimate of the fraction of acidity—and associated mineral weathering—not attributable to carbonic acid.[231] At the solution or catchment scale, charge-balance approaches—building on earlier work assessing the CO_2 impacts of liming[354]—compare the sum of major base cations with bicarbonate alkalinity; any excess positive charge is attributed to strong-acid anions or to bicarbonate that has been consumed through reaction with strong acids, thereby quantifying lost alkalinity and associated CDR loss.[243] Changing geochemical conditions downstream (e.g., the loss of nitrate via denitrification) will also shift the charge balance of a system, increasing or decreasing the total amount of bicarbonate in solution and shifting strong acid impacts, albeit over variable and uncertain timescales.

3.4.3.3 Magnitude of strong-acid weathering losses

The magnitude of CDR losses via strong-acid weathering reported in experimental and modelling studies varies widely and is strongly controlled by soil acidity, fertiliser inputs, and system boundaries. In neutral or weakly acidified systems, strong-acid corrections are often small. For example, large-scale lysimeter experiments following basalt application on agricultural fields show similarly modest losses, on the order of 0.1–1.8% of potential CDR.[353] Similarly, inclusion of strong-acid corrections typically results in only minor downward adjustments to net CDR estimates in field trials using concrete.[299]

In contrast, substantially larger impacts are observed in acidic systems or where strong-acid inputs or soil acidity pools dominate weathering reactions. For example, controlled experiments combining silicate amendments with acidifying fertilisers show large increases in cation release without corresponding increases in inorganic carbon, demonstrating that cation-based proxies can substantially overestimate CDR under these conditions.[237] In acidic tropical soils in Australia, most mineral dissolution has been attributed to strong acids and exchangeable acidity rather than carbonic acid, with weathering dominated by surface-bound protons associated with exchangeable acidity (~45.9%), followed by nitric acid (~19.3%) and organic acids (~16.4%), while carbonic acid contributes only a minor fraction (~2.0%) of total dissolution; as a result, substantial mineral weathering can occur without commensurate CO_2 uptake, and apparent CDR may remain low until accumulated soil acidity is neutralised and BS increases.[216]

Overall, the literature indicates that strong-acid weathering losses are often small in neutral soils but can be large in acidic or heavily fertilised systems, especially prior to exhaustion of legacy acidity pools. However, the importance of strong-acid weathering depends on the spatial and temporal boundary used for CDR accounting. Although strong acids can substantially reduce initial or near-field alkalinisation efficiency, downstream re-equilibration under less acidic conditions may partly recover bicarbonate alkalinity, for example through acidity neutralisation, nitrate removal, or transport into waters with greater buffering capacity. This distinction is relevant for full-system alkalinity approaches,[345] but the magnitude, timing, and verifiability of any downstream recovery remain uncertain and may be difficult to align with project-scale MRV or crediting (cf., Section 4).

3.4.4 Biomass uptake

A portion of the base cations released through EW may be taken up by plants. While this can be a co-benefit of EW to farmers via increasing crop yields (Section 7.3.1), it negatively affects realised CDR. To maintain charge balance, uptake of base cations by plants is accompanied by a release of protons,[355] resulting in a direct loss of realised CDR. The limited number of studies that have estimated the proportion of base cation plant uptake relative to soil retention indicate that this is a minor loss term. A field experiment on maize/soy and miscanthus cropping systems reported that 1.1 and 4.5% of base cations were taken up by maize and miscanthus, respectively, with most of the remainder having leached to deeper subsurface soils.[353] Similarly, a mesocosm experiment lasting for 101 days found that plant base cation charge equivalents were about two orders of magnitude lower that those retained in the soil.[356]

In addition to the direct loss of realised CDR due to base cation plant uptake, there may be an indirect gain in CDR through stimulating ecosystem productivity and storing more CO_2 as organic carbon in vegetation or soils. This indirect CDR clearly dominates over the direct loss of realised CDR and may even be larger than the CDR directly resulting from EW. A global modelling study in which basalt was applied on sparsely populated, vegetated regions indicated that this productivity effect accounts for 32–80% of the realised CDR.[75] A modelling study conducted by [70] on a wollastonite-amended forested watershed impacted by acid rain found that the CDR by increased wood production was 1–2 orders of magnitude higher than that from mineral weathering. However, organic and inorganic CDR pathways differ fundamentally in durability, reversibility, and monitoring requirements: enhanced organic carbon storage is generally much less durable than inorganic CDR from EW,[2] and decomposition of additional biomass can re-release both carbon and retained base cations. It is therefore important that CDR estimates do not conflate organic and inorganic effects into a single undifferentiated value. In this review, we assess only the inorganic CDR effects associated with EW, which are linked to longer storage timescales and higher durability.

The fate of biomass determines the net impact of plant base cation uptake on realised CDR through EW and the timescale at which this is relevant.[220] Annual biomass turnover results in soil litter input, directly returning base cations to the soil when decomposed. Upon harvesting, crop residues added to the soil as well as root biomass can also result in quick biomass decomposition and cation release into the soil. Indirectly, most base cations removed through harvesting may travel through the food chain and be released again via animal and human faeces and detritus. However, the amount, timescale, and location of return is uncertain and will depend, for example, on whether cations are retained in wastewater treatment plants and where wastewater is released. However, one may assume that a portion of these base cations will re-enter ecosystems and, at that point, contribute to the realised CDR.

3.4.5 Surface passivation and reduction in rates

Researchers have shown that surface area-normalised weathering rates of minerals are slower in the field than in the laboratory, at least partly because such rates slow with the duration of weathering. For example, based on published EW data, rates of basalt weathering decrease by factors of 10–100 over 10 years.[357] Only a few very fast- or slow-dissolving minerals such as calcite, wollastonite and quartz have been reported to react at comparable rates in the field and laboratory.[71,358,359] The simplest explanation for the lab-field discrepancy is that the processes that limit the rate in larger systems do not occur in smaller systems nor in experiments [357]. Factors that decrease rates in field systems that are of interest to EW (i.e., over 1–100 y) can be roughly categorised as those that (i) change the surface of the feedstock; (ii) accumulate the weathering products without release to pore fluids; (iii) are hydrologic in nature; or (iv) are related to structural changes and system-level feedbacks.

Most practitioners predict the rate of CO_2 uptake into pore fluids based on the mass or surface area of feedstock added at time zero. However, rates may decrease over time due to changes to feedstock surfaces such armouring of surfaces by organics or inorganic precipitates or sorbates.[360,361] Similarly, the surface can be restructured by episodic wetting and drying[362] and the

number of reactive sites on a mineral decrease as natural fluids approach chemical equilibrium.[363] All of these processes slow dissolution by removing, occluding, or deactivating reactive sites.

Some factors related to the accumulation of weathering products without release to pore fluids are also related to mineral surfaces, but here we emphasize processes that decrease the rate as measured in leachate. When aqueous cations are not released to solution but instead accumulate on the soil exchangeable pool, in biota, or in precipitates, the rate of release of cations and production of alkalinity in the aqueous phase appears slow even if initial weathering may not. Such cation uptake can also be accompanied by proton release which may further decrease the uptake of atmospheric CO_2. Once the ion exchange pool saturates with respect to the amendment-released cations, or biota stop growing, further release of cations from EW and associated CDR equivalent to the true rate of amendment dissolution can be realised. Where secondary minerals precipitate, however, the mass of precipitate can increase and limit CDR indefinitely. Particularly, if carbonate minerals precipitate, then CO_2 is sequestered in the solid phase: this represents a net sink for atmospheric CO_2 if the amendment is a silicate, whereas the system is neither a sink nor a source for atmospheric CO_2 if the amendment is a carbonate.

Furthermore, the mass or surface area of the amendment may not always be the appropriate scaling factor for extrapolating rates to the field. Hydrologic effects can cause rates to be limited by the transport of products or reactants instead of surface reaction. In fully water-saturated soils, for example, transport of CO_2 may limit dissolution. Such factors do not affect the soil homogeneously because soils are not mineralogically, biogeochemically, nor physically homogeneous.[364] The wide variety of pore sizes in soils mean that macropores can channelize water, leaving the rest of the surface area coated only in a stagnant, chemically equilibrated fluid film. In addition to such spatial heterogeneity that can slow bulk rates, temporal heterogeneity also limits rates: wetting and drying exposes amendments to fresh, flowing water only for short periods of time.

Lastly, effects related to structural changes and system-level feedbacks in the soil—driven by coupling among transport and chemical reactions—can also reduce rates of EW in field systems. Such rate-inhibited coupling between chemical reactions and concentration gradients is often driven by biota.[365] In a macro-example, bioaccumulation of weathering products in plants eventually stops when plants die and decompose, releasing elements back to porewaters near the soil surface, where high concentrations can slow reaction rates.[366,367] The result of such spatial structuring of soil composition is that bulk reaction rates slow.

If all the rates in every pore could be modelled accurately for appropriate averaging across the entire porous medium of the soil for all points in time, the bulk-observed rates could be predicted in EW—but this is well beyond model capability. Ultimately, accurate rate prediction is difficult for any given natural system, and most practitioners have resorted to tuning the "hydrologically accessible reactive surface area" as a way to predict rates in natural systems.

3.4.6 Lag times

Time delays between EW deployment and the emergence of measurable alkalinity export or climate-relevant CDR are widely observed across laboratory, field, and modelling studies. These

lag times arise not only from limits on mineral dissolution, but also from constraints on the realisation of CDR after weathering has occurred (e.g., soil chemical buffering and secondary phase formation), such that alkalinity generation does not necessarily translate into immediate downstream alkalinity export and associated CDR.[68,172] In particular, a RTM-based study identifies temporary storage of weathering-derived Ca^{2+} and Mg^{2+} on soil exchange sites as a dominant mechanism generating lagged CDR, because exchange reactions can decouple mineral dissolution from immediate alkalinity export and CO_2 uptake (cf., Section 3.4.2; Figure 15c). These simulations demonstrate that alkalinity generation and realised CDR can be fully decoupled in time, even when feedstock dissolution is effectively instantaneous.[68]

Lag times can be further extended by temporary retention of alkalinity in secondary solid phases, including carbonate precipitates and secondary silicates, and by re-equilibration of DIC under conditions where soil or porewaters approach saturation. Such phases can sequester weathering-derived cations and alkalinity for extended periods, suppressing export and, potentially, realised CDR until renewed undersaturation, episodic acid inputs, or hydrological flushing drive their dissolution. Depending on environmental conditions, these processes may represent either permanent losses or reversible, time-dependent storage of alkalinity.[68,172]

Model results indicate that lag times between alkalinity input and realised CDR can range from less than one year to many decades, depending primarily on soil CEC, initial BS, target soil pH, and hydrological throughput.[68] In large parts of the U.S. Corn Belt and Great Plains, less than 50% of CDR potential is realised within the first decade after EW deployment, with multi-decadal timescales required to approach steady state, whereas soils with lower CEC and/or higher initial BS—particularly in parts of the southeastern United States—exhibit substantially shorter lags.[68] Consistent with these results, shorter duration laboratory and field studies frequently observe rapid silicate dissolution and increases in exchangeable Ca^{2+} or Mg^{2+} without corresponding short-term increases in alkalinity or DIC export.[184,222,223,237,326,343]

Long-term catchment-scale evidence for lagged alkalinity export is also provided by the wollastonite addition experiments at the Hubbard Brook Experimental Forest.[70–74,368] Wollastonite application led to rapid initial mineral dissolution, increases in soil pH, and large increases in exchangeable Ca^{2+} and BS in surface soils, yet the appearance of wollastonite-derived Ca in stream waters was delayed by several years—demonstrating substantial interim storage within soils and vegetation.[70,71,369,370] Mass-balance studies show that even 6–12 years after application, only ~2–5% of the feedstock Ca was exported in stream waters, while most Ca remained either undissolved (54%) or retained on exchange sites (44%) particularly in the organic horizon, consistent with strong CEC buffering in an acid-depleted system.[72,74] Elevated Ca:Si ratios (>2:1) in soil solutions further indicated non-congruent dissolution and/or post-dissolution immobilisation of Si via secondary mineral formation, implying that both cation exchange and secondary phase processes contribute to delayed solute export.[71,74] Even after Ca export increased, bicarbonate-based CDR remained suppressed for more than a decade because a large fraction of weathering-derived alkalinity was neutralised by strong acids associated with legacy sulphate deposition and ongoing nitrogen cycling.[70] As sulphate deposition and exchangeable acidity declined over time, the fraction of Ca^{2+} export charge-balanced by bicarbonate increased, producing a delayed but sustained increase in catchment-scale inorganic CDR.[70] While Hubbard Brook demonstrates how lag times can emerge from interacting processes such as cation exchange, biological uptake,

secondary phase formation, and strong-acid weathering, it represents an intentionally Ca-limited, strongly acidified forest system in which tree uptake constituted a major Ca sink and the wollastonite amendment itself was originally applied to alleviate stand-level Ca deficiency.[72,74] Consequently, the observed delays likely approximate an upper bound on lag magnitude relative to many agricultural soils with lower biological Ca demand that are less impacted by historical anthropogenic acidity.

Across systems, lag times are expected to be longest in environments characterized by low pH, high CEC, and large reserves of exchangeable acidity, as well as in settings where secondary phases are near saturation and hydrological flushing is limited. Seasonal variability in infiltration and redox conditions can further modulate lag behaviour by controlling the timing of cation remobilisation and alkalinity export.[68,172,252] These lag effects pose a fundamental challenge for MRV frameworks that assume a near-term correspondence between feedstock dissolution and climate-relevant CDR. Indeed, many short-term experiments capture systems in the early, lag-dominated phase where dissolution, cation release, and bicarbonate export are strongly decoupled, making long-term inference difficult. In the long term, translating such observations into robust CDR estimates may therefore require process-based models that explicitly represent cation exchange, secondary phase dynamics, strong-acid weathering, and hydrological transport.[68,252,371] At present, attribution of delayed CDR based on models alone is not MRV-ready and requires rigorous validation against long-term field datasets which are largely lacking.[68] In the interim, conservative crediting approaches must either discount delayed CDR ex ante or rely on ex post verification of alkalinity export along the soil–river continuum, explicitly accounting for lag effects.

An important contextual point is that delayed net CDR is not unique to EW but is a general characteristic of many CDR approaches, which often exhibit "payback periods" before net climate benefits emerge.[372] For example, LCA of biochar systems show that pyrolysis can generate an initial pulse of emissions that would otherwise have been released more gradually during natural biomass decay, such that net removal depends on both feedstock counterfactuals (i.e., what would have happened with the biomass in the absence of biochar production) and time horizon.[373,374] Similarly, chemical CDR systems such as DACCS can be net-emitting during early operational phases when accounting for construction, sorbent production, and energy supply, with net removal achieved only after sufficient operating time under low-carbon energy inputs.[32,375] These examples underscore that temporal asymmetry between deployment and realised climate benefit is a pervasive feature of CDR, reinforcing the need for accounting frameworks that explicitly track time-dependent net fluxes rather than assuming instantaneous removal.

One proposed approach for managing such temporal mismatches in EW crediting is to pair delayed, durable CO_2 removal with near-term mitigation of short-lived climate pollutants, particularly methane.[376] Because methane reductions generate rapid climate benefits while EW-derived CDR may be realised over longer timescales, such "stacking" could help bridge radiative-forcing gaps during lag-dominated phases of EW deployment. This approach would require careful accounting to avoid double counting and to ensure that methane reductions and durable CDR are tracked as distinct climate benefits.

3.4.7 Degassing in rivers and oceans

Waters passing through soils become enriched in CO_2, such that their concentrations could exceed thousands of parts per million by volume (ppmv). A soil water in equilibrium with 10^4 ppmv CO_2 gas will contain ~340 µmol kg^{-1} of aqueous CO_2 as expressed by Henry's law (at 25°C; ref. [377]):

$$[CO_{2\,(aq)}] = H_{CO_2}(T) \cdot pCO_2 \quad (10)$$

where $[CO_2]$ is the concentration of dissolved CO_2, $H_{CO2}(T)$ is the temperature dependent Henry's solubility constant, and pCO_2 is the partial pressure of CO_2. When this water is brought into contact with the atmosphere in drainage waters the partial pressure difference between the liquid and gas drives CO_2 degassing (i.e., evasion).

In most drainage waters, this evasion is not instantaneous, and many terrestrial waters contain elevated pCO_2 relative to the atmosphere, releasing CO_2 during transport to the ocean.378 While alkalinity is conservative and remains constant during CO_2 evasion, loss of dissolved CO_2 shifts carbonate equilibria toward higher pH and a greater fraction of DIC as CO_3^{2-} rather than HCO_3^-. It is helpful to conceptualise this using the charge balance shown below:

$$Ca^{2+} + 2HCO_3^- \rightleftharpoons Ca^{2+} + CO_3^{2-} + H_2O + CO_2 \uparrow \quad (11)$$

The overall effect is that some of the CO_2 converted to bicarbonate during weathering is lost during evasion, in addition to the CO_2 lost due to Henry's law equilibration with atmospheric CO_2. Zhang et al., (2025)[66] explore the evolution of this loss within North American watersheds by dynamically simulating the mass transfer of CO_2 into the atmosphere (f_{CO2}) which allows for the decrease of aqueous CO_2 through evasion and subsequent equilibration of the carbonate system (using [378]), as shown below:

$$f_{CO_2} = A \cdot k_{CO_2}\,([CO_2]_r - [CO_2]_{atm}) \quad (12)$$

where $[CO_2]_{atm}$ is the aqueous CO_2 concentration in equilibrium with atmospheric CO_2 and $[CO_2]_r$ is the aqueous CO_2 concentration in the river. k_{CO2} and A are the mass transfer coefficient (often referred to as 'gas velocity') and the interfacial area per unit volume of liquid, respectively, which impact the rate of outgassing. A general solution to this is included in Figure 16, which shows the cumulative loss over time or distance within a water course.

Figure 16 illustrates that initially losses are low[214] and tend towards, but do not exceed, 5% as the river degasses an initially elevated pCO_2. This is consistent with the findings of Zhang et al. (2025)[66] who show losses between 1-5% for North American catchments. This is not surprising given that full CO_2 equilibration may not occur in rivers even when the advection time exceeds the time for evasion.[379] Photosynthesis,[67] respiration,[380] riverine mineral dissolution, and secondary mineral precipitation (Section 3.4.1) can also impact the dissolved carbon concentration and alkalinity of rivers, potentially altering CO_2 evasion.

Long-term storage of carbon dioxide (Section 3.3) will predominantly occur in the ocean. Mixing of river water with alkaline saline seawater will further alter the carbonate species equilibrium governing the distribution in Eq. 1 and 2, such that more of the alkalinity will be occupied by

carbonate rather than bicarbonate, accompanied by a further loss of CO_2. While this thermodynamic effect varies across the ocean (<10% loss at high latitudes and >20% at low latitudes; ref. [214]), basin-scale mixing of surface ocean waters occurs over years, which means that an average value of ~15-20% cumulative loss (both rivers and ocean) is more appropriate for long-term storage.[37]

It is possible that perturbation of the carbonate system may require years to decades to equilibrate through air-sea gas exchange,[381] and that unequilibrated rivers discharged into the surface ocean may be mixed into deeper water without their alkalinity fully equilibrating with atmospheric CO_2. This may be the reason why lower losses are returned from EW simulations coupled to an ocean biogeochemistry model.[22,279] Similarly to rivers, secondary precipitation of minerals in the surface ocean may result in further evasion of CO_2 (Section 3.4.1).

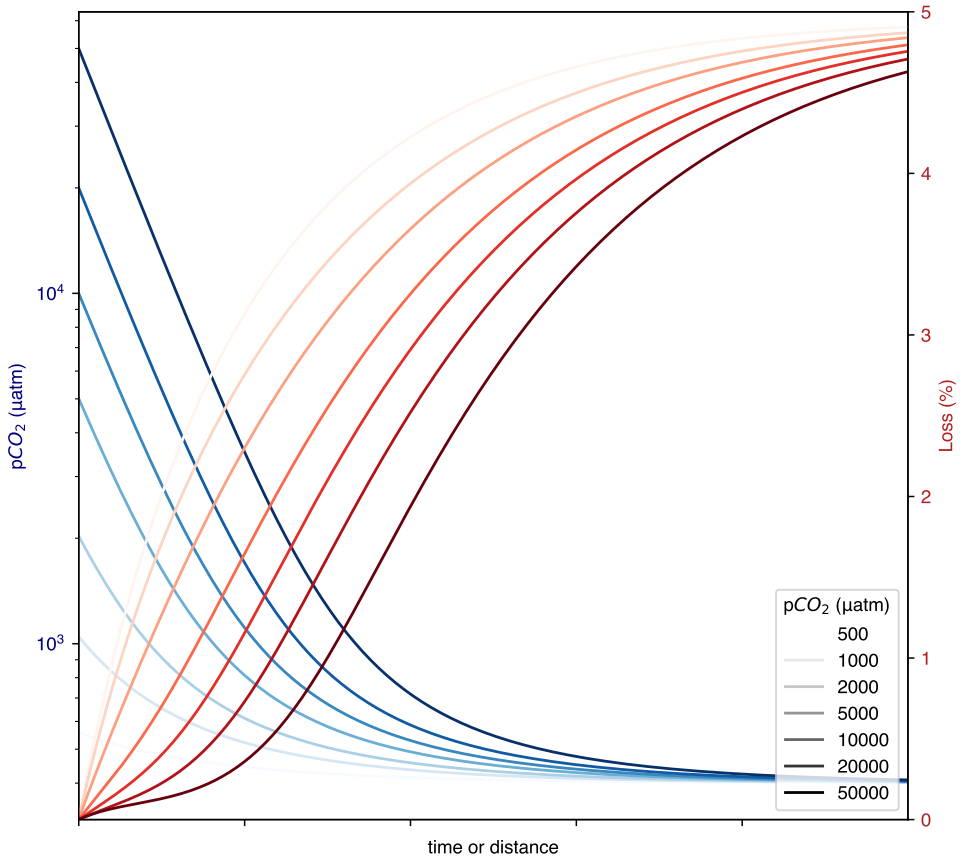

Figure 16: An idealised simulation of mass transfer from a drainage system following the approach of Zhang et al., (2025)[66] assuming a baseline and change in alkalinity of 1000 and 100 µEq kg^{-1}, respectively (at 25°C). Initial pCO_2 represents the water as it leaves the weathering environment, equilibrating to 400 µatm downstream.

3.5 Overview of ecological and environmental risks of EW

Enhanced Weathering combines elements of a long-established agricultural practice with features that are novel at scale. However, EW pursued for CDR differs in important respects, including the

broader range of feedstocks employed, deployment on soils that are not necessarily too acidic for optimal agronomic performance, and the intention to operate at spatial and temporal scales far beyond conventional soil amendment practices, all of which introduce additional environmental and ecological uncertainties. A comprehensive evaluation of environmental risks and corresponding mitigation strategies is available in prior syntheses (e.g., ref. [131]). The following section provides a high-level overview of key considerations relevant to deployment, while a more detailed empirical assessment of how frequently these risks have been observed in field trials is presented in Section 7.

3.5.1 Environmental risks

Many feedstocks proposed for EW contain trace contaminants, including potentially toxic metals.[69,129,150,248,382] Even when present at low concentrations in feedstock, repeated applications could lead to accumulation in soils to levels considered hazardous for crop health or food safety.[129,131,383–385] Scenario analyses based on measured feedstock composition and typical application rates therefore emphasize the importance of careful feedstock screening and, where necessary, limiting application frequency or duration to prevent exceedance of regulatory thresholds.[69,129]

At the same time, total concentrations alone do not determine ecological risk. Mineral amendments commonly raise soil pH, which can enhance sorption of certain metals to mineral surfaces and thereby reduce their bioavailability to plants and soil organisms.[109,386] Consequently, both accumulation and speciation should be monitored[131] and, where possible, modelled as part of EW assessment frameworks. If elevated metal concentrations arise, mitigation options include phytoremediation approaches,[150] adjustments to application strategies,[129] and stricter feedstock selection criteria.[248]

Potential downstream transport must also be considered. Fine mineral particles are susceptible to erosion and may be mobilised into surface waters, where associated metals could contribute to ecotoxicological risks.[162,180,387] However, export rates are expected to be site-specific and often limited, and geochemical changes induced by mineral additions—such as elevated pH or adsorption to reactive mineral phases—may in some cases decrease metal bioavailability in aquatic environments.[388]

Alteration of aqueous chemistry is more likely to occur through shifts in alkalinity and pH, processes broadly analogous to those produced by conventional liming. Enhanced Weathering is therefore expected to increase alkalinity export from treated systems,[17,65–67] with predominantly beneficial effects in historically acidified catchments.[131,179] Under such conditions, mineral additions may contribute to partial chemical restoration of acidified waters, although site selection criteria optimized for CDR could extend these interventions to regions where liming has not traditionally been applied (e.g., ref. [389]).

Airborne particulates represent an additional potential exposure pathway; fine silicate dust generated during comminution and field application poses recognised inhalation hazards.[131,383,387] Certain lithologies may also contain asbestos-forming or naturally radioactive minerals, underscoring the need for strict exclusion criteria during feedstock sourcing.[131,383,390] Practices,

protocols, and equipment for safely amending agricultural land with fine particulates exist as a result of longstanding agricultural liming practice, but additional care should be taken with fine silicate dust. Beyond site-level effects, life cycle emissions associated with extraction, processing, and transport—such as NO_x released during fuel combustion—must be incorporated into environmental assessments for large-scale deployment.[385]

3.5.2 Strategies for Mitigation, Monitoring, and Safeguards

Prior to deployment, careful site selection, feedstock screening, and baseline characterization can substantially reduce environmental risks.[131] Site selection should prioritize locations with the lowest inherent vulnerability and should include establishment of baseline conditions for key indicators, including soil chemistry and function, aquatic ecosystem status, airborne particulate concentrations, and contaminant levels relevant to food safety and human health. Feedstock screening should exclude certain mineral sources entirely—for example those containing asbestos or radionuclides[131,383,390]—while documenting concentrations of other constituents such as heavy metals to enable adaptive management through adjusted application strategies thresholds.[129,131] In addition, predefined safeguard thresholds for environmental indicators should be established based on existing regulatory standards for air and water quality, occupational exposure, and contaminant concentrations in soils, crops, and ecosystems.[129,131]

During deployment, risks to air and water quality can be reduced through established environmental-management practices. Covering mineral stockpiles during storage and transport, maintaining setback distances from waterways and residential areas, avoiding unnecessarily fine particle-size distributions where compatible with CDR performance, and implementing engineering dust controls such as moisture conditioning, enclosed handling systems, dust collectors at crushing or grinding facilities, and dust suppression on haul roads can substantially limit particulate emissions.[20,131,383,387,391–394] Personal protective equipment and operational constraints—such as enclosed vehicle cabs, reduced vehicle speeds during spreading, and avoiding application during dry or high-wind conditions—further reduce inhalation exposure to respirable mineral dust, including crystalline silica, which is associated with well-documented occupational health risks.[131,395–397] High-frequency or continuous environmental monitoring allows for adaptive management should thresholds be exceeded, compensating for unknown variables such as the actual bioavailability or leachability of heavy metal contaminants.[131] Monitoring can also target key areas of concern, such as fisheries for water quality or neighbouring schools for air quality.

Environmental risks can also be reduced through careful avoidance of high-risk deployment contexts. Sites with elevated background metal concentrations, whether from historical agrochemical use or geogenic sources, may be unsuitable because additional inputs could increase the likelihood of exceeding ecological or food-safety thresholds.[131] Sensitive ecosystems and downstream environments should likewise be excluded, especially those characterized by highly acidic soils such as wetlands or bogs that host specialized microbial communities with limited tolerance to geochemical disturbance,[398] as well as habitats dominated by dust-sensitive organisms including lichens, liverworts, and sphagnum.[399] Certain soil and climatic conditions can further elevate risk: poorly drained or fine-textured soils may favour secondary mineral accumulation that degrades soil structure and hydrologic functioning,[131] while arid or windy environments can

enhance dust generation and persistence that increases ecological and occupational exposure hazards. Incorporating these environmental constraints into site-selection frameworks therefore represents a primary preventative strategy for minimizing unintended impacts.[131]

Should exceedance of predefined environmental thresholds occur, mitigation measures should be supported through contingency funds held in reserve by project developers. Potential responses include targeted phytoremediation using hyperaccumulating plants to remove trace metals,[150] soil-management interventions such as organic amendments—including biochar[400,401]—to enhance sorption capacity and stabilise contaminants, or tillage and related practices to maintain soil structure and water-holding capacity.[400] Establishing financial and operational provisions for such responses in advance can help ensure that corrective actions are implemented rapidly and effectively.

Beyond project-level monitoring, evaluation of cumulative or spatially distributed impacts requires coordinated data sharing and governance across projects and jurisdictions. Because many environmental effects propagate downstream, protecting freshwater and marine ecosystems will often depend primarily on managing upstream inputs. Accordingly, response strategies for downstream impacts should be incorporated into project design from the outset, with monitoring frameworks structured to detect watershed-scale changes and enable coordinated mitigation where necessary.

3.6 Modelling approaches in EW research

Modelling has played a central role in EW research from its early conceptualization to more recent assessments of deployment potential, costs, and co-benefits. Across the literature, models are used for several partially overlapping purposes:

- simulating mineral dissolution kinetics and CO_2 consumption under idealized or site-specific conditions for individual deployments;
- representing soil, hydrological, and catchment-scale transport and transformation of weathering products;
- embedding EW into regional-to-global carbon cycle and Earth system frameworks;
- estimating life cycle emissions, economic costs, and system-level trade-offs for at-scale deployment; and
- estimating the contribution of EW to climate change mitigation and emissions/CDR pathways scenarios.

The diversity of modelling approaches reflects both the multi-scale nature of EW processes and the current scarcity of long-term, spatially resolved empirical datasets against which models can be comprehensively validated.

3.6.1 Simulating weathering reactions and CO_2 uptake

3.6.1.1 Reactive transport and geochemical process models

Many EW modelling studies employ RTMs or closely related soil geochemical frameworks to represent mineral dissolution, aqueous speciation, formation of secondary phases, and acid–base reactions in soils and porewaters, particularly when inferring large-scale processes and CDR potentials.[19,20,24,68,402] These models typically resolve mineral kinetics explicitly, often using rate laws derived from laboratory experiments, and track the evolution of soil and solution chemistry as a function of time, depth, and hydrological fluxes.

Common implementations include one-dimensional vertical RTMs based on existing chemical models[403] such as PHREEQC[108,125,404] and TOUGHREACT,[97] as well as more EW-tailored models such as SCEPTER,[24,68,110,232] and the LC3M models[19,20,22] amongst others. An alternative approach, SMEW, employs zero-dimensional models that focus exclusively on the upper soil layers prioritizing the temporal dynamics of ecohydrological fluxes at the expense of explicit vertical process representation.[252,371,405] All together, these approaches allow detailed interrogation of how pH, temperature, fluid residence time, and mineral surface area jointly control dissolution rates and alkalinity generation. Their principal strength lies in mechanistic transparency: individual processes (e.g., kinetic inhibition, secondary mineral precipitation, or cation exchange buffering) can be isolated and tested. Importantly, however, RTMs do not all include the same processes or represent them in the same way, such that the specific mechanisms that can be interrogated depend on model structure and parameterization. As a result, RTMs are often data-intensive and sensitive to parameters that are frequently poorly constrained, including reactive surface area, saturation thresholds for secondary phase formation, effective flow paths, vertical variability in soil physicochemical properties, and the treatment of biological feedbacks.[406,407]

3.6.1.2 Shrinking-core and particle-scale models

Most of the RTMs introduced above implement or include options for a shrinking-core–type formulation to represent the progressive dissolution of primary mineral grains, explicitly tracking particle size as dissolution proceeds. In addition to their use within RTMs, shrinking-core approaches are also applied as stand-alone models that do not explicitly simulate soil geochemical evolution. In such cases, grain size (or a grain-size distribution) is coupled to assumed area-normalised weathering rate laws to generate dissolution or weathering-rate trajectories under prescribed solution conditions.[185,408–412] These models are computationally efficient and well-suited for sensitivity analyses of grain size distributions and application rates. Their simplicity, however, comes at the cost of limited environmental realism. Most implementations assume uniform exposure to solution—neglecting aggregation and evolving saturation state increases that limit weathering rates—and therefore tend to represent upper-bound dissolution scenarios rather than field-realistic behaviour.

3.6.2 From soils to rivers to oceans: transport and fate of weathering products

To link in-field weathering to downstream CDR, EW models increasingly extend beyond soils to include catchment-scale river transport and ocean biogeochemistry. River network and mass-balance models are used to estimate the capacity of fluvial systems to export alkalinity and DIC as

well as to assess CDR losses along riverine transport.[17,22,30,65,275,379,413] These approaches highlight the importance of residence times, gas exchange, and carbonate precipitation in regulating net CDR efficiency. In parallel, some foundational assessments constrain storage efficiency in the oceans using thermodynamic equilibrium carbonate chemistry—placing first-order bounds on persistence and CO_2 outgassing without explicitly resolving kinetics or hydrologic routing upstream of delivery to the coastal ocean.[18,37,126]

At larger scales, ocean biogeochemical and Earth system models such as cGENIE, BIOGEM, and BICYCLE have been used to explore the long-term fate of alkalinity additions and their feedbacks on ocean chemistry and atmospheric CO_2.[17,20,22,24,279,413–415] Although these models have been indispensable for assessing large-scale durability and leakage risks, they operate at spatial and temporal resolutions far removed from individual EW projects. Many of the modelling tools developed for terrestrial weathering, nutrient cycling, and carbon–climate feedbacks in other contexts are, in principle, applicable to EW.[403] However, EW-specific parameterizations remain sparse, and cross-model consistency in parameterization, scenario testing, and interoperability remains a significant challenge.

3.6.3 Integrated assessment and life cycle models

Beyond process-based simulations, EW is represented in integrated assessment models (IAMs) such as GCAM, EPPA, and REMIND-MAgPIE, where it appears as a stylized CDR option with prescribed costs, efficiencies, and deployment constraints (see Section 11, e.g. refs. [23,78,416–419]). These models are essential for exploring system-level interactions with energy, land use, and climate policy but necessarily rely on simplified representations of weathering processes. Similarly, LCA and optimization models (e.g., refs. [250,420–430]) are used to quantify upstream emissions, energy requirements, and trade-offs associated with mining, comminution, transport, and application (see Section 8). While critical for bounding net CDR, these models typically treat weathering efficiency as an external input rather than an emergent property.

3.6.4 Model requirements and challenges

Across all model classes, a recurring limitation is the lack of long-term, spatially resolved validation datasets that capture the coupled dissolution, transport, and downstream fate of carbon and alkalinity. Many models are calibrated using laboratory kinetics or short-term field trials, then extrapolated across years to decades and from plot to national, continental, or global scales. Discrepancies between model predictions and empirical observations have been reported at multiple scales,[252] underscoring the need for coordinated field measurements designed explicitly for model testing and validation. Key data gaps include effective reactive surface area (and surface area evolution) under field conditions, hydrological residence times relevant for alkalinity export, rates and saturation state thresholds of secondary phase formation and dissolution, mineral distribution heterogeneity in feedstocks, and biologically mediated feedbacks.[252] Without harmonised datasets and a coordinated effort to transparently evaluate parametric and structural uncertainty across models, intercomparison between models remains challenging and apparent agreement may reflect shared assumptions rather than independent validation.

The diversity of EW models has not yet been matched by systematic model intercomparison efforts. Studies using different RTMs, shrinking-core formulations, or Earth system models often produce divergent estimates even under nominally similar scenarios. Harmonisation of inputs, boundary conditions, and reporting conventions would enable clearer attribution of differences to model structure versus parameter choice. Hence, despite their value, existing EW models are not currently suitable as stand-alone tools for MRV.[68,431,432] Most models are insufficiently constrained by observations, lack robust uncertainty quantification at project scales, and cannot independently verify that modelled CO_2 uptake corresponds to durable CDR. In practice, models are best viewed as complements to empirical monitoring, helping to interpret measurements, estimate unobservable and downstream losses fluxes, and test consistency across system boundaries. Until models can be routinely validated against long-term field and catchment-scale datasets, MRV frameworks that rely solely on modelled outcomes risk overstating confidence and underrepresenting uncertainty. This limitation is not unique to EW but is particularly acute given the complex and lagged nature of EW-driven CDR.

3.6.5 Experimental and model disparities

Estimating CDR from EW requires consistent quantification of three coupled elements: the amount of mineral that dissolves, the subsequent fate of weathering products within the soil–water system, and the interactions between weathering products and the soil–water–air carbon system. Accurately representing this sequence of processes in models remains challenging due to the diversity of soil chemical, biological, and hydrological processes that regulate weathering kinetics (many of which remain poorly constrained quantitatively), as well as the partitioning of released base cations among dissolved, exchangeable, biological, and secondary-mineral pools. Site-specific disparities between observations and their representation in models for a given component are therefore common and propagate directly into uncertainty in EW projections. Within this framework, the largest source of uncertainty is typically associated with weathering rates, for which observed-to-expected deviations span several orders of magnitude.[172,357]

Discrepancies between modelled and observed weathering rates represent a long-standing challenge in geochemistry.[119] Most mineral-specific weathering rate formulations account for climate effects through temperature dependence, soil acidity via pH, and saturation state with rate constants derived from laboratory experiments performed under abiotic, well-mixed, and far-from equilibrium conditions.[123] When extrapolated to field conditions, these formulations tend to overestimate dissolution rates across a wide range of minerals, environmental settings, experimental scales, and lithologies. Evidence for this mismatch in the EW context is reported in natural weathering analogues,[102] laboratory-scale soil experiments,[162,184] and field applications,[405] such that agreement between observed and modelled weathering kinetics is the exception rather than the rule.

Recent model–experiment comparisons further illustrate this variability.[252] Using the ecohydrological–biogeochemical model SMEW,[252] some experimental EW datasets could only be reproduced by reducing modelled weathering rates by up to two orders of magnitude,[98,162,177] whereas only one required no adjustment.[108] Subsequent comparisons with field data confirm this

range, with inferred deviations spanning from order unity[353,371] to reductions of approximately two orders of magnitude.[405] While other model–data comparisons, such as those based on the STELLA model,[433] indicate good agreement with extrapolated experimental results as well as with a field dataset,[353] these field estimates have been questioned due to large measurement uncertainty—suggesting that inferred CDR rates may not be statistically distinguishable from zero.[434,435]

Despite advances in RTMs, including increased emphasis on hydrological controls,[238,436,437] persistent discrepancies between modelled and observed weathering rates are generally attributed to rate-limiting processes in heterogeneous, multiphase soils that are not captured by agitated-flask experiments or standard dissolution formulations. These include diffusive limitations at mineral–water interfaces, incomplete or transient mineral–water contact, and surface passivation by secondary mineral or organic coatings.[172,438] Available evidence suggests that, when combined, these inhibitory processes often outweigh potential biotically driven enhancements of weathering rates in soils.[439,440] In addition, the spatial and temporal scales of analysis play a critical role: as integration scales increase, effective weathering fluxes tend to decrease.[357] Collectively, these findings indicate that weathering rates are strongly regulated by site-specific rock–soil interactions, limiting the generalizability of model-based estimates and motivating caution when estimating EW CDR potential without observational validation.[161,352]

Beyond uncertainties in weathering kinetics, additional uncertainty arises from the fate of weathering products—specifically base cations released during dissolution—and their contribution to CDR. This distinction underlies the difference between potential and effective CDR in soils. Measurements that quantify weathered material directly (e.g., soil-based approaches),[352,441] as well as model-based estimates of weathering fluxes,[19,20,23] provide upper-bound (potential) CDR estimates under the assumption that released cations fully equilibrate with the water–air system, yielding bicarbonate formation at near-stoichiometric efficiency (base cation:bicarbonate charge ratios of 1:1 in soils or ~0.8 in the surface ocean).[37,214]

Experimental evidence and soil biogeochemical observations indicate, however, that a substantial fraction of weathering-derived cations can be retained within soils through cation exchange, plant uptake, or secondary mineral precipitation, thereby limiting the fraction available for realising CDR and downward transport.[108,162,172,184] Bridging potential and effective CDR therefore requires either coupled measurements, such as leaching-water-based estimates for effective removal[442] combined with soil-based estimates for potential removal, or observational constraints integrated with process-based models. In this context, available evidence suggests that uncertainty associated with the partitioning of weathering products in soils, when represented explicitly, is smaller than the uncertainty associated with weathering rates themselves.[110,252] Additionally, recent studies indicate that EW applications alter SOC through biotic pathways,[224] even dominating the overall carbon balance in some settings. Accurately assessing total CDR therefore requires observational constraints and emerging coupled representations of inorganic and organic carbon dynamics.[92,371]

4 Monitoring, verification, and reporting

Monitoring, verification, and reporting are central to establishing EW as a credible, scalable, and durable CDR strategy. Demonstrating net CDR from EW at the field scale under real-world agronomic and environmental conditions may draw from a variety of methodologies. Field-scale MRV must resolve the fate of weathering-derived carbon across solid, aqueous, and gaseous pools while accounting for spatial heterogeneity in soils, hydrology, and management practices. Many of these approaches[325,383] and related quantitative measurements[192] have been reviewed in detail elsewhere. In this section, we provide a brief overview and develop a synthesis comparing different MRV approaches. We structure MRV around three primary domains: (i) the solid phase, including soil mass balance approaches, the formation of SIC, and changes in CEC and BS; (ii) the aqueous phase, encompassing alterations in alkalinity and DIC fluxes, shifts in cation export, and the use of electrical conductivity (EC) and resin devices; and (iii) the gas phase, addressing potential CO_2 and other GHG exchanges. We then evaluate isotopic approaches that can be utilized in MRV, including emerging isotopic tracer techniques. Finally, we evaluate spatial and temporal mismatches among these approaches and their implications for uncertainty and attribution.

4.1 Solid phase

Quantifying CDR in the solid phase is central to field-scale MRV of EW, as soils represent both the primary reaction environment and a potential medium for carbon storage. Solid phase measurements aim to capture mineral dissolution, secondary mineral formation, and the stabilisation of weathering-derived carbon within soil pools. This includes: (i) soil mass balance approaches to constrain elemental gains and losses associated with weathering; (ii) the formation and accumulation of SIC as paedogenic carbonates; and (iii) shifts in soil chemical properties such as cation concentrations adsorbed to exchangeable sites (usually CEC and BS based), which reflect changes in exchangeable base cations and soil buffering status. Together, these metrics provide complementary lines of evidence for tracking weathering progress, attributing carbon sequestration pathways, and evaluating the permanence and co-benefits of EW within managed field systems.

4.1.1 Soil Mass Balance

Soil mass balance approaches to MRV in EW are used to estimate the extent of chemical weathering of rock feedstocks that have been applied to soils.[231,243,280,335,343,352,435,439,441,443–445] These approaches involve measuring the elemental composition of the solid or mineral phase of a soil-feedstock mixture after a period of time, during which weathering is likely to have occurred, and calculating the loss of mobile metal cations (Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}) from this phase relative to the initial soil-feedstock mixture. This loss can be used to determine a potential amount of CDR from weathering if the assumption is made that these cations charge-balance (bi)carbonate alkalinity in solution.

The initial amount of feedstock relative to soil at individual sampling locations analysed for element loss can be estimated based on sampling immediately after spreading feedstock or can be

assumed based on known application and mixing. However, this is unlikely to be uniform in the field because of spatial heterogeneity in both feedstock addition and background soil composition and is susceptible to errors given bioturbation and soil erosion. Instead, the concentration of immobile—or detrital—elements that are enriched in the feedstock relative to soils (e.g., Zr, Ti, Ni, Cr) can be used as tracers for feedstock addition.[335,352,353,434,441,444,446] This approach builds on the well-established use in geochemical studies of natural weathering of mass transfer coefficients that normalise cations to immobile elements in order to compare a starting mass of parent material with an equivalent mass of soil, e.g. ref. [447]

The benefit of using soil mass balance as an MRV tool is that weathering signals are time-integrated, unlike gas- or aqueous phase approaches. However, there are several considerations when interpreting these results for the purposes of MRV of CDR. First, depending on the treatment of samples (e.g., not washing with a leaching agent such as ammonium acetate to remove soluble cations held at soil exchange sites) and the method of bulk soil measurement (e.g., X-ray fluorescence (XRF); aqua regia digest or hydrofluoric acid digest followed by inductively coupled plasma mass spectrometry (ICP-MS) or inductively coupled plasma optical emissions spectroscopy (ICP-OES)), cations already released through weathering held at exchange sites may be identified as still being present within the primary solid phase, or elements may not be measured at all due to the recalcitrance of the mineral phases they are held within. This can cause under- or overestimation of feedstock weathering, respectively. For example, total digestion of Zr-bearing phases often requires HF digestion at high temperatures and pressures.

Second, the signal-to-noise ratio for changes in elemental composition of soils in response to mineral feedstock addition and weathering can be low depending on the differences in composition between soil and feedstock endmembers, their relative amounts in the post-amendment mixture, and underlying heterogeneity in soil and feedstock composition.[335,352,435,441,446] In order to resolve EW using soil mass balance approaches, it is necessary to demonstrate that signal-to-noise for specific soils and feedstocks used in deployments can be overcome by appropriate choices in analytical technique,[352] sampling strategy, and data aggregation,[280,441,446] else values derived using this method will be unreliable. In general, a signal will be most easily resolvable the greater the amount of feedstock applied, the greater the extent of weathering, the more enriched a feedstock is in cations and immobile elements of interest relative to the soil, the lower the analytical uncertainty and soil heterogeneity, and the greater the number of samples and area over which the calculation is performed.[280,335,441]

Finally, the calculation for mobile element loss by solid phase mass balance is agnostic to the chemistry of weathering reactions, or the fate of weathered cations. This means that the contribution to weathering from acids other than carbonic acid, or non-acid chelating agents, is not known and must be constrained with additional measurements in order to relate cation release to potential CDR (see e.g., Section 3.4.3). This also applies to (bio)geochemical processes downstream of the site of weathering—such as ex situ clay or carbonate formation, degassing of CO_2 due to pH shifts, and incorporation into biomass—which may impact the realisation of CDR.

4.1.2 SIC formation

Soil inorganic carbon includes carbon present in mineral form, primarily as carbonate minerals, rather than as organic matter. This includes both inherited lithogenic carbonates and secondary or paedogenic carbonates formed within soils. Where new carbonates form from Ca^{2+} or Mg^{2+} and bicarbonate generated by silicate weathering, they can represent a net carbon sink because atmospheric or soil CO_2 has been converted into a solid carbonate phase. However, carbonate precipitation is less efficient than storage as dissolved alkalinity because it removes cations and alkalinity from porewaters and releases a fraction of the previously weathering-derived DIC as CO_2 during precipitation. Consequently, changes in SIC stocks can be used to quantify the solid-carbonate storage component of EW-derived CDR where the carbonate can be attributed to silicate-weathering products.[103,108,112,170,178,234,237,246,384,404,448–451] However, SIC accumulation does not capture dissolved bicarbonate export and should not be added to, or subtracted from, soil mass-balance or aqueous-flux estimates without a clearly defined accounting framework that avoids double counting. Soil inorganic carbon formation, redissolution, and downstream carbonate precipitation must therefore be accounted for explicitly in a full CDR budget.

Carbonate formation depends on feedstock characteristics, soil porewater chemistry, and hydrological conditions. It is favoured where Ca^{2+} and Mg^{2+} concentrations, DIC/soil pCO_2, and pH drive porewaters toward carbonate supersaturation, particularly under warmer and drier conditions that concentrate solutes via evaporation.[452] Under wetter conditions, cations and bicarbonate are more likely to be transported to deeper soils or groundwater, creating spatial and seasonal variability in carbonate formation and complicating sampling programmes. The presence of paedogenic carbonate before EW application is therefore a useful indicator that carbonate formation may be favoured at a deployment site. Local precipitation, typically as calcite, may also occur in soil microenvironments such as mineral–water interfaces, even where bulk porewaters remain undersaturated; however, such carbonates may be transient unless saturation is sustained.

Paedogenic carbonate formation may be favoured when using fast-weathering Ca-rich feedstocks, as these rapidly increase alkalinity and supply the Ca needed for calcite formation. For example, in a wollastonite EW trial in rice paddies in Northeast China, Wang et al. (2024)[453] measured SIC increases in near-surface soils (0–20 cm) corresponding to a solid-carbonate CDR of 1.20 ± 0.03 $tCO_2\ ha^{-1}$. Similarly, Khalidy et al. (2025)[103] used a wollastonite-rich feedstock in Southwestern Ontario, Canada, to track carbonate formation to depths of up to 1 m and reported SIC-derived CDR rates of 0.55 $tCO_2\ ha^{-1}\ month^{-1}$, with evidence for greater SIC accumulation in deeper layers. Because SIC often accumulates deeper in soil profiles (e.g., >1 m; ref. [100]), carbonate-based MRV may require deeper sampling than conventional near-surface approaches to avoid underestimating this storage pool.

Soil inorganic carbon is measured using a variety of techniques, including calcimetry, combustion, thermogravimetric analysis, and coulometry;[192,454,455] analyses that could be routinely performed by commercial laboratories. Although SIC measurement is simple and accurate, even small increases will correspond to significant CDR; however, these changes may not be discernible from background SIC, particularly in calcareous soils. The measurement of SIC for CDR quantification will be further complicated if aglime is applied in conjunction with EW amendments, which often contain carbonates,[125] and their addition could be misinterpreted as CDR.[161]

4.1.3 Changes in Soil Exchangeable Base Cations

Changes in soil exchangeable base cation concentrations represent one pool of weathering products generated during EW that can be used to infer minimum bounds on primary mineral weathering rates.[91,108,177,216,231,237,244,352,443] Increases in exchangeable Ca^{2+}, Mg^{2+}, K^{+}, and Na^{+} relative to a control or baseline reflect weathering-derived cations retained in the surface soil at the time of sampling, prior to export to deeper soil layers or incorporation into secondary mineral phases. The exchangeable cation pool is quantified using standardised extraction methods, most commonly employing ammonium acetate to displace base cations from exchange sites for analysis in solution.[456] Changes in this pool are determined by comparing measurements from treated and control soils, or by repeated measurements of the same soil taken before silicate application and after a period during which weathering has occurred.

Where losses of weathering products from the system can be confidently excluded, such as in short-term soil incubations[177] or pot experiments without drainage,[171,457,458] changes in soil exchangeable base cations can independently serve as a useful proxy for weathering—although plant uptake and secondary mineral precipitation must still be considered where relevant. In settings where leaching from the surface soil occurs, including field applications[216,243,444,459] and flow-through laboratory experiments,[91,92,98,108,237,343,352,439,460] differences in soil exchangeable base cations must be paired with approaches that quantify cation export from the surface soil in order to fully assess total cation release from feedstocks. These include soil mass-balance methods[335,352,441] and direct monitoring of cation fluxes in leachate or drainage water. In these contexts, exchangeable cation measurements ensure that retained weathering products are included in EW estimates, avoiding underestimation that may result by only considering cations exported to deeper layers.[343]

4.2 Aqueous phase

The aqueous phase also provides a critical pathway for tracking carbon mobilisation and export under EW, as dissolved products of mineral dissolution are transported through soil porewaters, drainage networks, and receiving waters. Field-scale monitoring of the aqueous domain focuses on quantifying changes in alkalinity and DIC fluxes, which directly reflect CO_2 consumption during weathering and its stabilisation in bicarbonate form. Complementary measurements of cation fluxes (e.g., Ca^{2+}, Mg^{2+}, K^{+}, Na^{+}) provide stoichiometric constraints on mineral dissolution rates and help distinguish weathering-derived signals from background variability. Electrical conductivity measurements can serve as a rapid, high-frequency proxy for changes in total dissolved ion concentrations, helping to resolve temporal variability and identify weathering-related solute pulses. In addition, ion-exchange resin devices act as passive sorbents for dissolved solutes moving through soil porewater, offering a time-integrated approach to quantifying cumulative solute release in soils and shallow subsurface environments.

These approaches share the requirement for independent constraints on water fluxes, which can be challenging given the episodic nature of soil and downstream water movement following precipitation events. Depending on the measurements used, particularly alkalinity versus cation

fluxes, aqueous-phase approaches can capture different subsets of upstream losses, although losses beyond the MRV system boundary may still occur (cf., Section 4.5). Nevertheless, they have the potential to establish the aqueous mass balance needed to attribute CDR, assess transport and durability downstream from initial weathering reactions, and reconcile field observations with solid- and gas phase measurements.

4.2.1 Changes in alkalinity/DIC fluxes

The production of HCO_3^- during EW in soils serves as the primary indicator for CDR effectiveness. Consequently, increases in DIC and TA are frequently monitored in soil pore water, leachate from soil cores, and streams and rivers, as indirect proxies for HCO_3^- levels derived from EW.[30,63,65,91,102,174,216,237,343,384,404,437,443,461–464]

Dissolved inorganic carbon in the aqueous phase includes dissolved CO_2, HCO_3^-, and carbonate CO_3^{2-} ions, with HCO_3^- as the dominant dissolved carbon species in fresh water with pH between ~6.3 and 10.3. Dissolved inorganic carbon is often measured by converting all the inorganic carbon species into CO_2 through acidification (e.g., carbon analyser).[237] Streams and rivers are often oversaturated with CO_2 due to terrestrial and aquatic ecosystem respiration, with temporal fluctuations which may bias the measured DIC.[442] In addition, the carbonate equilibrium system in water is dynamic, and can be complicated by the potential for CO_2 loss through carbonate precipitation and re-equilibration if samples are not properly stored and handled.[442]

Alkalinity represents the acid-neutralising capacity (ANC) of aqueous systems,[465] with HCO_3^-, and CO_3^{2-}, and OH^- as the primary relevant anions in the environment. The measurement of TA is typically performed through titrating the water to a pH endpoint using a strong acid (e.g., manually or with an alkalinity titrator) and is frequently reported as equivalents per mass or volume (eq kg^{-1} or eq L^{-1}), mg L^{-1}, or ppm of $CaCO_3$ equivalent.[192] However, several limitations identified in recent literature demonstrate that alkalinity is not always a perfect surrogate for bicarbonate.[220] Non-carbonate anions—such as organic acids—can contribute to ANC, potentially leading to an overestimation of CDR if additionality compared to control conditions is not strictly apportioned.[192] In addition, if a sample is not properly stored, carbonate precipitation may cause underestimation of the alkalinity level in the water.

The need for high-frequency sampling to capture temporal variability of DIC or alkalinity in the aqueous phase makes these methods resource-intensive.[442] Several studies investigated the use of EC as a cheaper proxy for alkalinity (cf., Section 4.2.3).[466,467] While these studies show a strong relationship between EC and ANC in controlled mesocosm settings, field lysimeter deployments suggest that low-cost soil EC sensors are not sufficiently reliable as standalone quantitative proxies for alkalinity concentrations and may have high failure rates; as with all aqueous-phase approaches, quantifying alkalinity export further requires water-flux constraints.[466]

4.2.2 Changes in cation flux

Base cations (Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}) released from silicates during EW can be mobilised in soil pore water and subsequently exit the weathering environment and transport into drainage water (e.g., as leachate measured via soil core or column experiments), groundwater, streams or rivers.[65,325] Monitoring concentrations and fluxes of these ions in the aqueous phase has been widely applied in laboratory and field studies to estimate EW and associated CDR rates.[65,91,108,158,162,179,184,223,237,243,246,343,352,384,468,469]

For every mole equivalent of cation released by silicates through carbonic acid weathering, one mole of CO_2 is removed and transformed into one mole of HCO_3^- (whereas the ratio is 2:1 for carbonates). By summing up the total cation concentrations in the aqueous phase within or exported from the weathering environment, EW and CDR rates can be calculated. These concentrations can be measured via ICP-OES or Ion chromatography (IC).

To identify the source of the weathering products, stable and radiogenic isotopes are sometimes integrated into cation analysis (see also Section 4.4). Strontium isotopes ($^{87}Sr/^{86}Sr$) have been applied to distinguish between cations derived from the feedstock and those from native soil and minerals. Peters et al. (2004)[71] utilized Ca/Sr ratios and $^{87}Sr/^{86}Sr$ isotopes to trace the dissolution of wollastonite in a watershed-scale experiment. Similarly, magnesium ($^{26}Mg/^{24}Mg$) and lithium ($^{7}Li/^{6}Li$) isotopes provide insights into weathering process and can further track the formation of secondary minerals.[326] These isotopes can be measured via multi-collector ICP-MS or thermal ionization mass spectrometry (TIMS) for high-precision isotopic ratios.

The aqueous cation-based method for estimating EW and associated CDR rates has several limitations that can compromise its accuracy. Firstly, this approach cannot distinguish whether weathering is driven by carbonic acid (which consumes atmospheric CO_2) or by strong acids (e.g., nitric acid from fertilisers) which release cations without providing a CDR benefit—potentially leading to significant overestimates.[237,344] Furthermore, a substantial portion of released cations is not immediately transported but is instead retained within the soil exchangeable pool or incorporated into secondary clays, causing a lag that leads to underestimation of actual feedstock dissolution rates.[91,223,343] Cation sorption and uptake via secondary precipitation makes measurement particularly challenging, as standard extractions may fail to fully account for exchangeable cations. Finally, depending on sampling location within the watershed, these methods cannot capture potential downstream cation precipitation as carbonate minerals in drainage waters, a process that can release part of the captured CO_2 back to the atmosphere.[192,246,442]

4.2.3 Electrical conductivity

Electrical conductivity reflects the integrated concentration of dissolved ions in solution and therefore provides a first-order proxy for the accumulation of weathering products in porewaters and drainage waters.[442,467] Because EC can be measured continuously with in situ sensors, it offers a potentially scalable complement to laboratory-based alkalinity measurements for monitoring EW-induced solute release. Under conditions where alkalinity and major cations covary during silicate dissolution, systematic relationships between EC and TA can emerge (cf., Section 4.2.1).[442]

In soils with low CEC, combined measurements of bulk EC and volumetric water content can further constrain carbonate alkalinity additions, although organic matter content, clay mineralogy, and pH-dependent exchange reactions strongly influence the EC–alkalinity relationship.[467] As a result, EC-based approaches require site-specific calibration and explicit treatment of ionic composition before being used to infer bicarbonate concentrations through geochemical speciation modelling (e.g., using PHREEQC or The Geochemist's Work Bench) and are likely to be most applicable in systems with relatively simple and stable solute chemistries.

4.2.4 Resin devices

Ion-exchange resin devices (e.g., resin cartridges/capsules, or self-integrating accumulators) act as passive sorbents for dissolved solutes moving through soil porewater, yielding time-integrated estimates of weathering product export (e.g., Ca^{2+}, Mg^{2+}, K^{+}, Na^{+}, and relevant anions) over weeks to months. The integrative nature of this measurement is quite useful, as it has the potential to eliminate the labour intensity associated with high-frequency porewater sampling and the corresponding uncertainties of aqueous sample storage, processing, and analysis. Resins also act passively, allowing for the capture of a more representative pool of soil water, as compared to tension lysimeters or rhizons. Resins have been used to quantify leaching in agricultural soils, demonstrating that they can capture seasonal cumulative signals, however, substantial within-field variability requires high replication and careful installation to preserve natural flow paths above the collector.[470,471]

A core limitation is hydrologic sensitivity: capture depends on solute delivery to the exchange surface, so dry conditions can suppress adsorption by limiting transport, while preferential flow can bypass the device if placement/geometry do not intercept dominant pathways. However, design and installation (i.e., high replication, large size, deep installation) can significantly reduce flow-field artifacts. Current designs explicitly target cumulative flux measurement (e.g., resin-column "passive flux meter" variants) to reduce flow-field artifacts. Nevertheless, these systems still benefit from co-located hydrologic techniques (water balance, lysimeters, or modelling) to convert accumulated mass to area-normalised fluxes when possible.[472]

For EW applications, resin approaches are conceptually aligned with other soil-integrating MRV methods but inherit similar challenges: strong spatial and temporal heterogeneity in dissolution and drainage, uncertainty in representativeness of sampled flow, and potential disturbance effects during installation. The only truly comparable empirical method is embedded zero-tension lysimeters, which come with their own host of issues (e.g., water availability, soil column disturbance, heterogeneity, preferential flow from installation, etc.). Evidence from ion-exchange membrane/probe deployments show that measured supply rates can be highly placement-dependent, reinforcing the need for standardised depth targeting, adequate replication, and site-specific calibration.[473] Finally, resin chemistry and extraction protocols can introduce bias (e.g., as dissolved ion sinks, resins can drive local mineral undersaturation and weathering). Given the remaining uncertainties in applicability of this technique with different soil conditions at this time, EW deployments utilizing resins will benefit from controls, paired porewater sampling where feasible, and validation experiments under relevant soil moisture and texture regimes.[474]

4.3 Gas phase

Gas phase MRV involves the measurement of the rate of GHG emissions to the atmosphere from soil. Greenhouse gases are transferred between ecosystems and the atmosphere as a result of biotic and abiotic processes in soil and plants. Gas flux monitoring enables the detection of changes induced by EW.[91,175,240,353] This is complicated by the complex spatiotemporal variation in gas exchange driven by the physical, biological, and chemical heterogeneity of soils at microscopic to ecosystem scales. This variability can make the separation of an EW signal from natural variation challenging. As a result, there is no established gas phase MRV methodology specific to EW, though secondary validation through gaseous monitoring has been previously approved by registries.

Quantifying gas fluxes offers valuable context of the ecosystem impacts of EW. CO_2 fluxes allow us to characterise the impact of EW on organic and inorganic carbon cycling. Organic carbon cycling is the major contributor to soil CO_2 efflux, making it difficult to rely on gas phase measurements to quantify inorganic processes.[325] However, the fluxes are useful for tracking whole ecosystem dynamics and monitoring the fate of SOC.[91,353,475] Another compelling reason for the use of gas phase measurements is that, unlike other MRV approaches, they can target non-CO_2 GHG emissions, particularly nitrous oxide (N_2O) and methane (CH_4).[83,106,157,476] Even small emissions of these highly potent GHGs could offset any benefits due to CDR. Therefore, the utilization of gas flux measurements remains a compelling method for inclusion in MRV packages.

Existing studies in the context of EW have largely used static, portable, or dynamic chambers that are installed on the soil in a field or in mesocosms.[83,177,353,477,478] Chamber measurements can be used for mechanistic studies and quantifying GHG budgets alike. Chambers can be opaque to exclude photosynthesis or transparent and are most insightful when coupled. In a static chamber approach, gas samples are manually extracted from the chamber over a short closure time and analysed on a gas chromatograph or using spectroscopy-based gas analysers (e.g., Picarro, Licor, Aeris, Gasmet). However, this discrete sampling approach has largely been replaced by static chambers coupled with gas analysers for high resolution in situ measurements over a chamber closure period. This approach benefits from the comparatively low cost and straightforward deployment inherent to static chambers but offers a more reliable flux measurement with real-time verification. There has been work done to resolve the inorganic weathering signal using in-soil CO_2 sensors as a secondary validation technique[479] but the context in which it is used and the applicability of the techniques across deployment contexts remain debated.[480,481]

Automatic, dynamic chambers that can be deployed long-term and run continuously have the potential to better account for temporal trends. Automatic in situ measurements are spatially limited due to the required proximity to a gas analyser and are associated with a significant cost. Micrometeorological methods such as eddy covariance can also capture field-scale gas dynamics at high frequency and produce suitable datasets for calculating GHG budgets.[353] This method requires a large, unobstructed measurement area on the hectare scale, free from farm infrastructure and other barriers, so that GHG fluxes can be quantified without undue influence from microscale wind dynamics. Eddy covariance measurements require significant technical and analytical skills in addition to having high installation and maintenance costs.

Recently, there have been promising developments for gas phase MRV in the context of EW. Low-cost sensors, remote sensing, and open-access analysis pipelines combined with increasing knowledge about the influence of EW on processes in soils may increase momentum around gas phase MRV approaches. In addition, isotopic signatures of GHGs can now be measured in situ. As the isotopic signatures of CO_2 differ between organic and inorganic sources, these measurements could provide new mechanistic insights. While gas phase MRV does not currently play a great role in EW research, there is scope for increased application.

4.4 Isotopic approaches

Isotopic methods involve the measurement of stable (e.g., ^{13}C, ^{15}N, ^{18}O) and radioactive (e.g., ^{14}C) isotopes as tracers, both of which are analysed using mass spectrometry, to quantify precise measurements of carbon sequestration. While most MRV strategies quantify elemental stocks and fluxes, isotope ratios encode additional information because they respond to both source mixing and process-driven fractionation.[482] Radiogenic isotope systems are primarily used for source attribution, since instrumental correction removes mass-dependent fractionation and leaves only source-controlled variability. In contrast, stable isotope systems preserve both source and process signals and are therefore powerful tools for identifying mineral transformations, secondary phase formation, and biogeochemical cycling (Table 2). Isotopic methods are typically more costly and analytically demanding than concentration-based measurements and thus are unlikely to form the backbone of routine MRV for EW. Nonetheless, they provide constraints on sources, pathways, and transformation processes that cannot be inferred from bulk chemistry alone and are therefore particularly valuable for mechanistic understanding and model calibration. In this sense, isotopes complement rather than replace conventional MRV approaches by adding information on the mechanisms that drive elemental fluxes, in addition to quantifying their magnitude.

4.4.1 Stable carbon and oxygen isotopes

Stable carbon and oxygen isotopic ratios ($\delta^{13}C$, $\delta^{18}O$) provide valuable constraints on the origin and transformation of carbon in soil waters and carbonate phases. If isotopic endmembers are well characterized, these systems can be used to apportion contributions from atmospheric CO_2 processed through respiration, carbonate amendments (e.g., agricultural lime), and lithogenic carbon.[483] Typical soil-respired CO_2 is strongly depleted in ^{13}C, allowing mixing models to distinguish biologically derived carbon from carbonate-derived inputs.[484] In the context of EW, $\delta^{13}C$ and $\delta^{18}O$ have been applied across a range of experimental and observational settings—including agricultural, engineered, urban soils, and natural analogues—to investigate carbon sources, sinks, and carbonate dynamics associated with reactive mineral amendments.[277,450,485–490]

4.4.2 Radiocarbon

Radiocarbon (^{14}C) provides a more direct fingerprint of atmospheric carbon incorporation into EW products, albeit at substantially higher analytical cost due to sample preparation and accelerator

mass spectrometry requirements. Carbonates precipitated from modern atmospheric CO_2 inherit a characteristic ^{14}C signature that decays predictably over time, enabling the residence age of newly formed carbonates to be constrained.[482] In contrast, geologic carbon sources such as bedrock carbonate or shale contain effectively "dead" carbon with no detectable ^{14}C, facilitating separation of newly captured carbon from background pools. When coupled with total carbon inventories, radiocarbon data can be used to estimate rates of atmospheric CDR by EW,[277,490] although interpretation requires caution, as isotopic exchange between dissolved and solid carbon pools and kinetic fractionation during precipitation can bias apparent ages or mixing proportions.[490]

Table 2: Overview of the use of isotope systems in the Enhanced Weathering (EW) literature landscape.

Isotope system	Primary MRV-relevant use	Processes/constraints resolved	EW-specific applications	Key general references
$\delta^{13}C$, $\delta^{18}O$ (stable C, O)	Carbon source attribution	Partition atmospheric vs biogenic vs lithogenic C; carbonate formation & redistribution	277,450,485–490	483,484
^{14}C (radiocarbon)	Identify atmospheric C uptake	Age of newly formed carbonates; separation of modern vs geologic C	277,490	482
$^{87}Sr/^{86}Sr$ (radiogenic Sr)	Cation source tracing	Silicate vs carbonate weathering contributions; lithology-specific fluxes	108,243,277,443	482,491
$\delta^{7}Li$ (stable Li)	Secondary mineral formation	Clay formation; incongruent silicate weathering	326	336,492,493
$\delta^{26}Mg$, $\delta^{25}Mg$ (stable Mg)	Carbonate & silicate transformations	Carbonate precipitation/dissolution; biologically mediated silicate weathering	326,438	301,494,495
$\delta^{30}Si$ (stable Si)	Silicate weathering pathways	Secondary silicate formation; congruent vs incongruent dissolution	496	497
$\delta^{88}Sr$ (stable Sr)	Secondary mineral processes	Adsorption–desorption; carbonate fractionation beyond radiogenic Sr	—	498,499
$\delta^{44}Ca$ (stable Ca)	Carbonate cycling & cation redistribution	Carbonate precipitation/dissolution; ion exchange in soils and weathering zones, Ca sources	—	494

4.4.3 Radiogenic strontium

Radiogenic strontium isotopes ($^{87}Sr/^{86}Sr$) are widely used tracers of weathering sources and fluxes.[491] Because ^{87}Sr is produced by radioactive decay of ^{87}Rb, silicate minerals exhibit isotope ratios that depend on lithology and age, whereas carbonates typically reflect contemporaneous seawater values and are less radiogenic.[482,491] Strontium behaves similarly to calcium during dissolution and transport, allowing Sr isotopes in porewaters and streams to fingerprint the relative contributions of silicate and carbonate weathering to cation fluxes. In the context of EW, $^{87}Sr/^{86}Sr$ has been applied across agricultural, mesocosm, and field-trial settings, and in multiproxy approaches, to support interpretation of amendment-derived weathering signals and associated carbon accounting.[108,243,277,443] A key caveat is that even small carbonate fractions in silicate feedstocks can disproportionately influence dissolved Sr isotope signatures due to rapid carbonate dissolution, especially early in weathering.[500,501]

4.4.4 Lithium

Stable lithium isotopes (δ^7Li) are increasingly used to investigate secondary mineral formation in soils and weathering environments. Unlike radiogenic systems, Li isotopes are strongly fractionated by low-temperature processes, making them particularly sensitive to mineral–solution interactions rather than simply source.[482,492] Lithium isotopes are especially responsive to clay formation and other secondary silicate phases, as Li is preferentially incorporated into newly forming clays, leading to characteristic fractionation between dissolved and solid phases.[336,493] In the context of EW, this isotope system offers a means to probe whether applied silicates dissolve congruently or are rapidly transformed into secondary phases that may affect alkalinity generation and CDR efficiency. Enhanced Weathering-relevant applications remain limited but emerging, including the use of Li-Mg isotope signatures to constrain silicate weathering and secondary mineral formation in soil experiments.[326]

4.4.5 Emerging stable isotope tracers

A growing suite of non-traditional stable isotopes—including Mg, Ca, Si, and stable Sr—offer opportunities to resolve EW pathways beyond simple source attribution. Mg and Ca isotopes are sensitive to fractionation during carbonate precipitation, dissolution, and ion exchange, enabling constraints on cation redistribution and secondary carbonate formation during weathering.[301,494,495] Silicon isotopes fractionate during secondary silicate formation and biogenic uptake, making them useful tracers of congruent versus incongruent silicate weathering and Si cycling.[497] Stable Sr isotopes complement radiogenic Sr by resolving secondary mineral processes and adsorption-desorption dynamics not captured by source-controlled ratios alone.[498,499] Although most foundational work on these systems has focused on natural weathering environments and fractionation factors, EW-specific applications are emerging. These include the use of Mg isotopes to investigate biologically mediated olivine dissolution,[438] combined Li-Mg isotope signatures to constrain silicate dissolution pathways in soil experiments,[326] and Si isotopes to track soil processes and cation exchange in EW mesocosms.[496]

4.5 Comparison of MRV approaches

The MRV approaches outlined above differ fundamentally in their spatial and temporal specificity as well as system boundaries (Table 3, Figure 17), the CDR loss processes they directly quantify (Table 4), and exhibit method-specific nuances related to analytes, sampling strategies, and analytical protocols.[325,383] As a result, divergence between MRV estimates obtained using different approaches at the same site does not necessarily indicate errors or flaws in the underlying protocols. Instead, such divergence often reflects the fact that different MRV approaches quantify distinct subsets of processes and operate under different spatial and temporal assumptions. Understanding these differences is therefore essential for interpreting, comparing, and integrating MRV estimates of CDR from EW.

A key distinction between MRV approaches lies in their degree of spatial integration.[325] Some measured variables are inherently location-specific, such as soil compositions or pore-water chemistry, and thus require explicit consideration of spatial heterogeneity when extrapolating from point measurements (often multiple) to larger spatial units such as fields. In contrast, other variables integrate signals over much larger areas by virtue of the system being sampled. For example, stream-water chemistry integrates the sum of biogeochemical processes occurring across an upstream catchment, such that a single measurement reflects spatially integrated signals at that scale. Similar contrasts exist for gas phase measurements, where in-soil sensors or chamber-based approaches reflect highly localised fluxes, while eddy covariance towers integrate fluxes over a much larger footprint.

Table 3: Summary of spatial and temporal integration, system boundaries, and additional notes for different monitoring, reporting, and verification (MRV) approaches.

	MRV approach	Spatial integration	Temporal integration	System boundary	Note
Soil	Soil mass balance	No	Yes (years)	Sampling depth	Weathering rate estimate, losses need to be accounted for separately.
	SIC	No	Yes (years)	Sampling depth	Does not include CDR due to alkalinity export. May not be stable carbon pool in all soils
	CEC & BS	No	Yes (weeks-months)	Sampling depth	No standardised leaching protocol. Cations can be exported between measurement intervals, causing erroneously low detected CDR.
Water	Pore water based	No	No	Sampling depth	Can be challenging to sample at required volumes. Installation of some forms of lysimeters may alter water flow paths & estimates of water infiltration.
	Stream water based	Yes (catchment)	No	Catchment of sample location	High temporal sampling frequency required, particularly during high discharge events where much of the flux may occur.
	Flux accumulating devices	No	Yes (years)	Installation depth	Limited academic record relating to EW applications. Potentially challenging to estimate water fluxes and ensure device installation does not alter flow paths.
Gas phase	In-soil sensors	No	No	Sampling depth	High spatial and temporal variability and signal-to-noise. May be dominated by organic C cycling.
	Gas chambers	Yes (chamber footprint)	No	Soil gas diffusion depth	High spatial and temporal variability and signal-to-noise. May be dominated by organic C cycling.
	Eddy towers	Yes (flux footprint)	No	Soil gas diffusion depth	High spatial and temporal variability and signal-to-noise. May be dominated by organic C cycling.

Monitoring, reporting, and verification approaches also differ strongly in their temporal integration.[325] Some measurements represent instantaneous or short-duration snapshots, such as gas fluxes or stream-water concentrations at a given time, whereas others integrate over much longer periods. Soil-based approaches, including soil mass balance and changes in exchangeable cations, typically integrate temporally over weeks to years, whereas water- and gas phase approaches often require high-frequency sampling to capture episodic fluxes associated with rainfall events and floods during which much of the export flux may occur as well as seasonal

hydrologic dynamics. The lack of temporal integration in some approaches can, in principle, be compensated for through high-frequency or continuous sampling, while the lack of spatial integration can potentially be mitigated through aggregation across large numbers of sampling locations.[280]

Relating measurements with different spatial and temporal specificity within a single cohesive MRV framework therefore requires interpolation or extrapolation, informed by an understanding of the spatial and temporal heterogeneity of the system and chosen MRV approaches. The scales at which such heterogeneity occurs vary between measured variables and depend both on sampling design—such as the footprint of a flux chamber relative to an eddy covariance tower or the volume of soil or water collected for analysis—and on the physical, chemical, and biological properties of the medium being analysed. For example, soil elemental composition commonly co-varies with mineralogy and texture,[215] which are themselves governed by soil-forming processes. Similarly, water- and gas phase variables often exhibit predictable temporal structure in response to seasonal changes in temperature, moisture, and hydrology, as well as to discrete events such as storms. In general, soil-based approaches tend to integrate temporally, while water- and gas-based approaches may integrate spatially, depending on measurement design (Table 3; Table 2; ref. [325]). Robust long-term CDR estimates therefore require either explicit integration across missing dimensions or sufficient sampling intensity to compensate for their absence.

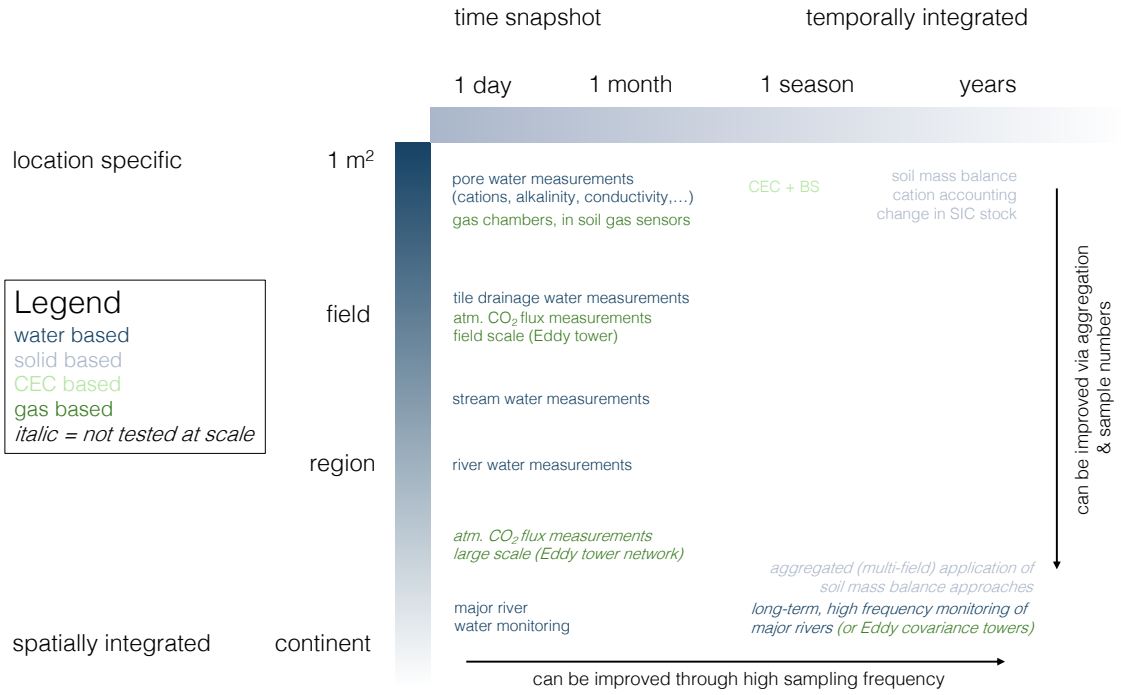

Figure 17: Sketch of temporal vs. spatial integration of various monitoring, reporting, and verification (MRV) approaches to quantify Carbon Dioxide Removal (CDR) from Enhanced Weathering (EW).

Direct comparison of MRV approaches is further complicated by the fact that different biogeochemical processes influence the variables measured by each method *within its system*

boundary (Table 4). Consequently, some processes that affect net CDR (and losses thereof; see also Section 3.4) from EW may be missed if only a single MRV approach is considered. Each approach has a characteristic profile of CDR loss processes that it captures within its system boundary, as well as processes that fall outside that boundary. For example, soil mass balance approaches cannot directly account for weathering driven by strong acids, whereas measurements of alkalinity in soil pore water can be used to quantify this contribution. As a result, internally consistent MRV estimates derived from different approaches may nonetheless reflect different subsets of the overall CDR budget.

Table 4: Approaches for monitoring, reporting, and verification (MRV) of Carbon Dioxide Removal (CDR) by Enhanced Weathering (EW), and whether they capture chemical processes constituting "loss" of initially removed carbon dioxide. PW = pore water

		Inorganic CDR losses, and if associated losses are captured by MRV approach *within MRV system boundary at the time of measurement* (see Table 3).								
	MRV approach	Strong acid weathering	CEC processes	SIC formation	Clay formation	OM stock change	Biomass uptake	Change in CH_4 & N_2O emissions	Degassing and losses in rivers	Degassing and losses in oceans
Solid	Mass balance	No	Yes/No*	Yes/No*	Yes	No	No	No		**Outside MRV boundary**
	SIC	Yes	Yes	Yes	Yes	Yes	Yes	No		
	CEC & BS	No	Yes	Yes	Yes	Yes	Yes	No		
Water	Cations PW	No	Yes	Yes	Yes	Yes	Yes	No		
	Cations stream	No	Yes	Yes	Yes	Yes	Yes	No	Yes/No**	
	Alkalinity PW	Yes	Yes	Yes	Yes	Yes	Yes	No		
	Alkalinity stream	Yes	Yes	Yes	Yes	Yes	Yes	No	Yes	
	Conductivity PW	No	Yes	Yes	Yes	Yes	Yes	No	No	
	Conductivity stream	No	Yes	Yes	Yes	Yes	Yes	No	Yes/No**	
	Flux accumulating devices	Yes (anion resin)/No (cation resin)	Yes	Yes	Yes	Yes	Yes	No		
Gas	In-soil sensors	Yes	Yes	Yes	Yes	Yes	Yes	Yes		
	Gas chambers	Yes	Yes	Yes	Yes	Yes	Yes	Yes		
	Eddy towers	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes/No	

* Depends on chemical treatment of soils and location of loss process.
** Degassing directly no, via alkalinity charge balance yes, carbonate formation yes.

An additional and closely related consideration is that any MRV approach is only sensitive to processes occurring within its defined spatial system boundary and up to the time of measurement. Spatial limitations are often intuitive; for instance, measurements of carbonate precipitation in topsoils do not account for carbonate formation occurring downstream in rivers. However, temporal limitations are equally important. Many processes affecting CDR from EW operate on

different characteristic timescales and can introduce substantial lag times between rock application and detectable signals. These include variability in chemical weathering rates driven by environmental conditions and negative feedbacks on weathering (e.g., refs. [172,357]; cf., Section 3.4), retention and storage of weathered cations on soil exchange sites prior to export,[68] hydrologic flushing of weathering products during transient rainfall events or seasonal changes in water storage,[502,503] and the loss of bicarbonate alkalinity in rivers and the surface ocean.[66,279]

Consequently, MRV approaches are primarily informative when their spatial and temporal system boundaries align with the scales over which measurable responses to EW can reasonably be expected. Comparisons of CDR estimates derived from different MRV approaches that operate over mismatched spatial extents or temporal windows are therefore of limited interpretability.[161] Rather than indicating methodological failure, such discrepancies often reflect differences in the timing, location, and processes captured by each approach. Taken together, these considerations underscore the need to view MRV approaches for EW as complementary, with robust CDR quantification requiring careful alignment of system boundaries, explicit consideration of spatial and temporal heterogeneity, and, where possible, integration of multiple lines of evidence.

4.6 Implementation of MRV practices in the Voluntary Carbon Market

4.6.1 Types of VCM MRV documents

Because EW appears capable of producing durable and measurable CDR at large scales, an array of private, public, and nonprofit stakeholders has emerged with the goal of integrating the latest scientific research into MRV methodologies and guidance documents for the Voluntary Carbon Market (VCM). Both types of MRV documents undergo private and public consultations for experts to review and comment. However, these documents are not peer-reviewed in the same manner as academic journal articles, and the governance during their construction is not always transparent (e.g., refs. [345,431,432,504–506]).

Monitoring, reporting, and verification methodologies (or protocols) are most often produced by carbon registries—organizations responsible for certifying that CDR has taken place and that a fungible carbon credit can be generated. These documents are binding, and a supplier of CDR (such as an EW company) cannot claim a creditable climate impact without meeting the methodology's criteria or receiving an exception from the registry. Private carbon registries have been the first to produce MRV methodologies for EW,[431,432,504,506] while nonprofit carbon registries currently do not have full publicly available methodologies.[507,508] Existing methodologies undergo minor and major updates (on monthly to yearly cadences), and it is possible that additional methodologies by new and/or existing registries will be produced in the near future.

In addition to registry methodologies, non-binding guidance, scoping, and synthesis documents have been produced by nonprofits, consulting firms, and jurisdictional governments. The goals of these are to systematically discuss current MRV practices, uncertainties, and opportunities, as well as to steer focus towards open MRV questions and opportunities for research and policy development (e.g., refs. [345,505,509]). While these expansive documents often substantially inform registry methodologies, they are not binding and do not currently appear to be updated on the same cadence. An exception to this update cadence is the Ithaka Institute (2022) registry

methodology.[506] Due to its age and the rapidly advancing state of EW research, most of the summary below is focused on "recent" MRV documents which have been published since 2024 and thus integrate the most up-to-date methods.

4.6.2 Structure and Content of VCM MRV documents

Monitoring, reporting, and verification documents follow a consistent method of calculating net CDR from an EW deployment by subtracting life cycle emissions and other carbon losses from the gross carbon drawdown that results from EW. Here, we briefly summarise the accepted MRV approaches from various VCM documents for quantifying gross carbon drawdown and carbon losses, but we do not discuss life cycle emissions. The focus is primarily on "required" MRV approaches, however MRV documents often "recommend" or "suggest" specific methods—these are noted when relevant.

Recent registry methodologies have adopted the framework from Mills et al. (2024)[505] of separating carbon quantification into a near field zone component (NFZ; the depth where initial CDR takes place and must be monitored empirically) and a far field zone component (FFZ; the spatial area beyond the NFZ where only carbon losses can be incorporated into net CDR calculations). Some methodologies require a primary MRV approach to be used as the main quantification method as well as a secondary MRV approach to be used for validation purposes.[431,432] For instance, when solid phase measurements are the primary MRV approach, a aqueous phase MRV approach must serve as secondary validation (and vice versa).

4.6.3 Solid phase MRV Approaches

Solid phase MRV approaches are accepted as the primary[431,432] or majority[504] quantification approach in all recent methodologies. Some recent methodologies allow for tracking the weathering of feedstock by measuring base cations either with or without the incorporation of an immobile trace elements[432,504] All recent methodologies require constraining NFZ losses by at least considering biomass uptake of cations, cation sorption onto soil particles, non-carbonic acid weathering, secondary carbonate formation, and secondary silicate formation.

For biomass uptake of base cations, all recent methodologies require that base cations must be measured in harvested and/or new growth biomass and converted into a total loss of CDR potential over a specified time period and deployment area. Cation sorption onto soil particles can be determined by measuring the product of CEC and BS over a specified time period for a deployment. This method is required in Rainbow (2025)[504], however in other recent methodologies it is only required after baseline sampling if alkalinity flux has not been constrained over the entire depth of the NFZ.

All recent methodologies provide the option of constraining non-carbonic acid weathering via direct measurement of anions in the aqueous phase, and some[432,504] allow for the use of a conservative proxy correction factor instead by following the methods in Dietzen and Rosing (2023).[231] Furthermore, some allow for estimation of non-carbonic acid weathering potential based on fertiliser application rates,[431,504] and Puro.earth (2025)[432] accepts indirect measurements derived by subtracting the sum of "bicarbonate + carbonate concentration" from cation concentrations.

Rainbow (2025)[504] also requires constraining pH-dependent speciation and acid buffering in addition to non-carbonic acid weathering.

Because recent methodologies primarily focus on the ocean as the long-term carbon sink for EW, secondary carbonate formation is mostly presented as a potential loss of CDR potential. Therefore, none allow for net new carbonate formation as a sole (or even primary) measure of CDR. In quantifying new carbonate formation as a loss term, as long as carbonates are retained during soil sample processing, directly measuring SIC is only required after baseline sampling in recent methodologies if alkalinity flux has not been constrained over the entire depth of the NFZ. However, some recent methodologies do allow for secondary carbonates to be counted as newly stored carbon[432,504] if SIC can be demonstrated to be a direct result of EW and can be shown to be stable over long periods. In contrast, empirically constraining secondary silicate formation as a loss term is not required in recent methodologies, since: i) an operationally viable method for early-stage commercial deployments is not available,[505] and ii) it is inherently corrected for if samples are taken over the entire depth of the NFZ.

4.6.4 Aqueous-phase MRV Approaches

Like the solid phase approaches, aqueous phase MRV approaches are accepted across all recent methodologies as the primary [431,432] or majority [504] quantification approach. All recent methodologies allow for constraining (at the base of the NFZ): i) the aqueous carbonate system, and/or ii) major dissolved ion concentrations. If measuring the aqueous carbonate system, two of the six parameters are always required to be empirically measured, with three parameters recommended in Rainbow (2025)[504] and four specific parameters (pH, TA, DIC, pCO_2) highly recommended in Isometric (2025). Puro.earth (2025)[432] presents two MRV options built around constraining either: i) TA while monitoring at least pH, or ii) measuring DIC alongside at least pH and temperature. If measuring major dissolved ion concentrations, the cation proxies used for CDR quantification (e.g., Ca^{2+}, Mg^{2+}, K^{+}, Na^{+}, etc.) must be justified and empirically measured, and non-carbonic acid weathering in the NFZ must be corrected for.

Methodologies state a preference for soil water to be sampled using porewater collection devices such as rhizons and/or lysimeters, however some allow for measurements on well-constrained catchment waters [431,504] or porewater extracted from soil cores.[432] Some methodologies allow for the use of ion exchange resins as a secondary validation approach[431,432] or on a case-by-case basis pending validation against other measurements.[504] However, resin-based measurements are not currently accepted as a primary or sole quantification approach. In recent MRV methodologies, registries note that they will continue to revisit ion exchange resins as the field develops.

4.6.5 Gas phase MRV Approaches

Gas phase measurements are not required, nor accepted as a primary form of CDR quantification in recent methodologies, although this does not preclude their potential use as a secondary validation method. Monitoring fluxes of CO_2 as well as additional GHGs such as CH_4 and N_2O is sometimes recommended via the use of gas flux chambers, eddy covariance towers, or in situ sensors (e.g., refs. [431,505]). The collection of high-resolution CO_2 gas concentration data alongside solid- and aqueous phase approaches has also been encouraged to strengthen cross-validation of

these methods.[505] The interaction between EW and soil GHG species concentrations is also identified as a key research and development area for commercial deployments and academic studies.[505]

4.6.6 Additionality and counterfactual baselines for EW in the VCM

A core requirement for crediting of EW in the VCM—and carbon accounting in general—is that credited CDR must be additional to what would have occurred under a plausible counterfactual scenario, i.e., the carbon drawdown that would have occurred in the absence of a deliberate EW intervention.[510] This ensures that actual climate mitigation takes place, and in practice requires demonstrating that net CDR resulting from EW exceeds the carbon uptake and fluxes associated with baseline (or business-as-usual) land management, feedstock handling, and natural weathering.

A major challenge for demonstrating additionality of EW arises from its strong overlap with existing agricultural practices, particularly liming and the use of silicate rock flour for soil conditioning, fertilisation, or pest control in some settings (e.g., refs. [29,47,511–514]). In situations where EW feedstock is substituted for conventional lime or other alkaline amendments, a portion of the observed alkalinity generation and associated CDR in following years may be due to these earlier practices rather than EW.[29,345] As a result, only the incremental carbon removal relative to displaced practices can be considered additional. Recent methodological assessments emphasise that baseline farming activity—including feedstock type, application rate, and application frequency—must therefore be explicitly characterised and reported via documents such as "Site Characterization Reports" rather than implicitly assumed to be zero (e.g., refs. [345,504]).

Additionality considerations also extend upstream to feedstock sourcing and handling. Quarry fines, industrial by-products, and demolition wastes proposed as EW feedstocks may already undergo passive weathering during storage, transport, or disposal. Moisture exposure, particle size distribution, and storage duration can enable some degree of baseline alkalinity generation and CDR prior to any deliberate EW deployment. Although these baseline fluxes remain poorly constrained empirically, they may be non-negligible in certain contexts, implying that conservative baselining assumptions or partial discounting of feedstock fractions may be required to avoid over-crediting (e.g., ref. [345]). Registry protocols incorporate these processes into net CDR calculation guidelines as "counterfactual weathering"[431,432] or "baseline feedstock CDR".[504]

Pre-existing in-field weathering and hydrological alkalinity fluxes further complicate counterfactual definitions, particularly for aqueous phase MRV approaches. Riverine and soil-water chemistry integrates both anthropogenic and natural signals, such that attributing observed changes in alkalinity or DIC to EW requires distinguishing project-induced fluxes from pre-existing activity and natural variability. Failure to adequately characterise these baselines and monitor variability—especially in regions with high inherent silicate or carbonate weathering rates—creates a risk of crediting CDR that would have occurred irrespective of the project.[345]

Taken together, these factors indicate that additionality for EW cannot be demonstrated through a single generic test but instead requires project-specific counterfactual baselines that integrate land

management history, amendment substitution effects, feedstock counterfactual uses, and natural geochemical fluxes. Current VCM methodologies address these challenges through combinations of practice-based eligibility criteria, financial additionality screens, quantitative baselining approaches, and monitoring of control plots (e.g.,ref. [431,432,504]). However, substantial uncertainty remains. Improving empirical constraints on baseline liming-related carbon fluxes, passive feedstock weathering, and natural alkalinity export is therefore a key priority for strengthening MRV robustness and safeguarding the environmental integrity of EW credits as deployment scales.

5 Carbon Dioxide Removal: fluxes, potentials, and the VCM

Building on the mechanistic understanding of EW and emerging MRV frameworks, this section assesses its quantitative contribution to climate mitigation. We distinguish between CDR fluxes, defined as area- and time-normalized removal (e.g., $tCO_2\ ha^{-1}\ yr^{-1}$); CDR rates, referring to total removal per unit time for a defined system (e.g., annual removal at national or global scales); and CDR potentials, which integrate assumptions over both space and time (e.g., cumulative global removal by 2100). We first synthesize CDR fluxes reported across the literature, then evaluate how these translate into regional and global rates and potentials. Finally, we situate these estimates within the evolving VCM, distinguishing between credit sales and realized deliveries to assess the role of markets in enabling and scaling EW deployment.

5.1 CDR fluxes

In this section, we focus on CDR fluxes expressed per unit land area and time. This framing reflects how EW is most commonly implemented and quantified at the field scale: silicate materials are applied over a defined land area, and weathering-derived carbon or alkalinity fluxes are subsequently inferred from soil, pore-water, runoff, or catchment-scale measurements integrated over time. As a result, per-area and time removal fluxes emerge naturally from experimental designs and are often used to compare findings with the literature as well as reviews of EW efficacy.[161] Importantly, area-normalised fluxes also serve as key inputs for scaling exercises, and some regional to global CDR estimates are derived by extrapolating such fluxes across assumed land areas and deployment scenarios.[245]

5.1.1 CDR fluxes across the literature

To enable comparison across studies, reported CDR estimates were harmonised to a common unit basis of $tCO_2\ ha^{-1}\ yr^{-1}$ using the spatial scales, durations, and system boundaries described in the original publications. Where studies reported CDR per unit time or deployment area only, fluxes were converted based on the reported experimental or modelling conditions. In a limited number of cases, additional assumptions—such as soil mixing depth or bulk density—were required where these parameters were not explicitly reported; given the ~6-order-of-magnitude spread in reported EW fluxes (Figure 18), such assumptions are unlikely to affect the identification of dominant patterns. The compiled dataset includes both potential CDR estimates derived from weathering rates and realized CDR estimates that account for loss processes. Where studies reported both, only the realized CDR after losses was retained; where only one estimate was available, that value was used. As a result, the dataset combines estimates derived under different system boundaries, MRV approaches, and assumptions regarding included loss processes (see discussion below).

Across the compiled literature, the dataset includes 511 unique CDR flux datapoints derived from 61 independent studies that report, or could be converted to, units of $tCO_2\ ha^{-1}\ yr^{-1}$. 466 of these datapoints have CDR fluxes larger than 0. These datapoints represent distinct combinations of study type, experimental or modelling context, feedstock characteristics, application conditions, and MRV approach and are shown in Figure 18. Together, this compilation captures the current quantitative landscape of EW CDR flux estimates across experimental scales, methodological

approaches, and deployment contexts. The median CDR flux across datapoints contributing to Figure 18 is 0.84 tCO_2 ha^{-1} yr^{-1}. Out of all experimental studies (field and laboratory studies), ~84% of all reported fluxes are above 0, with ~34% being below 0.1 tCO_2 ha^{-1} yr^{-1}, ~25% being between 0.1-1 tCO_2 ha^{-1} yr^{-1}, and ~33% being between 1-10 tCO_2 ha^{-1} yr^{-1}, though many of the higher estimates do not necessarily reflect realistic field conditions (see discussion below). This range aligns closely with earlier non-peer-reviewed synthesis efforts[515] as well as previous reviews.[161] Despite this wide range, central tendencies differ only moderately between modelling and experimental studies: median CDR fluxes are 1.10 tCO_2 ha^{-1} yr^{-1} for modelling studies and 0.55 tCO_2 ha^{-1} yr^{-1} for experimental studies, while arithmetic means are 3.12 and 3.77 tCO_2 ha^{-1} yr^{-1}, respectively, reflecting the influence of rare high-end values. These apparent differences are not statistically significant when comparing modelled and experimental fluxes either on raw values or on log-transformed positive values.

When placed in the context of broader discussions of EW deployment potential, the fluxes compiled here fall within—but generally toward the lower end of—the ranges often assumed to be feasible in large-scale assessments. For example, global projections of ~0.35–0.76 $GtCO_2$ yr^{-1} by 2050 and ~0.7–1.1 $GtCO_2$ yr^{-1} by 2100 have been derived by scaling a baseline sequestration flux of 1.5 tCO_2 ha^{-1} yr^{-1} according to local temperature and soil water surplus, with some tropical hotspots reaching ~4 tCO_2 ha^{-1} yr^{-1}.[245] Similarly, cumulative removals of ~64 $GtCO_2$ from agricultural hotspots and ~215 $GtCO_2$ when extrapolated to all croplands over the period 2006–2080 have been simulated using a soil reactive transport framework.[24] These projections correspond to modelled CDR fluxes of ~1.2–1.6 tCO_2 ha^{-1} yr^{-1} across favourable cropland regions in Africa, Thailand, eastern China, and India, around ~1 tCO_2 ha^{-1} yr^{-1} across major agricultural regions of North America, South America, and Europe/Russia, and <0.6 tCO_2 ha^{-1} yr^{-1} in arid regions such as Australia.[24] The empirical and modelling fluxes compiled here overlap with these assumed magnitudes but tend to cluster toward lower values and exhibit substantial variability across study designs and environmental settings. Moreover, not all compiled fluxes fully account for loss processes or transient early weathering pulses (see below), which may further widen the gap between reported plot-scale fluxes and the sustained fluxes assumed in large-scale EW simulations (see Section 5.2).

The highest experimental CDR fluxes appear to cluster in studies where conditions are expected to favour unusually high realized CDR fluxes, for example by using high feedstock application inputs. The two field-scale outliers are engineered/artificial soil systems using Ca-rich construction-derived materials or engineered substrates, with CDR inferred from inorganic carbonate accumulation.[448,450] The laboratory data exceeding 10 tCO_2 ha^{-1} yr^{-1} are likewise dominated by short-duration pot or column experiments using highly reactive feedstocks, relatively high amendment loadings, humid or intensively irrigated conditions, elevated CO_2 or otherwise optimized reaction environments, that do not (fully) take into account loss processes beyond the system boundary, and/or use chemical reagents to mimic chemical weathering.[63,177,237,244,246,449,469] Thus, the >10 tCO_2 ha^{-1} yr^{-1} values should be interpreted primarily as high-end estimates from favourable or engineered settings, rather than as representative central estimates for field-scale EW deployment.

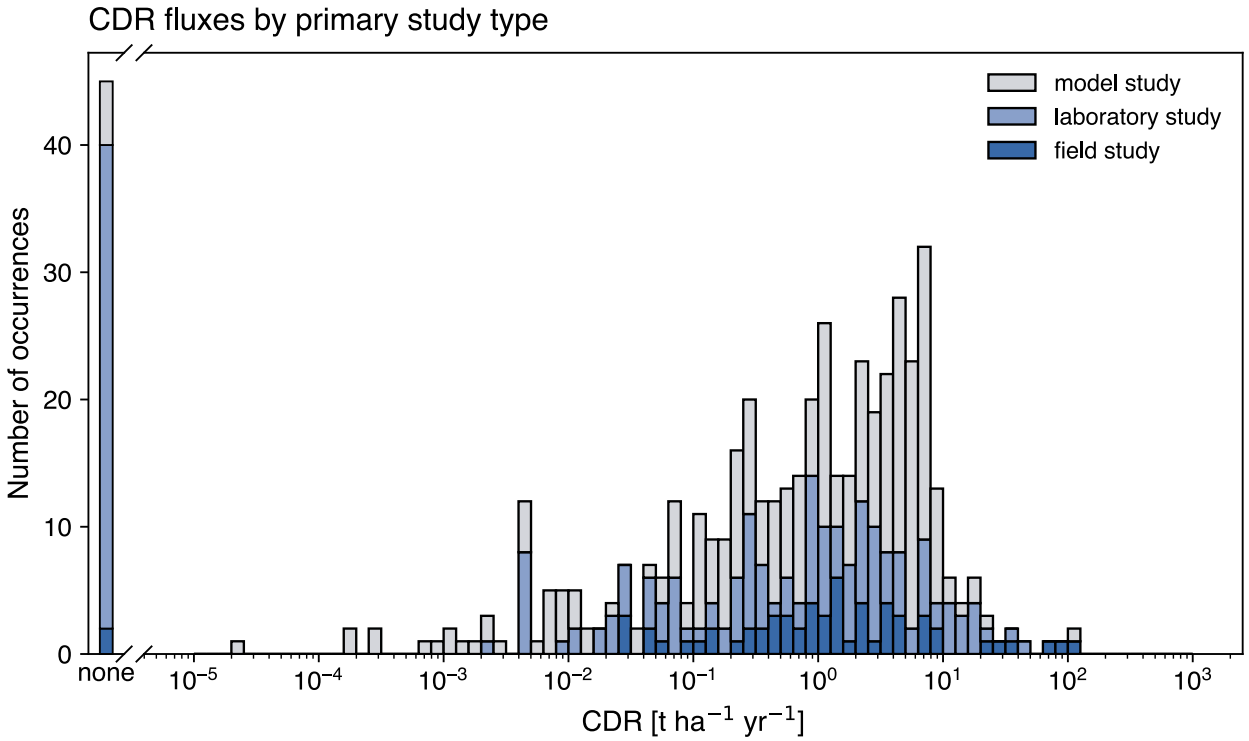

Figure 18: Distribution and scale dependence of Enhanced Weathering (EW) Carbon Dioxide Removal (CDR) fluxes: Overview of harmonised EW CDR fluxes expressed in tCO_2 ha^{-1} yr^{-1} across laboratory, field, and model studies. Positive CDR estimates are shown on the logarithmic axis after filtering out within-study duplicates, while zero and negative CDR estimates are grouped in the "none" bar.

5.1.2 Key limitations of CDR flux synthesis and literature estimates

Enhanced Weathering is governed by a highly coupled set of physical, chemical, and biological processes (see Section 3.2), such that observed CDR fluxes reflect the interaction of primary mineral dissolution kinetics, soil hydrology and chemistry, biological cycling, feedstock properties, application strategies, and the MRV approaches used to quantify removal (Section 4). Consequently, variability in reported EW fluxes reflects the combined influence of multiple interacting parameters that differ across studies and deployment contexts, such that attributing CDR outcomes to any single variable—such as feedstock type, application rate, MRV approach, or study design—necessarily oversimplifies the system. When data from many sites and experiments are compared simultaneously, the scatter introduced by unaligned deployment details and process regimes can obscure underlying parameter dependencies, such that genuine trends or causal relationships may not be apparent in cross-study comparisons broken down by a single parameter, even when they exist. In contrast, controlled laboratory or field experiments that isolate individual variables while holding others constant can often resolve such dependencies more clearly, even if their results are not directly transferable to larger spatial or temporal scales as well as conditions outside of those explored within the experiment.

The flux compilation presented above (Figure 18) carries several important limitations. Most notably, it aggregates estimates derived from fundamentally different MRV approaches and modelling frameworks. As discussed in Section 4, these approaches differ in the extent to which

reported fluxes represent potential CDR associated with initial weathering versus net CDR after accounting for post-weathering losses (or only a subset thereof). Consequently, the compiled fluxes combine fundamentally different definitions of CDR, reflecting divergent assumptions about system boundaries and timescales: weathering-derived CDR may be realized progressively over time, whereas changes in export fluxes can reflect more immediate CDR signals. These conceptual differences can translate into order-of-magnitude variation in reported CDR fluxes across both modelling[68,437] and experimental studies.[216,343,445]

Another key limitation of both the existing literature and the synthesis presented here is the extreme heterogeneity in temporal and spatial scales over which EW CDR fluxes are derived. Large spatial extents and long durations are almost exclusively captured by models (Figure 19), whereas experimental studies are largely confined to small spatial scales and short durations. While this reflects the essential role of models in exploring scales not accessible to current experiments (see also Section 3.6), it also introduces fundamental challenges for comparability and interpretation across approaches.

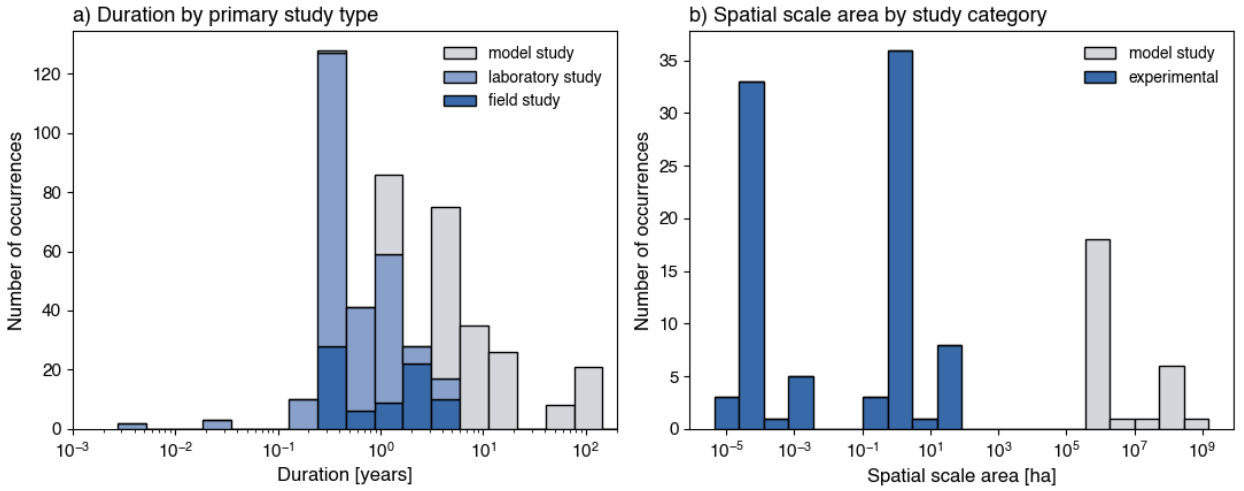

Figure 19: Distribution of study duration and spatial scale with experimental studies grouped. (a) Distribution of study durations (years) by primary study type (field, laboratory, and model studies). (b) Distribution of spatial scales (hectares), where field and laboratory studies are combined into a single experimental category and compared against model studies. Values reflect the subset of studies for which duration and spatial extent was extracted from the literature.

Extrapolating short experiments to annual removal fluxes, unless accounted for through comprehensive geochemical modelling,[252,371] assumes linear scaling of weathering fluxes over time, an assumption that can overestimate long-term CDR fluxes when applied to short experiments, particularly where early-time dissolution kinetics [116] driven by fresh mineral surfaces and availability of fine particles[161,172] dominate observed fluxes. As highlighted in recent non-peer reviewed synthesis work,[515] such linear extrapolation is widespread in the primary literature and reflects a structural challenge rather than an analytical choice unique to this review. Accordingly, where studies reported CDR per unit area without explicit annualization, our harmonisation

procedure likewise converts these values to $tCO_2\ ha^{-1}\ yr^{-1}$ based on the reported study duration, inheriting the same assumptions and limitations as the underlying literature.

A similar limitation applies to spatial scaling. Experimental systems are typically conducted at small spatial extents (Figure 19b) under well-mixed and highly controlled conditions, whereas field systems integrate across heterogeneous flow paths, mineral surface accessibility, and transport regimes that evolve with scale. As a result, apparent weathering rates tend to decrease with increasing spatial extent, reflecting the growing influence of transport limitation, hydrologic variability, and system heterogeneity, such that direct extrapolation of fluxes from plot- or reactor-scale studies to field or regional scales may likewise overestimate sustained CDR if these scale-dependent constraints are not explicitly represented.[116]

A further limitation is that reported CDR fluxes are not always accompanied by sufficient information on application rate, cumulative applied mass, or the elemental basis of the CDR calculation. This is important because identical soil, climate, and feedstock conditions can yield very different area-normalised CDR fluxes depending on the amount of material applied. Moreover, many studies report elemental release, alkalinity generation, or partial cation-based estimates rather than complete CDR fluxes, and some calculations consider only Ca and Mg rather than all relevant alkalinity-generating major cations. As a result, apparent differences in reported CDR fluxes may partly reflect differences in dose, reporting conventions, and calculation boundaries rather than true differences in EW performance.

Fully disentangling the drivers of EW CDR across the literature will ultimately require a dedicated meta-analysis employing approaches such as mixed-effects modelling, Bayesian inference frameworks, or machine-learning-based pattern recognition to account for hierarchical structure, collinearity, and context dependence across studies. However, implementing these methods lies beyond the scope of the present review and is further constrained by inconsistent reporting practices and incomplete metadata across studies (see section 3.2).

5.2 Global and regional CDR rates and potentials

This section examines EW potential across three linked dimensions. It first clarifies how different types of potential—particularly technical, economic, and sustainable potential[1]—can be understood in the context of EW, and why many existing estimates do not fit cleanly into these categories. It then reviews the range of currently available global and regional rate and potential estimates, highlighting how these depend on model structure, process representation, and deployment assumptions. Finally, it moves beyond climate-centric optimization to highlight that EW performance is governed by multiple interacting parameters and objectives, such that no single globally optimal deployment environment is likely to exist and system-specific strategies may can successfully optimize for various outcomes.

5.2.1 Types of potentials in the context of EW

Estimates of the potential scale of CDR are commonly categorized using standard assessment terminology developed in climate-mitigation research. Three categories are typically distinguished: technical, economic, and sustainable potential.[1] Technical potential refers to the

quantity of CDR achievable under biophysical, geochemical, and technological constraints while assuming deployment is unconstrained by economic, policy, or social considerations. Economic potential represents the portion of technical potential that remains once costs, infrastructure, market responses, and system-wide trade-offs are considered. Sustainable potential further restricts deployment by incorporating environmental safeguards and broader governance or societal constraints. In the EW literature, an additional category sometimes reported is the CDR potential implied by available feedstock resources, which quantifies the theoretical CO_2 uptake associated with the total mass of suitable rock assuming complete dissolution by carbonic acid.[18,133,410] These estimates provide an upper-bound resource perspective by assuming complete dissolution with carbonic acid[18,37,126] but do not represent deployable removal because they abstract from kinetic limitations, hydrologic transport, and environmental feedbacks that determine whether dissolution products translate into durable carbon storage.[325]

For EW, the distinction between these potential categories reflects the wide range of processes controlling whether applied rock ultimately results in net CDR. Estimates of technical potential must account for a number of processes to realistically reflect the CDR potential of EW—such as mineral weathering kinetics, saturation feedbacks in soil porewaters, formation of secondary minerals, cation exchange reactions, hydrologic export of alkalinity, and downstream losses.[19,68,172,325] Upstream emissions associated with mining, grinding, transport, and spreading can also substantially reduce net removal and must therefore be considered in technical potentials. Economic potential further incorporates constraints related to energy costs, supply chains, infrastructure, and monitoring and verification requirements, while sustainable potential introduces environmental and governance limits such as soil and water quality thresholds, ecosystem impacts, and social acceptance. The relationships between these categories and the processes influencing them are summarised in Figure 20.

In practice, however, no existing EW assessments can be fully classified into any of these potential categories based on the criteria defined above and in Figure 20. This limitation primarily reflects the evolving understanding of EW system dynamics rather than shortcomings of earlier studies at the time of conception. Several processes now recognised as key controls on realized CDR—such as cation exchange reactions, secondary phase formation, and other soil-chemical feedbacks—have only recently been highlighted through field observations and modelling studies.[19,68,172,216,325,343,445] At the same time, empirical datasets required to constrain many of these processes remain limited, meaning that incorporating them into models can introduce substantial additional uncertainty—one of the main motivations behind the database effort behind this systematic review. Modelling EW across heterogeneous soils, climates, and hydrologic systems is therefore inherently complex, and the progression of modelling approaches reflects successive efforts to incorporate these controls as understanding and data availability improve.

Consequently, reported EW scale estimates must be interpreted in light of the processes represented in each analysis, the assumptions and system boundaries of each study, and the empirical data they are based on. The literature spans a wide range of modelling approaches that capture different subsets of processes. Some assessments rely on simplified or empirically constrained formulations that extrapolate from limited geochemical observations, enabling extensive sensitivity analysis while requiring relatively few input parameters. Such studies that

	technical potential	economic potential	sustainable potential
definition	Maximum realizable CDR achievable with current scientific understanding and physical constraints, assuming deployment is unconstrained by cost, policy, environmental, or social constraints but *including geochemical feedbacks, dissolution limits, losses, and resource availability*.	Portion of technical potential achievable when deployment is limited by cost, supply chains, market dynamics, infrastructure, and competing land or material uses.	Portion of technical potential that remains once environmental safeguards, social acceptance, governance limits, and long-term sustainability constraints are applied.
needs to include	• Feedstock application amount limits based on rock availability (geological + logistical) • Weathering kinetics + saturation feedbacks • Secondary mineral formation • Climate + soil controls • Hydrologic export and downstream losses • Particle size–reactivity trade-offs • LCA emissions (mining, grinding, transport)	Everything in technical plus: • Cost of mining, grinding, transport • Energy prices + carbon intensity • Labor + infrastructure • Opportunity costs (land, materials) • Market and policy demand for credits • Monitoring and verification costs • Discount rates + financing	Everything in technical plus: • Soil, water, and air quality thresholds • Trace-metal accumulation limits in soils, biomass, and downstream waters • Ecosystem and biodiversity impacts • Long-term agronomic performance and soil health • Regulatory standards and liability frameworks • Social acceptance • Intergenerational risk and reversibility • If defined as such also economic potential criteria
can be excluded	• Application amount limits based on agronomic, environmental, or public acceptance constraints • Costs • Policy constraints • Public perception • Market behavior • Adoption barriers • Other factors defined for economic & technical potential	• Application amount limits based on agronomic, environmental, or public acceptance constraints • Political feasibility	If defined as subset of economic potential all physical, economic, environmental, and social limits should be included.
risk	Overestimates deployable CDR if feedbacks or losses are omitted. Not implementable under realistic constraints.	Overestimates real deployment potential if environmental or social limits ignored.	If defined as subset of economic potential some environmentally acceptable projects that are not economically viable are excluded.
afforestation analogy	Acceptable to assume that trees are planted across all climatically and biophysically suitable managed and marginal land ignoring other land use needs, *provided growth rates & carbon storage functions, disturbance regimes, and ecosystem limits remain realistic—i.e., forests are only simulated where environmental conditions permit establishment* (i.e., not deserts, ice sheets, or unsuitable soils).	Trees are planted only where afforestation is both biophysically and financially viable, meaning establishment, maintenance, and land opportunity costs are covered under assumed market conditions, even if additional environmentally suitable land exists that would be biophysically feasible but not profitable.	Trees are planted only where doing so does not compromise biodiversity, water resources, food production, land rights, public acceptance, or ecosystem functioning in addition to being viable for tree growth.

Figure 20: Definitions and criteria for different Carbon Dioxide Removal (CDR) potential types in the context of Enhanced Weathering (EW).

e.g. prescribe mineral dissolution rates without accounting for feedbacks such as porewater saturation, secondary mineral formation, hydrologic export, or other loss pathways can still provide useful bounding or sensitivity insights but should be interpreted as theoretical or exploratory estimates rather than deployable technical potentials. Others employ more mechanistically explicit frameworks, such as RTMs, that simulate mineral dissolution kinetics, soil chemistry, hydrologic transport, and downstream carbon storage pathways (see Section 3.5).[19,20,22,24,68,97,108,110,125,232,252,371,404,405] These models provide critical insights into system behaviour and are becoming increasingly important, but they also require numerous input parameters that are often poorly constrained empirically and can drive order-of-magnitude differences in predicted outcomes.[97,172,252,371,437,516] They are also computationally intensive, which can limit systematic exploration of parameter uncertainty. Simpler models therefore remain valuable because they enable broader testing of assumptions and parameter space, even though they may omit processes influencing net CDR.

These differences are particularly relevant for scenario-based assessments, including those embedded in IAMs discussed in Section 11. Such frameworks explore EW deployment within mitigation portfolios optimized under economic and policy constraints but currently differ substantially in how explicitly they represent weathering processes and associated geochemical limitations. Their implied CDR magnitudes therefore depend strongly on underlying assumptions about weathering kinetics, environmental feedbacks, and loss processes. The following section reviews some of the most widely cited EW CDR potential estimates while interpreting them in light of the processes and assumptions represented in each study.

5.2.2 Global rates and potentials

In this section we first examine three global EW CDR potential estimates that have been particularly influential in shaping the literature over the past decade. These studies differ substantially in their modelling approaches, assumptions, system boundaries, and the extent to which key soil and transport processes are explicitly represented. Understanding these differences is essential for interpreting reported removal magnitudes and for placing individual estimates within the broader range of projected EW potentials. The underlying methodological choices and process representations of these studies—interpreted in relation to the potential definitions and criteria outlined above—are summarised in Table 5. Together, these analyses represent successive methodological steps toward quantifying the large-scale CDR potential of EW. We then discuss additional global estimates, including both simplified modelling approaches and empirically constrained assessments, some of which have emerged more recently.

One of the earliest global-scale assessments provided a landmark quantitative framework that helped structure subsequent research by integrating weathering kinetics, particle size, deployment extent, and cost considerations within a single analysis.[23] This work demonstrated the strong sensitivity of projected removal to grain size and reaction rates and reported global cropland-scale capacities of up to ~95 $GtCO_2$ yr^{-1} for dunite and ~4.9 $GtCO_2$ yr^{-1} for basalt under favourable assumptions.[23] As summarised in Table 5, these estimates are based on prescribed dissolution kinetics and do not include negative feedbacks on primary weathering rates or subsequent loss processes (cf., Section 3.4). Because these processes influence the translation of mineral

Table 5: Comparison of selected key Enhanced Weathering (EW) potential studies across model design, represented loss pathways, and accounted carbon reservoirs, highlighting differences between an approach that neglects negative feedbacks on weathering[23] vs. those that employ soil reactive-transport modelling frameworks.[19,24] Rows summarise how each study treats process representation (e.g., soil domain, feedbacks, climate forcing), inclusion of carbon dioxide removing (CDR)-reducing mechanisms, and which carbon pools are explicitly tracked, highlighting major differences in system boundaries and process completeness that affect comparability of reported CDR estimates.

	Strefler et al. (2018)[23]	**Beerling et al. (2020)[19]**	**Baek et al. (2023)[24]**
I. model details			
model type	Parameterized global/regional EW potential; weathering depends on grain size and assumed weathering rates (Temperature (T) and pH dependence explored)	PHREEQC based 1D-RTM, weathering based on laboratory dissolution rates, soil chemistry evolution taken into account, equilibrium of secondary phase dissolution/precipitation	1-D- RTM (SCEPTER) coupled with climate model experiments
soil model domain	soils not simulated	top 15 cm	top 50 cm
weathering feedbacks	no	increasing saturation, no surface passivation	increasing saturation, no surface passivation
grid size	2 climate zones	1/6 °	1° grid
climate	classified into warm and temperate	gridded GCM (CESM) output 2040-2050, constant water flow	runs using RCP scenarios
II. CDR losses taken into account			
mining	no	yes	no
grinding	no	yes	no
transport	no	yes	no
paedogenic carbonates	no	yes	yes
secondary clays	no	~no (gibbsite)	yes (kaolinite, Ca-/Mg-beidellite)
oxide formation	no	yes (goethite)	yes (goethite)
plant biomass uptake	no	no (assumed to be reversed on the relevant timescales, instead biological increase of weathering rates simulated)	no
CEC processes	no	yes (but no exchangeable acidity)	no
strong acid input	no	no	no
other secondary phases	no	amorphous silica, gibbsite	no
rivers	no	no	no
oceans	no	yes (14%)	yes (14%)
III. carbon reservoirs considered			
atmosphere	implicit (equilibrium assumed)	implicit (equilibrium assumed)	yes (quantifies change in diffusive flux from soils)
alkalinity/DIC export	implicit yes	yes (estimated based on cation export)	yes (quantifies DIC flux from soils)
paedogenic carbonate	implicit no	yes	yes

* based on thermodynamics-based estimates of ocean degassing [18,19,24,37,126]

Table 5 (continued): Comparison of feedstock assumptions, deployment constraints, and resulting CDR estimates across global EW assessments. Rows synthesize differences in rock type, particle size, application strategy, land availability, and reported annual or cumulative removal, illustrating how methodological choices and deployment assumptions propagate into divergent global potential estimates.

	Strefler et al. (2018)	**Beerling et al. (2020)**	**Baek et al. (2023)**
IV. feedstock & application			
feedstock type	dunite and basalt	basalt (multiple mineralogies)	basalt
source areas	not analysed/assumed abundant	non-protected GLiM source areas, no production limit	not analysed/assumed abundant
particle size	20 µm, kept constant	optimal p80 for maximum CDR (based on grinding and weathering trade-off), tracks particle size evolution	"fine" basalt p80 = 100 µm "coarse" p80 = 1,220 µm
application rate (t ha-1)	replacement of annually weathered rock @ 150 t ha^{-1}	40	10
application limit	150 t ha^{-1} applied everywhere	no	no
application area	~50% of global cropland area (sufficiently warm and humid)	10-57% of national cropland areas depending on global target CDR rate	Agricultural hotspots (>70% coverage), global croplands
mixing depth	/	15 cm	25 cm
V. CDR			
annual rate	4.9 $GtCO_2$ yr^{-1} (basalt) 95 $GtCO_2$ yr^{-1} (dunite)	0.5-2 $GtCO_2$ yr^{-1} (by 2050?)	1.15 tCO_2 yr^{-1} **ha^{-1}** over 75 years based on fine basalt (1.14 for RCP4.5, 1.16 for RCP8.5)
cumulative (Gt)	Not assessed	25-100 over 5 decades	64 for agricultural hotspots (>70% ag area) 217 $GtCO_2$ all agricultural land Based on fine basalt and 2006-2080 period
note	No loss processes or rate (i.e., saturation) dependent negative impacts on weathering rates considered no environmental limits for dunite application.	CDR rates prescriptive, drive required area to achieve CDR rate most efficiently (lowest land footprint)	Passivation effects not included; coarse feedstock can leave large fraction undissolved over decades

dissolution into geochemically achievable net CDR, the resulting values should not be interpreted as realizable technical potential, even though the study itself remains a seminal and influential benchmark. This interpretation is further supported when expressed as implied area-normalized fluxes: global average CDR fluxes derived from Strefler et al. (2018)[23] correspond to ~120 tCO_2 ha^{-1} yr^{-1} for olivine and ~6.2 tCO_2 ha^{-1} yr^{-1} for basalt across croplands. These values are higher than most of the data compiled in Section 5.1, and it seems unlikely that such fluxes could be achieved globally on average, reinforcing that such estimates represent upper-bound potentials under idealized assumptions rather than field-realistic technical potentials. Earlier conceptual modelling work similarly suggested that EW of olivine could sequester on the order of ~3.7 $GtCO_2$ yr^{-1} under favourable conditions, although these estimates relied on simplified representations of weathering environments and deployment logistics.[17]

More recent global assessments employ RTM frameworks that simulate the coupled evolution of mineral dissolution, porewater chemistry, and solute transport in soils, allowing chemical

feedbacks and loss pathways to influence projected removal.[19,24] These approaches explicitly represent processes that can regulate weathering efficiency, including feedbacks associated with porewater saturation and the partitioning of weathering products among carbon reservoirs. At the same time, projected removal remains sensitive to model structure, parameterization, and deployment assumptions, and therefore depends strongly on which processes are represented and how they are constrained empirically. As summarised in Table 5, the studies differ substantially in the processes represented, parameterization strategies, and underlying assumptions regarding feedstock deployment. One modelling framework chooses CDR rates emergent from the model solution space and then optimizes the spatial extent of rock application to minimize land footprint and cost. Under these assumptions, national-scale simulations suggest that applying basalt to roughly 10–57% of cropland areas—depending on country and global CDR targets—could contribute on the order of 0.5–2 $GtCO_2$ yr^{-1}, corresponding to cumulative removals of approximately 25–100 $GtCO_2$ over multi-decadal timescales.[19] Similarly, simulations using a fixed application rate of 10 t ha^{-1} yr^{-1} yield cumulative totals of ~64 $GtCO_2$ from agricultural hotspots (>70% ag-extent) and ~215 $GtCO_2$ when extrapolated to all croplands over the simulation period of 2006-2080.[24]

These frameworks[19,24] represent important advances in the explicit representation of soil geochemical processes and are among the most substantiated modelling approaches currently available for evaluating EW performance at scale. Nevertheless, the resulting estimates remain contingent on assumptions regarding system boundaries and deployment conditions—including life cycle emissions, sustained amendment inputs, feedstock availability, and mining constraints (Table 5)—as well as on the specific processes represented in the models. Taken together, these considerations indicate that projected removals should be interpreted in the context of model assumptions and may not represent definitive bounds on achievable technical potential. Rather, these studies reflect successive methodological advances toward increasingly comprehensive representations of soil weathering systems and provide a critical quantitative foundation for assessing the large-scale feasibility of EW.

Other global assessments use simplified, parameterized weathering formulations rather than fully coupled soil reactive-transport frameworks, allowing rapid sensitivity analysis but providing a less complete representation of system behaviour. For example, a global cropland model that expresses dissolution as a function of soil chemistry, temperature, and mineral type (wollastonite, forsterite, and lizardite; assuming 1 t ha^{-1} yr^{-1} application) produces highly variable outcomes across environmental settings, with cumulative sequestration spanning a wide range but reaching on the order of several tens of Gt by 2100 under favourable assumptions.[412] Similarly, another recent global assessment estimates EW removal by scaling a baseline sequestration flux (~1.5 tCO_2 ha^{-1} yr^{-1} under moderate silicate application; higher than median fluxes compiled here—Section 5.1) according to local temperature and soil water surplus, reflecting climatic controls on weathering kinetics.[245] Applying this approach at ~5 km resolution produces projected removals of ~0.35–0.76 $GtCO_2$ yr^{-1} by 2050 and ~0.7–1.1 $GtCO_2$ yr^{-1} by 2100, with strong geographic variability driven by climatic suitability and projected adoption patterns.

At the lower end of global estimates, one early empirically-constrained assessment compiled observed bicarbonate fluxes from mineral applications and natural weathering environments to derive a first-order removal potential of ~0.24 $GtCO_2$ yr^{-1}, grounding its calculation in measured

solute export rather than prescribed reaction kinetics and thereby inherently integrating many processes that can diminish net CDR.[14] Similarly, a later global analysis employed a simplified parameterization linking weathering rates to climatic and geochemical variables but calibrated these relationships against experimentally observed dissolution behaviour in EW experiments, yielding upper-bound estimates of ~0.22 ± 0.16 $GtCO_2$ yr^{-1} and ~17 ± 13 Gt cumulative removal by 2100.[517] Reflecting sparse experimental data particularly for the earlier study, both approaches rely on observational datasets that remain minimally available at the global scale, which limits both their ability to resolve environmental heterogeneity and the accuracy of the empirical grounding. As empirical datasets increasingly expand through published field trial results, approaches that scale removal from measured fluxes rather than assumed kinetics will become increasingly feasible for constraining global potentials over the coming years.

Sustainable potentials for EW have not yet been robustly quantified as none of the global-scale models discussed above contain feedstock application thresholds based on environmental considerations and assume high and/or continued rock application (cf., Table 5). Such deployment scenarios are unlikely to be achievable when considering both environmental[129,131] and agronomic limitations such as rock build up in soils or pH overshoot.[19,518,519] For example, simulations of basalt application at ~40 t ha^{-1} yr^{-1} indicate that concentrations of elements such as Cu and Ni could exceed national soil quality limits within years to decades depending on rock composition and jurisdiction, with limits for dunite being exceeded much quicker.[129] Even lower application rates may eventually lead to regulatory thresholds being reached, implying that deployment ceilings could be set not by geochemical potential but by environmental constraints.[129] It should be noted that these thresholds were originally developed for comparatively fast-reacting soil amendments rather than slowly dissolving silicates, and therefore may ultimately require reassessment.[129,131] However, even under revised standards, continued applications would still progressively accumulate trace metals in soils that may remain largely immobile for centuries due to slow weathering kinetics but could ultimately become bioavailable or soluble if soil conditions change. Collectively, these considerations indicate that existing large-scale potentials should not be interpreted as sustainable potentials, but rather as theoretical upper bounds pending integrated agronomic, geochemical, and regulatory evaluation.[129,131]

Taken together, current literature indicates that EW could contribute meaningfully to CDR at large scales, but that projected magnitudes remain highly contingent on model structure, deployment assumptions, and the extent to which key soil and transport processes are represented. Across the studies discussed here, empirically constrained flux-based estimates suggest lower-bound global technical potentials on the order of ~0.22–0.24 $GtCO_2$ yr^{-1},[14,517] while climate-based modulation of empirically constrained area normalized CDR fluxes project ~0.35–0.76 $GtCO_2$ yr^{-1} by mid-century and ~0.7–1.1 $GtCO_2$ yr^{-1} by 2100.[245] More process-explicit reactive-transport simulations indicate that sustained large-scale deployment could plausibly reach ~0.5–2 $GtCO_2$ yr^{-1} under favourable assumptions,[19] with cumulative removals of tens to hundreds of Gt over multi-decadal timescales, including global projections of ~64–215 $GtCO_2$.[24] Despite substantial differences in modelling approaches, parameterization strategies, and the extent to which loss pathways and environmental constraints are represented, most current assessments converge on a broadly similar order-of-magnitude range of roughly ~0.2–2 $GtCO_2$ yr^{-1} for large-scale CDR potential.[520] While uncertainties in currently poorly constrained input parameters could in principle shift future estimates upward, the more likely outcome is that these ranges represent the upper bound of

achievable technical potential, as existing assessments do not yet incorporate all CDR losses and constraints required to fully characterize deployable technical potentials (cf., Figure 20 and Table 5).[1]

Hence, interpreting these values requires careful distinction between what is chemically possible, what is physically achievable, and what is environmentally permissible, as they reflect different system boundaries and evidentiary bases. As modelling frameworks, observational datasets, and process understanding continue to improve, convergence among these estimate classes is likely, enabling more robust quantification of realizable and sustainable potentials. Until such harmonisation is achieved, global CDR projections for EW are best understood as scenario-dependent ranges rather than single best estimates, with their primary value lying in identifying governing controls, research priorities, and deployment constraints. A model intercomparison project could provide a much better view on key differences between models and likely a more accurate look at global EW potential.

5.2.3 Regional rates and potentials

This section breaks down the global-scale estimates discussed above into their regional components. Numerous additional regional assessments of EW potential exist, including those derived from LCA frameworks and IAMs. These studies are discussed elsewhere in this review (cf., Sections 8 and 11). However, many derive their geochemical removal estimates from earlier process-based studies or apply simplified weathering parameterizations.[19,23] To avoid duplication and maintain conceptual consistency, we therefore focus here on how the studies discussed above resolve spatial variability within their modelling frameworks. We also exclude analyses that extrapolate national or continental potentials from single experimental sites.

Early global modelling already embedded strong climatic controls into projections of EW potential by parameterizing silicate dissolution as a function of grain size and laboratory-derived mineral weathering kinetics, with reaction rates scaled by temperature and soil chemistry [23]. This parameterization leads to substantially higher modelled removal efficiencies in warm climates and consequently concentrates much of the global potential in tropical and monsoon-influenced agricultural regions. Under these assumptions, India, Brazil, Southeast Asia, and China together account for roughly 75% of the modelled global CDR potential.[23]

Similarly, as a direct consequence of model parametrization, subsequent spatially explicit RTM-based assessments highlight strong climatic controls on the geographic distribution of EW potential.[19,24] A global techno-economic cropland deployment analysis evaluating removal targets of 0.5–2 $GtCO_2$ yr^{-1} finds that the largest national contributions arise in countries combining extensive cropland area with favourable weathering conditions, particularly China, India, the United States, and Brazil.[19] Under scenarios reaching these global targets, national contributions range from roughly 0.13–0.53 $GtCO_2$ yr^{-1} for China, 0.11–0.42 $GtCO_2$ yr^{-1} for the United States, and 0.15–0.49 $GtCO_2$ yr^{-1} for India, with Brazil contributing approximately 0.04–0.17 $GtCO_2$ yr^{-1} depending on the assumed deployment fraction.[19] Consistent with the climatic controls on weathering kinetics, regions with warm and seasonally wet climates—including Brazil and Indonesia[19]—show comparatively high removal potential per unit cropland area, with tropical and monsoon regions showing the highest uptake rates.[24] Modelled CDR fluxes reach approximately

1.2–1.6 tCO_2 ha^{-1} yr^{-1} in cropland regions of Africa, Thailand, eastern China, and India, compared with roughly ~1 tCO_2 ha^{-1} yr^{-1} across major agricultural regions of North America, South America, and Europe/Russia, while arid regions such as Australia show substantially lower rates below ~0.6 tCO_2 ha^{-1} yr^{-1}.[24] These spatial differences translate into strongly uneven regional contributions, with Asia dominating modelled cumulative removal followed by Africa, Europe, South America, and North America, reflecting particularly strong removal across tropical and monsoon-influenced croplands (Figure 21).[245]

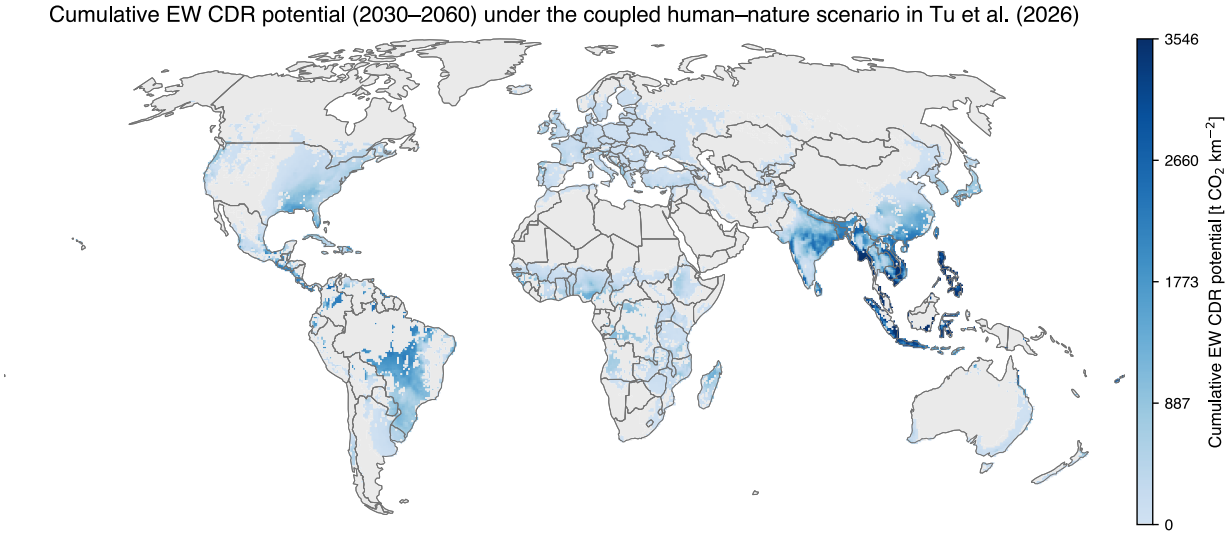

Figure 21: Cumulative Carbon Dioxide Removal (CDR) potential from Enhanced Weathering (EW) for 2030–2060 under a human–nature coupled scenario, expressed as tCO_2 km^{-2}. Values represent land-area-normalized CDR, integrating both cropland-area-specific EW efficiency and the fraction of cropland within each grid cell. As such, the mapped patterns reflect the combined influence of biophysical weathering potential and cropland availability, rather than per-cropland-area removal rates alone. The human–nature coupled scenario accounts for feedbacks between climate change and societal response, in which EW adoption accelerates as warming crosses key thresholds, reflecting increased policy action, risk perception, and deployment effort under escalating climate stress. Figure based on data in Tu et al. (2026).[245]

Scenario analyses (reflecting different policy choices) that incorporate technology diffusion dynamics further suggest that regional deployment patterns depend not only on geochemical potential but also on socioeconomic adoption trajectories.[245] In this global framework, spatially explicit CDR fluxes are first estimated by modulating nominal sequestration fluxes (1.5 tCO_2 ha^{-1} yr^{-1}) with temperature and soil water surplus, resulting in systematically higher potential weathering rates in warm and humid regions. When these biophysical estimates are combined with technology diffusion-based adoption trajectories derived from historical agricultural technologies, early deployment is concentrated in high-income regions such as North America and Europe, which lead adoption during the 2030–2040 period. However, as diffusion progresses, deployment expands rapidly in regions with large cropland areas and favourable climatic conditions. By mid-century, countries in South and East Asia and parts of Latin America dominate modelled removal, with South Asia projected to achieve the largest cumulative sequestration (~14 $GtCO_2$ by 2100), followed by East Asia & Pacific (~13 $GtCO_2$) and Latin America (~9 $GtCO_2$). In contrast, Europe

and North America contribute smaller cumulative totals (~3 $GtCO_2$ each) despite earlier adoption because of comparatively lower per-area weathering rates in temperate climates. These shifts are also reflected across income groups: high-income countries account for roughly 39–57% of global EW removal by 2040 but decline to <12% by 2100, while low- and lower-middle-income countries increase their share from ~20–29% in 2040 to nearly 60% by the end of the century as adoption accelerates in climatically favourable regions.

Taken together, these studies consistently indicate that regional variability in EW potential arises from the interaction of climatic controls on mineral dissolution, the spatial distribution of cropland available for deployment, and socioeconomic factors governing adoption. Across modelling frameworks, the highest geochemical efficiencies occur in warm and humid environments across tropical and monsoon-influenced regions, while the largest absolute contributions arise in countries combining favourable climates with extensive agricultural land area, including China, India, the United States, and Brazil. Importantly, these spatial patterns are not only empirical outcomes but also reflect how weathering is parameterized within many global models. Most approaches explicitly scale dissolution rates with climatic variables—commonly using Arrhenius-type temperature dependencies and hydrological constraints on reaction progress—such that higher modelled removal rates in warm and seasonally wet regions are partly embedded in the representation of weathering kinetics itself [19,23,24,245].

5.2.4 Multi-parameter optimization: identifying system-specific “sweet spots”

As demonstrated above, many current global assessments implicitly or explicitly optimize EW deployment through climatic suitability. In most modelling frameworks, higher temperatures and excess moisture accelerate silicate dissolution kinetics, leading to the conclusion that warm and humid regions provide the most favourable conditions for maximizing weathering rates.[19,24,245,521] While this framing is useful for identifying broad deployment envelopes, it represents only one dimension of a much larger optimization problem.

Enhanced Weathering outcomes are influenced by a wide range of interacting environmental, geochemical, and agronomic controls (Section 3.2)—these are illustrated in the inner ring of Figure 22. They include soil properties such as pCO_2,[15,63,97] CEC,[68] BS,[68,216] SOM,[42,43] as well as reactive soil mineral pools and soil acidity/pH [62,68,216]; climatic drivers such as temperature and water inputs;[62] and feedstock characteristics including mineral reactivity and dissolution rates,[62,119,120] contaminant concentrations.[129,130]

At the same time, absolute CDR (and initial CDR before losses, for which weathering rate is often used as a proxy) represents only one of several potential optimization outcomes. Others include minimizing environmental risks such as trace metal mobilisation[131,150] or non-CO_2 GHG emissions,[22,83,522] maximizing agronomic co-benefits including soil fertility and crop productivity,[22,47,523] minimizing losses of weathering products to secondary mineral formation or strong-acid weathering pathways,[172,216,343,445,460] and enhancing SOC storage or broader soil health outcomes.[94] These outcome variables illustrated in the outer ring of Figure 22 represent different objectives that deployment strategies may prioritize depending on environmental context, policy frameworks, and stakeholder priorities.

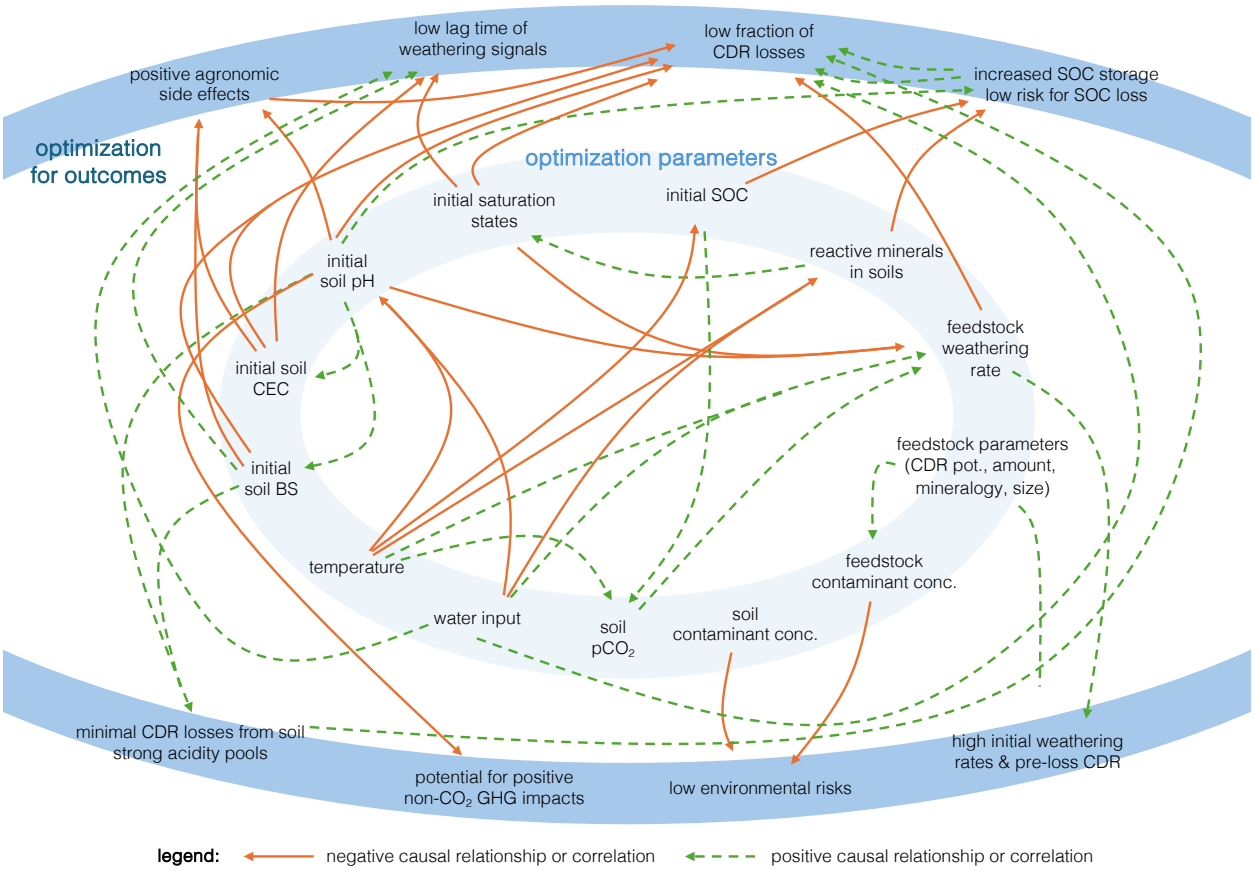

Figure 22: Conceptual overview of interacting parameters and outcomes in Enhanced Weathering (EW) optimization. The inner ring illustrates key parameters that can influence EW performance, including soil properties (e.g., pH, cation exchange capacity (CEC), base saturation (BS), soil organic carbon (SOC)), climatic drivers (temperature and water input), feedstock characteristics (reactivity, Carbon Dioxide Removal (CDR) potential, contaminant concentrations), and system properties such as existing reactive minerals or acidity pools. The outer ring highlights example outcomes that deployment strategies may seek to optimize, expressed as desirable system states (e.g., low environmental risk, high agronomic co-benefits, minimal CDR losses, or increased SOC storage). Arrows indicate generalized positive or negative correlations or causal relationships between parameters and outcomes. These relationships are illustrative rather than deterministic: many represent typical tendencies observed across studies, but exceptions exist and individual systems may deviate from the indicated directionality (for example, some high-CDR feedstocks may exhibit low contaminant concentrations even though generally there is a positive correlation between both parameters[130]). The figure therefore emphasizes the strong interdependence among environmental conditions, feedstock properties, and system responses, illustrating that optimizing EW often requires navigating trade-offs across multiple interacting factors.

Several examples illustrate the complexity of this multi-parameter space. For example, soil chemical properties such as CEC and BS influence whether newly released cations are retained on exchange sites and for how long.[68] Such factors can substantially delay alkalinity formation, extending lag times between mineral dissolution and realized CDR from years to decades.[68,343,445] Soil pH represents another critical control, influencing mineral dissolution rates,[119,120] nutrient availability,[215] and trace metal mobility,[524] while also regulating the balance between carbonic-

acid and strong-acid weathering pathways.[214,231] The saturation state of porewaters with respect to secondary mineral phases can determine how rapidly weathering products are incorporated into newly formed minerals,[172] thereby reducing the fraction of dissolved material contributing to net CO_2 sequestration both via providing a sink for cations (incorporation into clays and carbonates) as well as limiting weathering rates of primary minerals by forming protective coatings.[172]

Changes in many of these parameters often also influence other parameters directly or through shared environmental controls. For instance, feedstocks with higher intrinsic weathering reactivity or theoretical CDR potential may also contain elevated concentrations of trace metals,[129,130] meaning that selecting materials to maximize weathering efficiency can simultaneously increase environmental risk and regulatory constraints. Similarly, soil chemical properties such as pH, CEC, and BS are tightly coupled through ion exchange and buffering processes in soil porewaters, linking mineral dissolution, nutrient availability, and alkalinity retention.[215] Climatic factors such as temperature and water availability further exert first-order controls on a range of soil properties, including soil pH, mineral weathering rates, and the availability of reactive mineral pools.[15,215] As a result, optimizing one parameter—for example through site selection or feedstock choice—may simultaneously alter multiple other system properties, improving some outcomes while constraining others.

Hence these parameters and outcomes are not independent. Many co-vary across landscapes or exert causal influences on one another, and they simultaneously affect multiple outcome variables. Most global assessments do not explicitly represent many of these interactions or focus on a single objective—maximizing CDR—without accounting for potential trade-offs with other outcomes. The resulting network of correlations and causal relationships is illustrated schematically in Figure 22. The complexity of this network is itself the central point: EW performance emerges from a densely interconnected system of interacting parameters and outcomes, few of which can be adjusted in isolation. Because many of these relationships involve trade-offs, deployment strategies that optimize one outcome may simultaneously compromise another.

Returning to the climate-based optimization implicit in many modelling approaches (Sections 5.2.2 & 5.2.3), the environmental conditions that maximize modelled weathering rates primarily reflect optimization of reaction kinetics. However, these conditions do not necessarily maximize system capacity to retain and export weathering products, or even realizable CDR once losses are considered. Tropical and monsoon-influenced regions often exhibit high temperatures and abundant moisture, conditions that accelerate mineral dissolution and therefore favour rapid initial weathering. However, many soils in these regions are also highly weathered, strongly acidic, and nutrient depleted, typically characterized by low BS and limited exchangeable cation pools. While these properties can create opportunities for substantial agronomic co-benefits through pH buffering and nutrient supply, they can also increase early losses of potential CDR. In strongly acidic systems, a substantial fraction of the alkalinity generated by mineral dissolution may initially be consumed by neutralizing pre-existing acidity in soils and exchange complexes before net carbon sequestration occurs, effectively creating a transient sink for weathering products.[68,216] Both modelling studies[214] and field observations[216] suggest that overcoming soil acidity can represent a significant initial demand for alkalinity in such environments that could obviate CDR from EW in some tropical settings[216]—a process not considered in all global and regional model estimates.

Additional trade-offs in the optimization-via-climate framing may arise through interactions with SOM, as mineral amendments can influence microbial activity and carbon cycling processes affecting both soil carbon storage and non-CO_2 GHGs.[42,43] Evidence from the agricultural liming literature indicates that SOC responses to pH changes depend strongly on initial soil conditions and are often non-linear.[525] In moderately acidic, SOC-poor soils, liming can increase SOC through enhanced plant productivity and stabilisation, whereas in strongly acidic or SOC- and nitrogen-rich systems responses may be neutral or negative due to stimulated microbial decomposition.[42] In some cases, these effects can be substantial: increases in soil pH may enhance decomposition of existing organic carbon, as demonstrated for tropical peat soils where EW can stimulate CO_2 emissions and carbon mobilisation that partially or fully offset the expected CO_2 uptake.[111] More generally, these responses are not unidirectional: increases in pH and nutrient availability can initially enhance decomposition and CO_2 release,[112,114] while longer-term changes in soil structure and mineralogy may promote stabilisation via aggregation and mineral-associated organic matter (MAOM) formation.[92,526,527] As a result, EW can lead to both gains and losses of SOC, as well as redistribution among pools with contrasting turnover times, with outcomes varying across soils, feedstocks, and management contexts, including interactions with organic amendments that may modify SOC accumulation relative to rock amendments alone.[101,113,439,528] Consequently, environmental conditions that favour rapid mineral weathering do not necessarily maximize net climate benefits once SOC dynamics are considered, and in some cases may partially offset inorganic CDR on relevant timescales.[94]

Taken together, these interactions suggest that EW deployment is better understood as a continuum of more- and less-favourable conditions rather than a single optimal setting. Most existing global assessments have focused on estimating aggregate CDR potential, often emphasizing climatic controls on weathering rates, rather than identifying optimal deployment environments under interacting trade-offs. As a result, key factors such as agronomic benefits, environmental risks, GHG feedbacks, and losses and lag times in CDR are often underrepresented, despite their importance for determining realizable outcomes. This complexity increases further when multiple outcomes are considered simultaneously, as improvements in one dimension may constrain others.

Within this framework, deployment environments can be viewed along gradients reflecting trade-offs between reaction kinetics and system capacity, and thus between immediate and long-term CDR. Warm, humid tropical systems may offer high long-term CDR capacity due to rapid kinetics,[19,24,245] but can also exhibit substantial initial alkalinity losses, soil carbon responses, and delayed net CDR as acidity is neutralized.[111,216] In contrast, systems with higher BS or lower acidity may enable more immediate CDR, albeit with slower intrinsic weathering rates. These patterns highlight that optimal deployment is not defined by climate alone, but by system context and time horizon, including management practices that influence acid inputs, alkalinity export, and nutrient cycling. Accordingly, the central challenge is not identifying a single “best” environment but aligning site conditions and management with desired outcomes across timescales.

This perspective implies that multiple deployment pathways may be viable depending on local conditions and management objectives. For example, regions with relatively high soil pH and BS may experience smaller initial alkalinity losses to exchange reactions,[68] allowing efficient CDR even in cooler climates provided that weathering products can be exported from soils. In such

contexts, rapidly dissolving carbonate feedstocks may sustain high alkalinity fluxes,[30,31,516] overcoming climatic constraints on weathering rates due to multiple orders of magnitude faster weathering kinetics.[119] In some cases, sequential strategies may also be advantageous—for instance, using reactive (e.g., via grain size or mineralogy) silicate amendments initially to neutralize existing soil acidity and build exchangeable base cation pools, followed by carbonate amendments that maintain alkalinity export once the soil acidity has been buffered.

Conversely, in strongly weathered tropical soils where initial alkalinity losses may be larger but agronomic benefits substantial, in addition to optimizing for CDR,[19,23,24,245] deployment strategies may prioritize soil fertility improvement and pH buffering, with CDR emerging as a co-benefit of agronomic management; particularly over long time horizons once initial soil acidity is buffered. Because many tropical regions have contributed relatively little to historical GHG emissions yet face disproportionate climate risks,[1,5] prioritizing agronomic co-benefits in these settings may also align with principles of climate justice (see Section 9) and strengthen progress towards multiple SDGs[84] in addition to insuring benefit-driven stakeholder buy in. In this context, EW may function not only as a CDR strategy but also as a mechanism for coupling climate mitigation with equitable and development-oriented land management.

Recognising this diversity of possible pathways shifts the central question from identifying a single "best" location for EW to determining how deployment strategies should be matched to system boundary conditions and desired outcomes. Achieving this will require expanded empirical datasets and modelling frameworks capable of representing interacting soil processes, management interventions, and multiple optimization objectives simultaneously. As observational constraints and conceptual frameworks improve, future assessments may move beyond climate-centric deployment narratives toward more flexible multi-parameter strategies that capture the full complexity and opportunity of EW in managed landscapes.

5.3 Voluntary Carbon Market activity

A substantial portion of recent EW deployment has occurred through VCM purchasing, creating a growing record of project activity that can complement the academic literature. Publicly available order datasets (e.g., CDR.fyi) provide useful basis for tracking market participation and the evolution of EW crediting over time (Figure 23).

5.3.1 VCM market signals: cumulative activity and delivery status

Analysis of publicly available CDR.fyi order data indicates a rapid increase in EW crediting activity over the past several years (Figure 23a). Cumulative EW credits rise strongly through time, with particularly pronounced growth since late 2024, reflecting increased participation and purchasing commitments within the VCM, totalling ~0.8 Mt of CDR credits sold in early 2026. Despite this growth, EW has remained a relatively small component of overall CDR activity to date, contributing on the order of ~1–2% of cumulative credits sold since 2022 (Figure 23b, status by announcement not delivery date). This contribution is concentrated among a limited number of buyers—primarily Microsoft and Frontier Fund (an advanced market commitment funded by a consortium of corporations in the tech and banking sectors). This indicates that current EW market

activity is sensitive to purchasing decisions by a small subset of actors, a feature that has been noted more broadly for durable CDR markets.[529]

A pronounced distinction between credits sold and credits delivered or retired is also evident. Credit sales primarily indicate procurement commitments and willingness to pay, whereas delivery and retirement reflect realized CDR and therefore provide a more direct indicator of climate relevance. This divergence is illustrated in Figure 23b, which shows the fraction of cumulative EW credits that have been delivered or retired relative to total credits recorded by a given date based on CDR.fyi data. As expected, delivery fractions are higher for earlier vintages, reflecting the additional time available for project execution and verification. Approximately 20–40% of EW credits contracted prior to mid-2023 are recorded as delivered or retired, whereas delivery fractions for more recent cumulative sales are currently below 1%. Such discrepancies between sales and delivery are characteristic of emerging CDR pathways, where project development timelines, weathering kinetics, and evolving MRV frameworks introduce substantial temporal lags between procurement and verified outcomes.

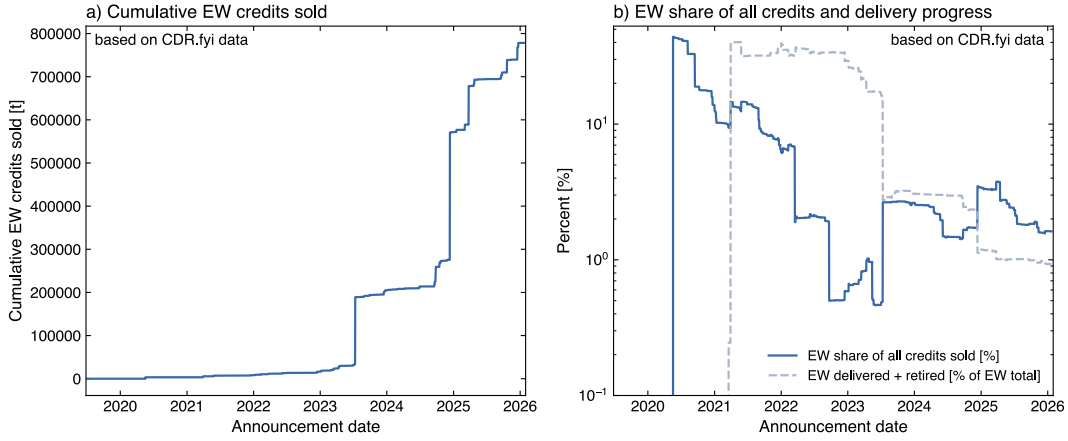

Figure 23: Enhanced Weathering (EW) credit deployment and delivery status in the Voluntary Carbon Market (VCM). (a) Cumulative EW credits recorded in the CDR.fyi orders database, aggregated across all project statuses (contracted, partial, delivered, and retired). Because project records may transition from "contracted" to "partial", to "delivered" or "retired" over time, all statuses are treated as representing credits that were sold with the announcement data reflecting the initial contracting time. (b) Two complementary indicators of EW market penetration and progress, shown on a logarithmic y-axis: the share of EW credits as a percentage of all CDR credits sold across all methods, and the fraction of EW credits that have been delivered or retired relative to total EW credits sold. The x-axis is based on announcement dates; the delivered or retired fraction therefore indicates the share of EW credits announced by a given date that have subsequently reached delivered or retired status, irrespective of delivery or retirement date.

5.3.2 Implications of VCM activity for evidence generation and learning

Activity supported through the VCM has the potential to contribute meaningfully to evidence generation for EW. Available information indicates that some VCM-backed deployments are occurring at scales that are orders of magnitude larger than published academic field trials in terms of treated land area and material throughput (e.g., Figure 23a). Nevertheless, direct quantitative

comparison remains challenging, however, due to, in most cases, incomplete public disclosure, heterogeneous reporting practices, and differences in study objectives and timelines. Academic field trials are typically reported years after deployment, whereas VCM contracting often occurs prior to material application, limiting quantitative comparability between the two bodies of evidence.

In principle, large-scale VCM-driven deployments could provide empirical constraints on aspects of EW that are difficult to assess in small or short-duration studies, including operational logistics, agronomic integration, spatial and temporal variability in performance, and cost drivers. Realizing this learning potential depends critically on whether deployment and monitoring data are retained in formats that enable synthesis and secondary analysis.[530] Robust evaluation of CDR claims and associated cost estimates requires broad access to underlying data, particularly while durable CDR pathways remain at early stages of field implementation. Accordingly, transparency mechanisms that extend beyond narrowly permissioned or project-specific access are important for independent evaluation, comparison across approaches, and informed governance.

Data-sharing platforms currently being used in for some EW offtakes may provide a structured mechanism for research access to EW datasets under defined conditions, including proposal review, time-limited access, and engagement with individual data providers. Such approaches could facilitate academic use while addressing legitimate concerns related to proprietary information and data misuse. At the same time, the need to apply separately for individual datasets and to coordinate with multiple data licensors can complicate cross-project synthesis, while restrictions on redistribution and use for site selection will limit the extent to which pooled data can be used to identify broadly favourable deployment conditions—one of the key motivations for aggregating empirical evidence. In addition, the notion of non-profit or NGO bodies policing broader access to data raises clear governance concerns that will likely need to be addressed in order to engender public trust. Overall, there has been little progress toward meeting repeated calls for cost transparency, even though such information is essential for assessing the feasibility of EW at scale and is rarely obtainable from academic field trials alone.

Some registries [431] have begun making portions of the datasets underlying EW crediting claims publicly accessible, representing an initial step toward empirical accountability. In this respect, emerging EW crediting frameworks—which typically require quantitative, measurement-based evidence for credit issuance—stand in marked contrast to SOC crediting, currently the largest category of agricultural carbon credits. Most SOC crediting approaches rely on measurement–model hybrids rather than direct observation and operate without standardised mechanisms for public data disclosure.[531–535] However, transparency provisions for early EW crediting remain limited relative to broader recommendations from the negative-emissions and carbon-market literature calling for open, interoperable datasets that enable independent validation and cross-project synthesis.[7,505,530,536]

Taken together, these considerations suggest that the VCM may function as an important early deployment and learning environment for EW, but only if accompanied by sufficiently transparent and interoperable data practices.[530] While the VCM alone is unlikely to support EW deployment at the multi-gigatonne scale implied by long-term mitigation pathways,[537] it may nonetheless contribute to methodological development, MRV refinement, and empirical understanding during

a period when public funding for large-scale field research and monitoring remains constrained. In this sense, VCM activity could represent one component of a broader portfolio supporting EW development, with its primary near-term value lying in learning-by-deployment, contingent on transparent, accessible, and sufficiently complete data sharing, while longer-term scaling is likely to depend on complementary policy, procurement, and regulatory frameworks beyond purely voluntary demand (see also Section 13).

6 Costs per tonne

Enhanced Weathering has attracted attention as a potentially scalable CDR option because it leverages mature supply chains in mining, comminution, and agriculture. Costs vary widely across deployment scenarios, depending on feedstock type and reactivity, whether materials are newly mined or available as waste streams, grinding requirements, transport distances, application logistics, MRV intensity, and the credited CO2 removal per unit mass applied. As expected, published cost estimates differ by more than an order of magnitude, reflecting both genuine variation in deployment configurations, feedstock types, CDR potential, and geospatial scopes. All cost estimates reported here are harmonised to 2024 US dollar values using inflation adjustments and, where required, currency conversions.

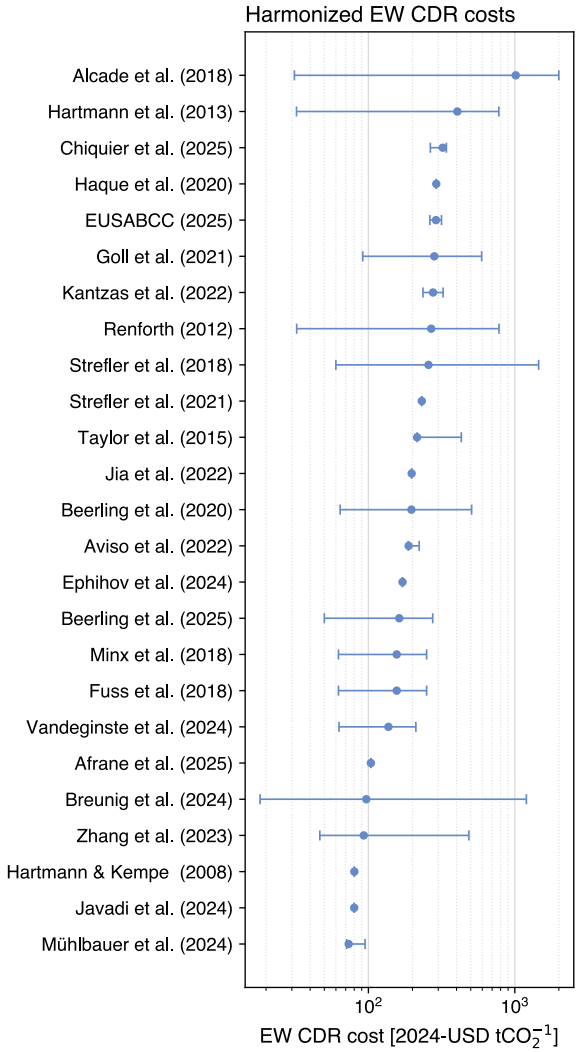

Figure 24: Study-level synthesis of harmonised Enhanced Weathering (EW) Carbon Dioxide Removal (CDR) costs (cost converted to 2024 USD). Study-level EW CDR cost estimates are reported as harmonised USD per tCO_2 removed. Each point denotes a study's central estimate (median across that study's extracted datapoints). Horizontal whiskers indicate the study-level minimum–maximum across extracted datapoints using reported bounds where available. For datapoints that report only a cost range (min–max) but no best estimate, the central value is computed as the midpoint of the reported range; datapoints with only a central value are treated as having zero-width bounds.

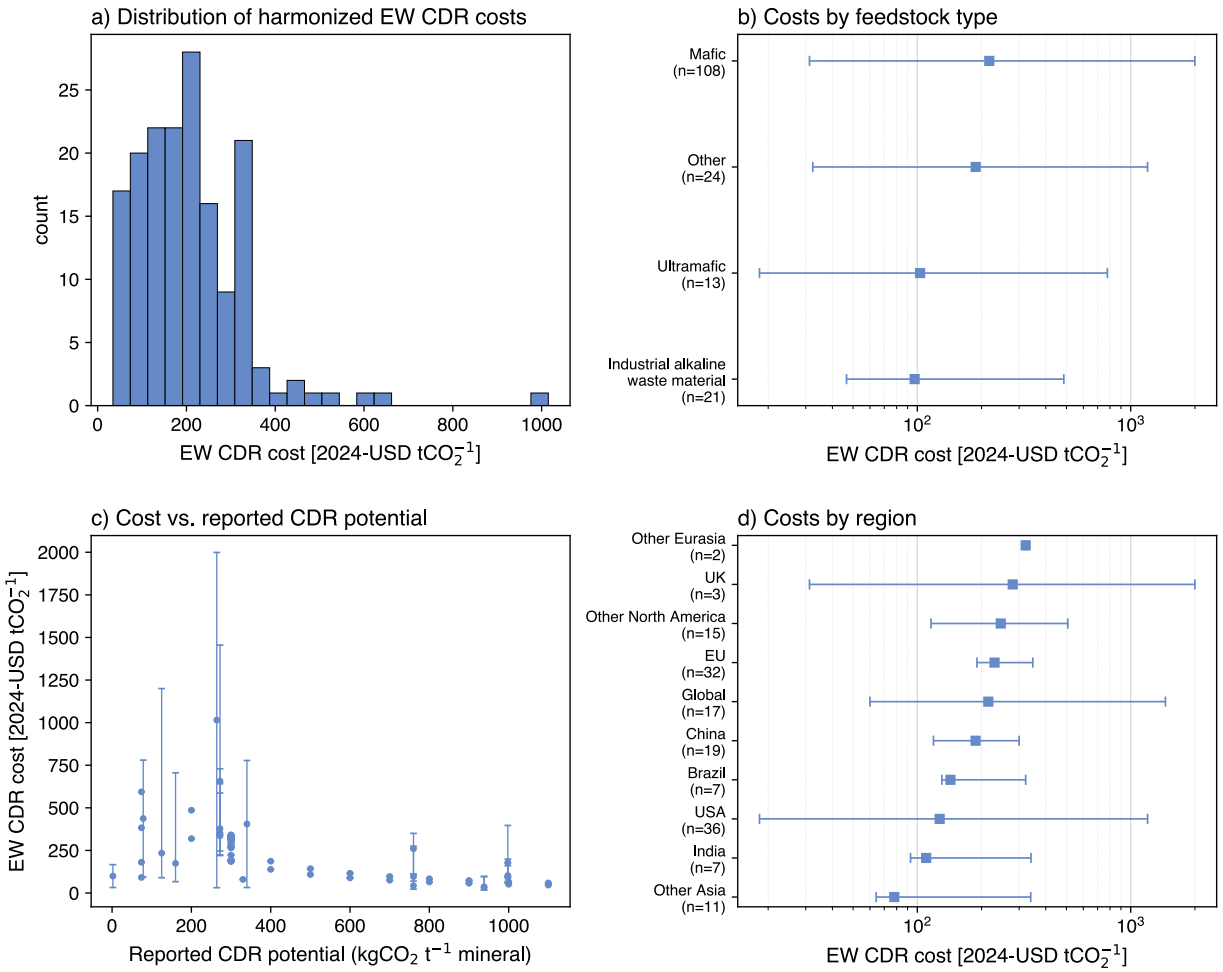

Figure 25: Overview of Enhanced Weathering (EW) Carbon Dioxide Removal (CDR)costs (converted to 2024 USD) by feedstock type, region, and CDR potential. A) Histogram of all extracted scenario-level cost datapoints (central values). Central values correspond to reported best estimates when available, or otherwise to the midpoint of reported min–max ranges. B) Costs by feedstock type. Category-level synthesis for feedstocks with ≥2 datapoints. Markers show medians of datapoint central values; whiskers span the full category envelope (minimum lower bound to maximum upper bound). C) Cost versus reported feedstock CDR potential. Relationship between harmonised cost and reported CDR potential for datapoints where this metric is available. Points indicate central costs; vertical error bars show reported cost ranges. Where no best estimate is reported, the midpoint of the range is used as the central value; datapoints reporting only a central estimate are plotted without visible error bars. D) Costs by world region. Category-level synthesis by region, where markers denote medians of datapoint central values and whiskers indicate regional envelopes (minimum lower bound to maximum upper bound).

From the literature compiled for this review, 25 studies contain EW CDR cost estimates,[6,7,14,15,18–20,22,23,75,78,178,387,414,416,418,419,423,428,430,440,538–541] contributing a total of 166 unique data points (n < 25 because some assess multiple feedstocks, geographies, etc.). These cost data synthesized in Figure 24, with each study summarised by the median of its extracted datapoints. The study-level summary yields a median harmonised (2024 USD) estimate of USD 197 per tCO_2 with an IQR of

USD 137–276 per tCO_2 and a 10th–90th percentile range of USD 85–310 per tCO_2, with study medians ranging from USD 73–1,015 per tCO_2. At a scenario level (e.g., unique feedstock or geography simulations with multiple possible per study; Figure 25a), harmonised central estimates span USD 34–1,015 per tCO_2, with a right-skewed distribution characterized by a median of USD 193 per tCO_2 and an IQR of USD 127–264 per tCO_2 (10th–90th percentile approximately USD 74–335 per tCO_2). These numbers indicate that while favourable cases exist, the literature supports a central tendency of roughly a few-hundred USD per tCO_2. This spread also reflects that studies evaluate materially different deployment configurations and often embed different assumptions about processing intensity, logistics, and credited CO_2 uptake per unit mass of material applied.

6.1 Cost breakdown by feedstock and geography

Enhanced Weathering cost estimates vary across commonly documented study context variables. Nevertheless, feedstock type (Figure 25b) shows substantial overlap among category envelopes, indicating that feedstock labels often act as proxies for bundled assumptions about comminution requirements, transport distances, and credited CO_2 removal per ton rather than functioning as independent cost drivers. Nonetheless, feedstock-level medians are systematically lower for ultramafic rocks compared to mafic feedstocks. Consistent with this, systematic tendencies are apparent in case studies: studies using ultramafic materials or alkaline wastes frequently report lower CDR costs, consistent with estimates that dunite can achieve removal costs near ~60 USD tCO_2^{-1} compared with ~200 USD tCO_2^{-1} for basalt under comparable transport assumptions,[23] and with analyses noting that EW costs depend strongly on grinding, transport, and application logistics.[19] Early conceptual estimates similarly suggested that olivine-based applications could achieve very low costs where materials substitute for agricultural inputs.[16]

Consistent with such feedstock-level differences, the relationship between cost and reported CDR potential (Figure 25c) shows that higher feedstocks with higher CDR potentials (as a result of higher cation content, and often higher weathering rates; see Section 3) coincide with lower calculated costs per ton of CO_2, reflecting the arithmetic effect of distributing similar operational expenditures across larger credited removals. However, wide vertical spreads at comparable potentials indicate that supply-chain assumptions, MRV requirements, and accounting choices remain major determinants of reported costs. For example, MRV expenditures alone can represent a substantial fraction of total EW costs, particularly in early deployment phases.[542] While regional cost ranges overlap, all included Global South locations fall toward the lower end of estimates with median values of <250 USD (Figure 25d), likely reflecting lower labour and associated operational costs as well as potentially more favourable weathering conditions in tropical climates.[19,23,24]

6.2 Cost breakdown by cost component

The cost breakdown by expenditure types clarifies the internal structure of EW economics. The CAPEX–OPEX (capital expenditure – operating expenditure) decomposition shows that studies assign very different shares to capital versus recurring expenditures, but variable OPEX clearly dominates across most datapoints (Figure 26a). Median contributions are 61.6% for variable

OPEX, 36.6% for fixed OPEX, and only 6.1% for CAPEX, and variable OPEX is the largest component in 9 of 13 cases. Substantial within-study spreads (median ≈ 55 percentage points) further indicate that cost structure is highly scenario-dependent even within individual analyses.

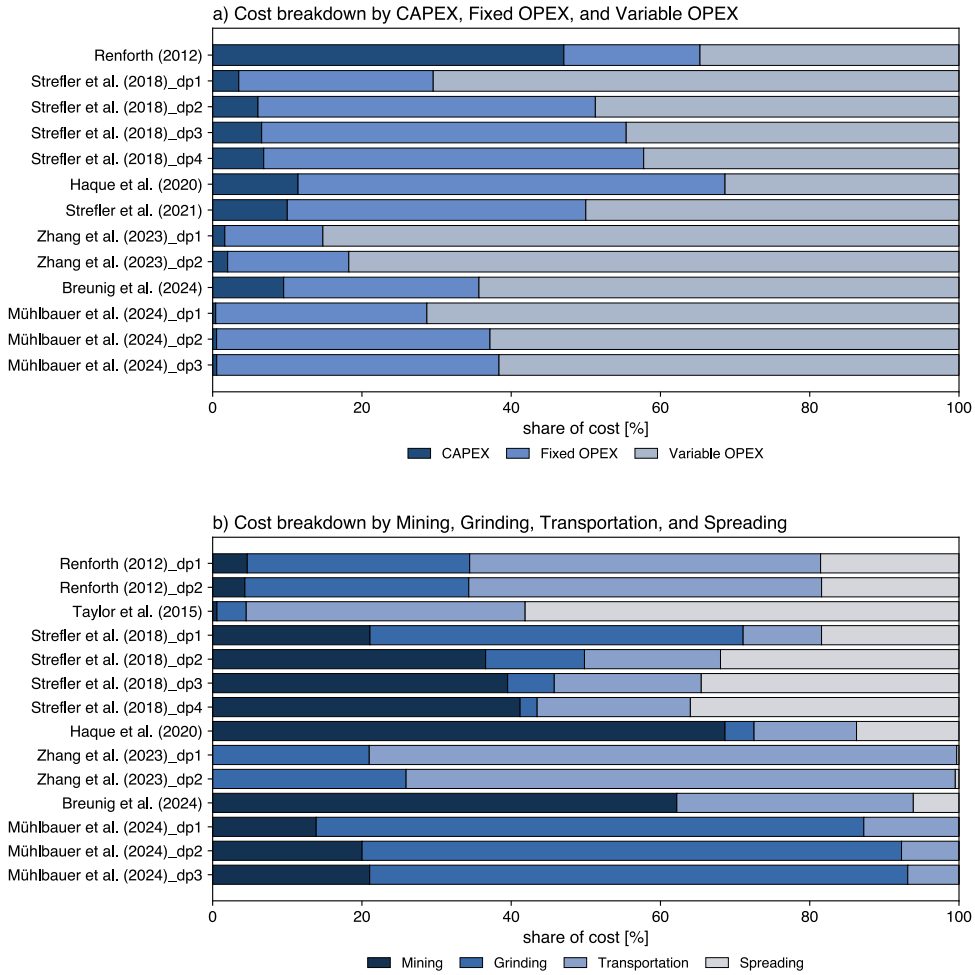

Figure 26: Carbon Dioxide Removal (CDR) cost breakdown. (a) Enhanced Weathering (EW) CDR cost breakdown by CAPEX (capital expenditure), fixed OPEX (operating expenditure), and variable OPEX. (b) EW CDR cost breakdown by mining, grinding, transportation, and spreading. Enhanced Weathering CDR cost breakdown by CAPEX, fixed OPEX, and variable OPEX (in 2024 USD).

The stage-resolved breakdown likewise shows that the dominant cost contributor shifts across datapoints (Figure 26b), with grinding, mining, and transportation each frequently emerging as the largest component, with the exception in Taylor et al. (2015)[414], where spreading is the biggest driver, as they assumed expensive air-spreading (Figure 4b). Median shares are broadly comparable (≈20–23% for grinding, mining, and transport), but variability is large (SD ≈ 23–27%), demonstrating that EW economics are governed by physical system configuration (feedstock type, particle size after grinding, transportation mode, and transportation distance) rather than a single universally dominant step. For example, the share of grinding costs varies significantly in Strefler et al. (2018)[23] among different datapoints due to a range of particle sizes (2 µm, 10 µm, 20 µm,

and 50 µm) assumed, indicating that lower particle sizes after grinding contribute to higher grinding costs (Figure 4b). This is consistent with multiple studies identifying grinding, transport, and field operations as principal cost drivers.[19,180] Quantitative estimates likewise show that individual process steps such as grinding can contribute several dollars per tonne of rock processed,[543] and that transport may constitute the largest single cost component in some scenarios.[428,544] Including the co-benefits of EW may reduce deployment costs,[106,544] and using local mining wastes as feedstock is likely to reduce the costs associated with mining and transport, while the required purification step (froth flotation) introduces a high cost that may equal or surpass these savings.[545]

Finally, MRV is increasingly treated as an explicit cost component rather than an afterthought. A recent report has collected the MRV costs for different CDRs using two datasets: Grantham Research Institute (GRI) and Frontier.[542] According to this report, the average MRV costs for EW have increased from USD 21 tCO_2^{-1} in 2022 to USD 71 tCO_2^{-1} in 2024 based on the Frontier dataset, while the average MRV cost was estimated at USD 15 tCO_2^{-1} in 2024 based on GRI's dataset.[542] The large gap in EW MRV cost estimates between the two datasets underscores that EW MRV is not yet standardised and is highly sensitive to project maturity, scale, and accounting boundaries. Therefore, MRV costs should be treated as a major uncertainty driver in EW cost estimates. Nevertheless, some efforts have been made to reduce the uncertainties in MRV cost estimation. Cascade Climate has released the EW MRV Cost Estimator (EMCE), explicitly structuring EW project costs into two primary direct buckets, i.e., Rock/Hauling/Spreading (RHS) and MRV, and compiling more than 1,000 EW-specific measurement cost estimates from published and sourced quotes to support stacked cost estimation for present-day deployments.[546] Together, these sources suggest that MRV can be a noticeable share of total costs in early EW projects. Over time, MRV costs will depend on how much projects can move from measurement-heavy monitoring to more model-enabled quantification. Measurement, reporting, and verification costs will also depend on how strict the crediting rules are and how much uncertainty those rules allow.

6.3 Price of EW CDR credits on the VCM

Additional information on the economic viability of EW can be drawn from prices observed on the VCM. However, it should be noted that prices are not equivalent to costs: costs reflect the expenditures required to generate and deliver one tonne of credited CDR, whereas prices reflect what buyers are willing to pay and may be higher or lower than underlying costs depending on developer margins, market strategy, subsidies, or temporary loss-making sales.

CDR.fyi's durable CDR market tracking reports that the weighted average price per tonne sold across durable methods fell from USD 490 (2023) to USD 320 (2024),[547] alongside rapid growth in purchased volumes. As shown in Figure 27, prices span a wide range: a small number of purchases occur at very high levels (up to approximately USD 1,600 tCO_2^{-1}), whereas most transactions in 2024–2025 cluster in the mid-hundreds (~ USD 250–450 tCO_2^{-1}). Larger purchases tend to fall within this central band rather than at extreme price levels, suggesting that scaling is occurring primarily at more typical transaction prices despite occasional high-price deals.

Although average prices declined from 2023 to 2024, minimum prices have risen over time, indicating tightening lower bounds as the market matures.

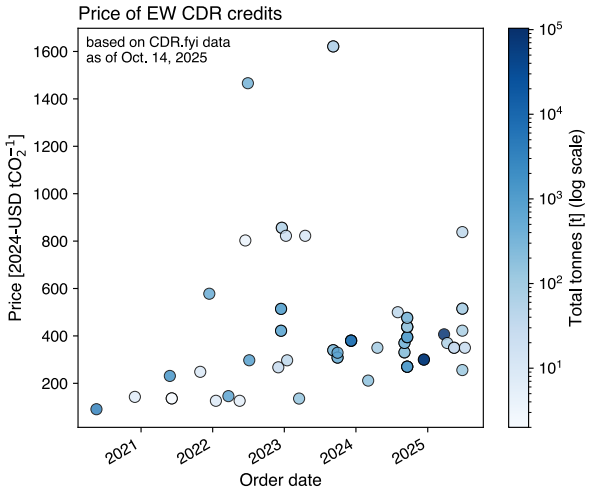

Figure 27: Price of Enhanced Weathering (EW) Carbon Dioxide Removal (CDR) credit sales over time as reported in the CDR.fyi database. All prices converted to 2024 US dollars.

A separate CDR.fyi pricing survey highlights a persistent affordability gap for EW. Suppliers report that roughly USD 349 tCO_2^{-1} is required for profitability, whereas buyers characterized USD 271 tCO_2^{-1} as "expensive" in 2025, with a similar gap projected for 2030.[548] This pattern is consistent with the academic literature: favourable configurations can yield costs in the low hundreds,[23,541] yet, as discussed above, published estimates span a very wide range depending on assumptions and system boundaries and several analyses indicate that deployment becomes economically viable only under sufficiently high carbon prices.[23,178,549]

7 Side effects, co-benefits, and trade-offs

EW on agricultural land has the potential to generate a multitude of side effects beyond CDR.[38,84,279,384,404,422,523,550–553] These side effects may be positive, representing co-benefits that strengthen the case for deployment (e.g., improved soil quality or yields), or negative, representing trade-offs that may constrain scalability, acceptance, or net climate benefit (e.g., emissions of non-CO_2 greenhouse gases (GHGs) or environmental impacts associated with mining and spreading). Understanding the balance, consistency, and context of these effects is therefore central to evaluating EW as a viable climate mitigation strategy.

The side effects discussed in the literature span multiple domains, which we group here into four broad themes: i) social, policy and climate justice, ii) environment and health, ii) agronomic impacts, and iv) CDR and climate. These themes reflect both biogeochemical processes operating at field to catchment scales and broader societal considerations that shape the feasibility and desirability of EW deployment. Together, they capture the diversity of impacts that have been assessed to date and provide a structured framework for synthesizing heterogeneous evidence.

To robustly assess agronomic co-benefits of silicate rock powder additions, we deliberately include not only EW-focused studies but also agronomic studies in which silicate rock powders are applied as soil amendments or fertilizers. This substantially expands the evidence base for agronomic outcomes and reveals overwhelming support for positive agronomic effects, particularly with respect to crop yields, soil quality, and resilience. While these studies are not always framed explicitly in terms of CDR, they provide critical insight into mechanisms, magnitudes, and variability of agronomic responses that are directly relevant for EW deployment.

Figure 28 synthesizes how frequently different side effects are discussed across the literature and whether they are reported as positive or negative. Importantly, a given side effect may be reported as both positive and negative within the same paper. This occurs either because effects are context dependent (e.g., varying across space, time, or experimental conditions), because different indicators yield opposing responses, or because a side effect was explicitly assessed but no net effect was detected, with positive and negative contributions balancing each other. We classify such cases as neutral (mixed) rather than forcing a binary interpretation. In the figure, horizontal bars show the balance of positive and negative mentions, while the y-axis labels additionally report the total number of papers mentioning each side effect and the number classified as neutral (i.e. discussed as both positive and negative within at least one study).

Across themes, several clear patterns emerge. Firstly, social, policy and climate justice side effects are discussed less frequently overall than biophysical impacts. Topics such as public perception, human health, energy demand, and climate justice appear across the literature but are characterized by fewer empirical assessments and necessarily a greater reliance on conceptual discussion, qualitative analysis, or broader systems framing. As a result, these dimensions remain less well constrained than agronomic or geochemical side effects and are addressed in more detail in the following sub-sections, where we discuss their implications for governance, monitoring, and responsible deployment.

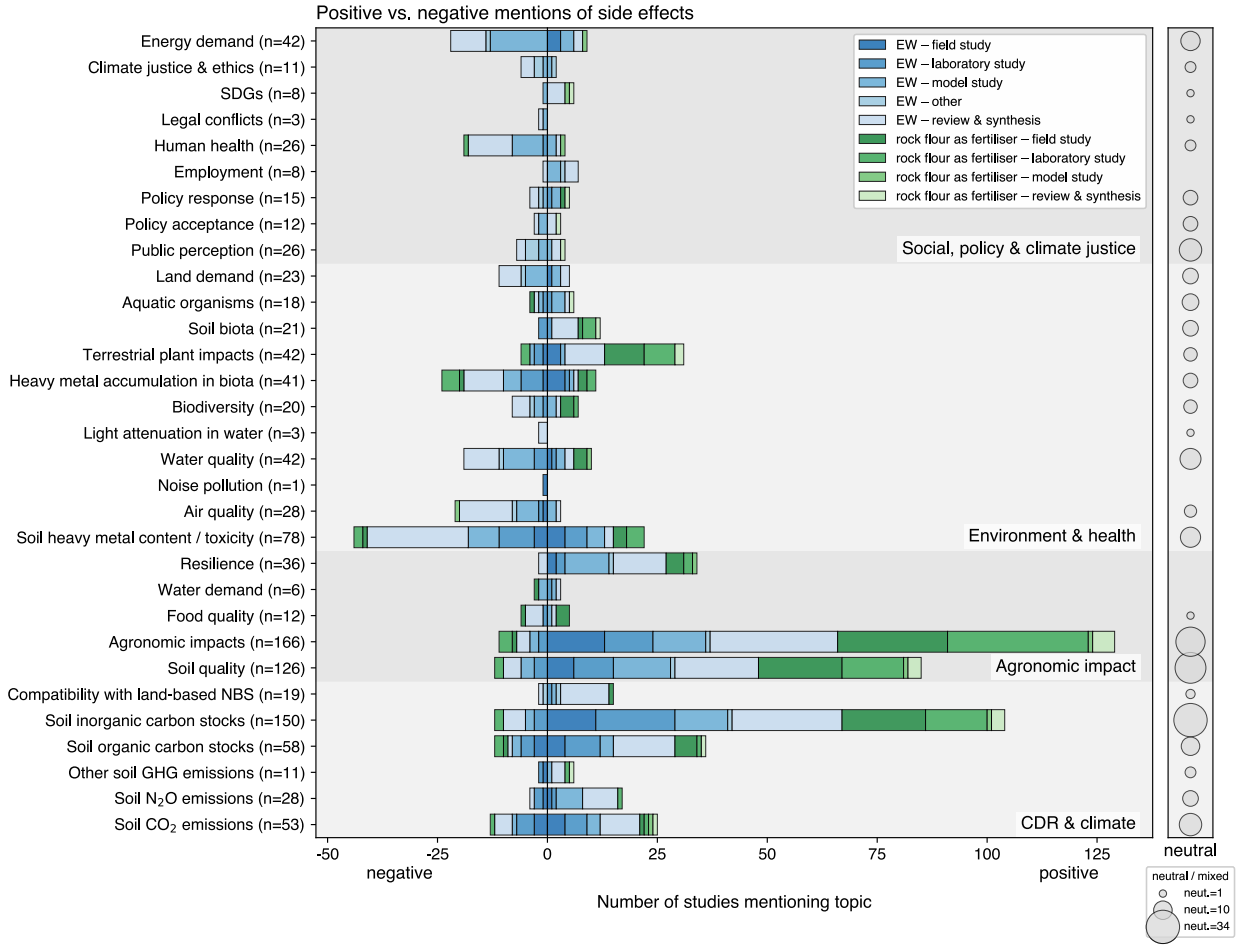

Figure 28: Positive, negative, and neutral or mixed mentions of side effects associated with Enhanced Weathering (EW) and silicate rock powder applications across the literature. Horizontal bars show the number of studies reporting each side effect exclusively as positive (right) or exclusively as negative (left), colour-coded by study type. Studies that discuss the same side effect as both positive and negative are classified as neutral or mixed and are not included in the positive or negative bars; instead, the number of neutral or mixed mentions is indicated by the size of the circle in the box on the right. Neutral or mixed classifications therefore reflect context-dependent or contrasting outcomes within a study, rather than the absence of an effect. Side effects are grouped into four themes: CDR & climate, agronomic impacts, environment & health, and social, policy & climate justice.

The environment and health theme shows a more heterogeneous and, in some cases, negative-leaning profile. However, this pattern is driven primarily by reviews and syntheses, whereas experimental studies more often report positive effects or, in some cases, no detectable negative impacts. Heavy metal content in soils, air quality impacts associated with dust generation, water quality, and metal accumulation in biota are among the most frequently discussed topics in this category. Notably, the overall negative trend for heavy metals is driven primarily by reviews, perspective papers, and risk-oriented assessments that highlight potential contamination pathways, rather than by experimental field or pot studies demonstrating systematic increases in soil or plant metal concentrations following silicate rock powder application. In contrast, several biological indicators within this theme—particularly impacts on terrestrial plants and soil biota—are more

commonly reported as positive, reflecting improvements in plant performance and microbial or faunal activity under certain conditions. Together, these patterns suggest that while environmental risks are an important focus of the literature, consistent with overall positive agronomic impacts, empirical evidence also points to beneficial biological responses, emphasizing the need for context-specific evaluation rather than uniform assumptions of harm.

In contrast, agronomic impacts are the most frequently discussed side effects overall and show an overwhelming dominance of positive-only mentions. Agronomic performance, soil quality, and resilience are consistently reported as improved following silicate rock powder additions. Importantly, this support is not driven solely by modelling studies, conceptual analyses, or narrative reviews, but is substantiated by a large body of experimental field and laboratory studies that directly measure crop responses and soil properties. Negative-only assessments within this theme are rare, and neutral classifications primarily reflect context-dependent outcomes rather than systematic trade-offs, indicating broad and empirically grounded support for agronomic co-benefits.

Finally, within the CDR and climate theme, SIC stocks are the most discussed outcome and are predominantly reported as positive. This pattern is consistent with the intended climate mitigation function of EW and reflects both mechanistic understanding and observational evidence linking silicate dissolution, alkalinity generation, and carbon stabilisation. Soil organic carbon and GHG effects are more mixed and not as well studied as inorganic carbon stocks. Overall, while some neutral classifications occur in this category, often related to variability in rates, lag times, or methodological uncertainty, negative-only assessments are comparatively uncommon, suggesting general alignment between theoretical expectations and reported outcomes.

Together, these patterns indicate that while agronomic and climate-related co-benefits of EW are relatively widely supported, particularly by experimental evidence, environmental, health, and societal impacts remain more uncertain and context dependent. In the following sections, we therefore examine several of the most consequential side effects in greater depth, focusing on mechanisms, sources of variability, and implications for monitoring, reporting, and responsible deployment. These sections generally follow the themes described in Figure 28, though there are some different areas of emphasis. For example, section 7.1 focuses on the social effects, with some discussion of human health impacts, but also a significant discussion of economic and policy impacts. The environment & health section (7.2) addresses impacts on biodiversity and organisms, water quality, air quality, and soil heavy metal concentrations and toxicity. The agronomic effects section (7.3) emphasizes findings related to yields, where much research has focused, as well as exploring the relationship of EW to NUE, soil pH, BS, CEC, and soil physical properties. Finally, section 7.4 addresses facets of CDR and climate effects not addressed elsewhere, including impacts on soil inorganic and organic carbon and on non-CO_2 GHGs.

7.1 Social, policy and climate justice effects

Though comparatively understudied, the balance of social (see Figure 28), political, and environmental justice effects—as well as broader impacts on ecosystems, economies, and human health—is central to assessing the real-world viability of EW. Beyond technical performance and geochemical potential, EW deployment will be shaped by how benefits and risks are distributed

across regions, sectors, and communities, and by the extent to which governance frameworks can ensure equitable and sustainable outcomes. These dimensions are particularly salient given the strong coupling of EW to land use, extractive industries, and agricultural systems, all of which are embedded in complex socio-political contexts.

The following subsections synthesize emerging evidence on employment, economic impacts, resilience, sustainable development linkages, human health, and policy and legal considerations. Several closely related issues such as public perception, stakeholder engagement, and broader climate justice considerations are addressed in greater detail in Section 10, with policy frameworks and implementation pathways explored further in Section 13.

7.1.1 Employment

There has been minimal discussion about the prospective benefits for employment and job creation in the literature. Where discussed, this is considered for the broad set of geochemical CDR technologies,[554] the potential to create jobs in mining communities[22,541,555] and strong connections between EW and the agricultural and forestry sectors.[385] To our knowledge, only Beerling et al. (2025)[22] has attempted to provide an estimate for job creation, suggesting that demand for up to 2 Gt of basalt could require 60,000 to 80,000 more workers to be employed in the U.S.

7.1.2 Economic, social, and climate resilience

Research has also identified and examined other vectors through which EW can improve resilience of socio-economic systems, including improved resistance to pests, diseases, and drought,[69,385] and more broadly difficult weather and planting conditions.[118] Overall, such resilience is understood to foster sustainability and food security,[69,180,385] including among smallholder farmers and in tropic regions.[173,243] Such opportunities may be challenged by the high economic costs from mining, grinding, and transportation of sourced rock and potential environmental risks, including from the release of heavy metals[180]—though this could possibly be mitigated by greater coupling with low-cost, efficient quarrying production, where available.[100] Seen through the lens of net-zero transitions, EW (and CDR technologies in general) is also envisioned to reduce potential disruption to the electricity and industry sectors, particularly for regions and countries more exposed to and dependent on fossil fuels.[416,419]

7.1.3 Further economic impacts

Economic impacts principally focused on potential benefits for farmers, notably how EW could result in a yield increase[523,556,557](cf., Section 7.3.1) or decreasing fertiliser costs[125,180,385,444] and other production inputs.[558] For instance, Haque et al. (2024)[556] calculate for their field trial with smallholder farmers in Kenya a potential revenue increase of USD 326 per hectare. There were also more forward-looking discussions of the carbon credits that could be generated for those farmers or regions supplying CDR, if and when global trading of emissions permits is allowed [78]. Conversely, Oppon et al. (2023)[559] raise concerns about unequal distribution of economic benefits of EW from developing to developed countries, if the former are increasingly used for siting and exporting silicate rocks to the latter.

7.1.4 Sustainable development goals

The development and deployment of EW are linked to the pursuit of a variety of SDGs. Most frequently, positive synergies with climate action (SDG13) and zero hunger (SDG2) are noted, by removing carbon dioxide and enhancing crop yields.[38,117,387,560–562] Potential increases in food production and lower need for traditional fertilizers are further connected to responsible consumption and production (SDG12), life below water (SDG14), life on land (SDG15) as well as no poverty (SDG1) and decent work and economic growth (SDG8).[84,387,560,563,564] Using a systematic mapping of the relationship between EW and the SDGs, Honegger et al. (2021)[560] however emphasize that its deployment could have adverse consequences from the need for more mining—i.e. on good health and well-being (SDG3), affordable and clean energy (SDG7), and life on land (SDG15)—and impacts on ecosystems and the environment—i.e. clean water and sanitation (SDG6) and life below water (SDG14) (see also ref. [565]). Thus, the relationship between EW and the SDGs will ultimately depend on the extent to which it is done in a sustainable fashion.[563,566]

7.1.5 Impacts on human health

Impacts on human health principally relate to harm from heavy metal toxicity on agricultural lands or leaching into water sources[129,179,251,385,567,568] and inhalation risks of dust and particles along the supply chain (cf., Sections 3.5 and 7.2).[22,131,209,248,426,568] Besides risks pertaining from heavy metal exposure (see also [129,131]), the majority of the adverse impacts to human health were tied to transportation and comminution processes.[209,250,426,427] As a result, Feng and Hicks (2023)[250] examine how varying rock types, application rates, transportation distance, and geographic application area could help to mitigate the environmental and health impacts. Regarding inhalation risks, the need for precautionary approaches and the availability of safety measures to mitigate such risks are also highlighted.[22,131,568]

7.1.6 Policy acceptance

Limited attention to public perceptions and acceptability of EW is reflected and exacerbated when it comes to perceptions of and support for relevant policies. On a general level, the need for policies to be perceived by the public as just and equitable is noted.[554] One global, large-scale survey exercise in thirty countries has revealed that support is generally highest for two types of CDR policies: national-level support and funding for public and private R&D and public information and engagement campaigns.[569] In contrast, there was the least support for independent national restrictions on use of CDR as well as for a global-level market for trading carbon credits and offsets. Respondents in the Global South were also much more supportive of a range of CDR-related policies—separate analysis of the dataset also found broadly higher support among indigenous and minority-group respondents.[570] Specific to EW, a companion set of focus-group exercises in 22 countries identified a preference for top-down, institution-led decision-making approach.[571,572] The greater understanding that EW decisions are better left to experts or regulatory authorities distinguished it from ecosystem-based approaches such as soil carbon sequestration as well as DACCS and Bioenergy with Carbon Capture and Storage (BECCS)—consultation with affected publics was perceived to be more crucial for the latter two. Indeed, a specific need for governments to provide coordination between sectors such as land-use and extractive industries as

well as across a variety of levels and regions and to conduct assessment on behalf of publics was a common discussion point in focus groups—though EW received relatively less attention than some of the other methods.[571] At the same time, questions about trust and the involvement of big business actors were frequent,[573,574] with members of the public tending to place much more trust in experts.[572] Taking a company perspective, some studies have discussed the importance of supportive policy frameworks and regulatory approaches for promoting EW adoption,[575] along with modelling how government policies influence company decisions about how to transport EW materials.[576]

7.1.7 Policy response

Discussions of effective or appropriate policy frameworks for EW remain a nascent topic in the literature. One of the most frequent sub-topics is the lack of long-term policy certainty and the divergence of frameworks (and protocols) between jurisdictions, stressing how this presents barriers to scaling-up EW.[2,552,575,577,578] Thus, Mercer et al. (2024)[577] recommend that policies be developed which are specific to each CDR method along with policy frameworks that are internationally aligned yet flexible to regional specificities. An example of a type of regulation which can be more generally employed is that for heavy metals in rock powder, already in force in countries like Brazil and Germany.[129] Transparent policies and greater financial support for R&D to reduce costs of MRV have been proposed,[577] while one survey exercise also found broad public support for national-level funding for R&D on CDR in general.[569]

Meanwhile, the need for region- or state-specific frameworks was supported with reference to regional and geographic differences in sequestration capacity and other resources.[579] Publics in one broad set of focus groups similarly noted the need for national governments to take the lead—and coordinate with one another—around decisions on how to employ their own advantages and develop a portfolio of CDR approaches.[571] Additional recommendations, and outstanding subjects of discussion, include how to integrate EW into GHG inventory methodology (e.g., ref. [509]), the extent that such activities are aligned with international mechanisms like Nationally Determined Contributions, notably for countries in the Global South,[580] and whether stocks of inorganic and organic carbon should be managed separately for carbon crediting purposes.[100]

7.1.8 Legal dimensions

Discussions of legal and governance issues are notable more for the tendency to highlight that such issues are under-represented or that such regulations are broadly absent.[38,129,131,509,581] Specific issues that are highlighted include the regulation of mine tailings [582] and risks of infringing heavy-metal regulations;[129] how legal barriers might impact the scaling-up of EW deployment;[554] and issues of responsibility, liability, and institutional accountability and how these signal the need for a dedicated EW legal framework.[129,581] The importance of international-level coordination as well as global frameworks and monitoring standards is also underscored, both for successful implementation of EW[583] and to avoid exacerbating climate justice risks and outsourcing pollution and damages (e.g. from mining) to the Global South.[131,565,580]

7.2 Environment and health effects

Beyond its CDR potential, EW can influence environmental quality and biological systems across the full deployment chain, from rock extraction and comminution to land application, weathering, and downstream transport of dissolved and particulate products. These effects can be beneficial, for example through soil improvement, acid neutralization, or enhanced nutrient supply, but may also introduce risks related to biodiversity, water and air quality, and trace metal release. The balance of these outcomes depends strongly on feedstock properties, application rates, local environmental conditions, and the spatial and temporal scale considered. The following sections therefore assess current evidence on the environmental and health implications of EW across terrestrial and aquatic systems.

7.2.1 Impacts on biodiversity and organisms

Impacts of EW on biodiversity and organisms may start at the source: concerns have been raised about biodiversity losses resulting from the upscaling of mining activities that will be necessary for large-scale EW,[541,580] potentially contributing to deforestation in tropical regions.[385] As shown in Figure 28, organism-related outcomes differ from the broader environment and health theme, which is overall more negative-leaning due to risk-focused categories such as heavy metal toxicity, air quality, water quality, heavy metal accumulation in biota, and land demand. In contrast, terrestrial plant impacts and soil biota are more frequently reported as positive than negative, while biodiversity and aquatic organism responses are more evenly split or often classified as mixed. This highlights that biological responses to EW are strongly context-dependent and should not be inferred directly from broader environmental risk categories.

7.2.1.1 Vegetation

Out of all biota, the impact of EW on vegetation has been most extensively studied. While the impacts of EW on yield and food quality are discussed elsewhere in this review, there are effects on plant health and biodiversity that go beyond these. For example, in a greenhouse experiment, applying rock dust either alone or in combination with a commercial organic fertiliser significantly reduced the severity of tomato bacterial wilt compared to the organic fertiliser alone[584] Another study showed that herbaceous species richness increased following rock dust application in a nutrient-depleted, acidified forest soils.[514] Both field observations of soils developed on artificial silicates[95] and pot experiments with freshly collected tephra[585] indicate quick succession, consistent with a fertilising effect of these materials. At the same time, concerns have been raised about negative impacts on biodiversity in forest edges due to rock dust spread on adjacent farmlands, especially for species adapted to low-pH soils.[385] Finally, a change in the rate and type of nutrients that is released by rock dusts compared to conventional fertiliser on grasslands can increase forage quality and thus cattle health, mostly due to an increase in Mg and/or Ca. This can be the case either when Mg-bearing rocks are applied[586] or when the K source is biotite, K-feldspar and/or nepheline instead of KCl salts.[587]

7.2.1.2 Microbes

In terrestrial systems, studies have mostly investigated the impact of EW on either the microbial community or on earthworms. In general, given that soil pH is one of the most important factors

determining the microbial community, increases in soil pH resulting from rock dust application are expected to affect microbial abundance and activity.[94,131,180] In a long-term liming experiment with a pH range spanning from 4.0 to 8.3, relative fungal abundance did not correlate with pH, while fungal diversity was only weakly related with pH. In contrast, relative bacterial abundance and diversity strongly increased with increasing pH.[588] This suggests that rock amendments may more strongly stimulate bacterial communities, particularly in acidic soils.[131] Indeed, microbial biomass carbon was significantly higher following wollastonite application in two acidic soils, while no significant difference compared to the control was observed in the alkaline soil.[114] Microbial growth may be additionally enhanced by rock dust application due to enhanced nutrient supply and mobilisation, stronger heavy metal immobilisation, and increased plant growth which stimulate root exudate formation.[589] Following basaltic rock powder amendment in temperate vineyard soils, basal respiration, a widely-used indicator for microbial activity in soils,[590] had increased by 50% compared to non-amended soil one-month post-application.[475] Finally, in a sandy soil Ca provided by the addition of fine dolerite was found to stimulate microbially induced calcite precipitation more than adding Ca as calcium chloride, further contributing to inorganic C sequestration.[489]

Besides overall abundance and activity, also microbial community composition may shift as a result of rock powder application. Studies in both a tropical rubber plantation[487] and a pot experiment[591] showed little changes in soil microbial diversity, but did observe changes in the microbial community. While in Wang et al. (2025)[487] a richer and more interconnected microbial network was observed following two years after wollastonite amendment, driving changes in soil multifunctionality, in the experiments of Reis et al. (2024)[591] the specificity of the bacterial community to the different rock powders applied was found to increase with time over the course of the 8-month experiment. In a two-year pot experiment using two nutrient-poor agricultural topsoils, Ramezanian et al. (2015)[592] observed a minor shift in the soil microbial community composition following rock dust application, which they attributed to Ca and K released by the rock dust. In contrast, Campbell (2009)[593] observed few significant changes in soil microbial communities in a three-year field trial.

Soil enzyme activities have mostly been shown to increase as a result of rock powder application. Following wollastonite applications in a tropical rubber plantation, statistically significant increases in root phosphomonoesterase, an enzyme released by plants to acquire P from organic molecules, were observed after two years.[594] In a pot experiment growing tomato plants for 30 days in an acidic yellow-brown soil, the combined application of rock dust and a commercial organic fertiliser significantly increased the activities of alkaline phosphatase, urease, catalase and sucrase compared to the control, while the latter two were also enhanced when only rock dust was applied.[584] In a greenhouse pot experiment of 120 days growing rice in slag-based silicate fertiliser, Das et al. (2019)[595] observed significant increases in both hydrolytic and oxidase enzyme activities.

Despite fungi having a wider pH optimum than bacteria, increased pH may still positively affect both fungal growth and diversity, as has been suggested for arbuscular mycorrhiza.[596] Indeed, the few experimental studies investigating fungi did observed significant changes, although results differ between studies. In the same greenhouse pot experiment as described above, Das et al. (2019)[595] found significant increases in both fungal species richness and diversity following slag-based silicate fertiliser, with further analyses revealing that the relative abundance of saprotrophic

fungi (fungi that decompose organic matter), but not symbiotrophic (e.g. mycorrhiza) and pathotrophic fungi, had significantly increased. This is in contrast to the results of Bi et al. (2024)[594] who observed significant increases in mycorrhizal colonization in the same field experiment on a tropical rubber plantation as above.

7.2.1.3 Earthworms

There is a keen interest in earthworm responses to rock dust application, e.g. due to their key role in ingesting, fragmenting, mixing and transporting inorganic and organic matter in soils, which may further stimulate weathering.[69,597] Studies to date have shown either neutral or positive effects on survival and activity of the various ecological types of earthworms. A single application of 20 t ha^{-1} of basaltic rock powder on temperate vineyard soils resulted in an average increase in earthworm abundance of 71% after one month.[475] In contrast, in a mesocosm experiment, no changes in the survival of endogeic (*Aporrectodea caliginosa*) or anecic (*Lumbricus terrestris*) earthworms were observed after basalt amendment, while earthworm activity was suggested to be enhanced.[91] Enhanced earthworm activity, as indicated by a twentyfold higher number of earthworm casts compared to non-amended plots, was observed six-month post-application of basaltic rock dust in a field experiment.[598]

As with the microbial community, the change in pH is likely a key driver in these effects[131] given that the optimal pH for earthworms is slightly acidic to neutral,[94] but changes in other soil properties may also be relevant. Variables influencing the structure and drainage potential were found to dominantly control survival and activity of the endogeic species *A. calignosa* and *Allolobophora chlorotica* in artificial organo-mineral systems, with best conditions achieved when using straw, basanite rock and coarse-grained rock dust.[599] In addition, heavy metals released from rock powders may be immobilised in earthworm bodies,[94] as shown in a vermicomposting experiment.[600] Yet, these authors also found that this immobilisation correlated with higher earthworm growth and reproduction, in line with results from an earlier experiment which showed neutral to positive effects on earthworms when using silicate rock powders for vermicomposting.[163]

7.2.1.4 Aquatic organisms

In freshwater ecosystems, changes in carbonate chemistry resulting from EW may alleviate acidification and modify ecological responses to nutrient enrichment resulting from e.g. agricultural activities, atmospheric deposition, wastewater input and acid mine drainage.[131,179] In such systems, higher pH and alkalinity have been suggested to reduce primary productivity in some contexts and support more diverse food webs, and more nutrient-rich phyto- and zooplankton, particularly in nutrient-rich ecosystems downstream of tropical agricultural areas.[131,385] However, photosynthesis by aquatic plants and phytoplankton in rivers is generally suggested to be stimulated after receiving terrestrial EW products, with multiple possible mechanisms proposed but not confirmed.[67]

In marine ecosystems, EW-derived alkalinity could partially alleviate some impacts of ocean acidification on organisms, although the dominant source of ocean acidification is enhanced atmospheric CO_2.[38] Shifts in carbonate chemistry resulting from EW, most notably an increase in

aragonite saturation state (Ω_{arg}), may benefit calcifying organisms, such as corals, by increasing carbonate saturation state.[131,415] The extent to which coral reefs may still be above the critical Ω_{arg} threshold of 3.5 by 2100 depends on both the scale of EW deployment and the CO_2-emission scenario used in future projections with Earth System Models.[279,413] The exact impact on marine food webs also depends on the type of rock dust that is used. With carbonate feedstocks, growth of calcifying phytoplankton such as coccolithophores is stimulated at the expense of both diatoms and cyanobacteria.[38] In contrast, when silicate rock dust is used for EW, it can act as a source of Si and alter the Si:N and Si:P ratios of waters, favouring the growth of diatoms and affecting the biological pump.[17,385] Also cyanobacterial growth may be stimulated as a result of the Fe and/or Ni released from silicate feedstocks.[38] Trace metals released during EW may however have both positive and negative effects on marine organisms in multiple ways. In a lab experiment using peridotite and basalt rock powder, Zhang et al. (2024)[601] found that the Ni, Fe and Co released from the rock powder stimulated growth and methanogenesis of a model methanogen, while the observed levels of Ni are suggested to be harmful to some marine phytoplankton. In contrast, Cong et al. (2024)[180] highlight the negative effects that Ni and Cr have on organisms via interrupting ion regulation, having toxic effects on the respiratory system, and causing oxidative stress.

Taken together, the evidence reviewed across terrestrial, freshwater, and marine systems is consistent with Figure 28: terrestrial organism responses are more frequently positive or mixed, whereas aquatic responses and broader biodiversity outcomes remain more uncertain and strongly context dependent.

7.2.2 Impacts on water quality and demand

EW can influence freshwater use, water quality, and freshwater ecosystems both during rock mining, comminution, and transport, and after land application when weathering occurs. Life cycle assessments have largely focused on the former, while experimental and modelling studies have emphasized the latter.

7.2.2.1 The role of water in mining, comminution, and transport of rock

Water is used throughout the EW life cycle both directly (e.g., dust suppression and rock-mill cooling) and indirectly via its involvement with energy production. Due to both direct and indirect water use, mining and comminution are the most water-intensive activities within the life cycle. The specific context, such as the energy mix and rock demand resulting from weathering efficiency, influences which of these two activities dominates water use.[426,602] However, the total water requirement for EW is small compared to existing freshwater withdrawals (<0.25%) and 10–100x lower than that of other CDR technologies like afforestation, DACCS, and BECCS.[426]

In LCAs, EW indirectly impacts freshwater quality through its reliance on fossil-fuel energy sources. The mining and combustion of fossil fuels releases pollutants—including nitrogen, phosphorus, and heavy metals—that cause eutrophication and ecotoxicity within freshwater systems. Thus, comminution (an energy-intensive activity) and transport processes that rely on fossil fuel are responsible for most of the negative effects on freshwater quality.[209,426,602] These impacts scale with operational variables such as grind size and transport distance and diminish as electrical grids and transport networks transition away from fossil fuels. Overall, these indirect

impacts on freshwater quality are small. For context, one study found the entire environmental footprint of mining, grinding, transporting and spreading rock in a coastal EW context was equivalent to 0.3% of an average European inhabitant's annual environmental load, with freshwater ecotoxicity and eutrophication contributing only a small fraction of that total.[602]

7.2.2.2 Effects on water quality and demand during weathering

The weathering process itself requires water, but the amount needed for weathering is substantially less than that required for crop growth. Thus, when practiced on agricultural fields, EW does not noticeably affect water demand. Rock application holds the possibility of affecting field hydrology by altering soil hydraulic conductivity via the formation of clays[63] or by altering plant-water use efficiency via the introduction of plant-available Si.[385] But these potential impacts are not well established, and at this point, are either only theorized or captured in isolated column studies.

Rock weathering can have both positive and negative effects on downstream freshwater quality. First and foremost, rock weathering adds alkalinity to a system, which eventually reaches downstream freshwater systems where it can neutralize acids and increase pH. Acid-neutralization has been detected in leachate of mesocosm experiments[179,237] and at the catchment scale at Hubbard Brook where wollastonite application (~3,800 kg/ha) increased stream pH and acid neutralizing capacity over at least six years.[603]

In locations affected by acidification processes (e.g., acid rain), acid neutralization improves water quality. At Hubbard Brook, for example, the pH increase resulted in a decrease in stream water aluminium concentrations and reduced toxicity risk to aquatic organisms.[72] However, the pH response to alkalinity additions depends on existing river chemistry, and at higher rock application rates, modelling studies show it is possible to push river pH above optimal thresholds for ecosystem health, suggesting that deployment limits might need to be set based on river pH thresholds.[17,65] Further, the ecosystem response to increased alkalinity and pH is unclear and likely site specific. These co-occurring water chemistry changes could alter food web community structure, rates of aquatic photosynthesis, and metal bioavailability to both positive and negative outcomes.[15,67,131,385]

Weathering also releases Si, which researchers have theorized could help mitigate eutrophication.[15,106,385] Past work has established that increased Si:P and Si:N ratios tend to support diatom populations over harmful algae blooms. Thus, the reasoning is that the addition of Si via weathering could positively alter nutrient ratios and reduce toxic algal blooms, particularly in agricultural landscapes where N and P are in ample supply. However, one of these reviews[15] also notes that changes in Si concentrations and altered nutrient ratios could be problematic for surface waters, highlighting the uncertainty in the biological ecosystem response to water chemistry changes.

Trace metal contamination is a persistent concern for EW and is one of the main reasons why the environment and health theme in Figure 28 shows a negative-leaning profile. Column, mesocosm, and pot studies indicate that silicate or alkaline mineral amendments can increase aqueous trace metal concentrations, particularly Ni and, in some cases, Cr, in soil porewaters or leachates.[63,179,184,223] However, reported concentrations vary widely and often remain low or below

regulatory thresholds, with exceedances or near-threshold values occurring mainly for olivine-rich or metal-rich waste-derived feedstocks. As such, multiple reviews and policy documents call for close tracking and further study of metal release from EW into downstream waters.[15,69,180,383,387,541] This risk-oriented emphasis contrasts with some experimental studies in which aqueous metal concentrations remained below levels of concern,[131] and with evidence that metal bioavailability can decrease where EW raises soil pH or promotes secondary mineral formation (cf., Section 7.2.4). Ultimately, the literature indicates that metal risk depends on feedstock choice, especially the use of olivine-rich or waste-derived materials; particle size, with finer particles potentially releasing more metals initially; and timescale, because metals may either accumulate through repeated applications or become immobilised in secondary phases over time.[98,222,436,518] These factors suggest that metal release risks can be reduced through careful feedstock selection, particle-size optimisation, conservative initial application rates, and long-term monitoring of soil, porewater, leachate, and biomass metal concentrations.

7.2.3 Impacts on air quality

As highlighted in Section 3.5, airborne particulates represent an important exposure pathway for EW, particularly during mining, crushing, grinding, transport, storage, spreading, and post-application erosion of fine rock powder. This is also reflected in Figure 28, where air quality is one of the more consistently negative-leaning topics within the environment and health theme. However, direct EW-specific evidence remains limited, and most current understanding is inferred from industry analogues, including silicate dust erosion, stone quarrying, rock crushing, road transport of crushed limestone, and agricultural liming.[131]

Inorganic mineral dust can have severe impacts on human health, depending on particle size, composition, and exposure level.[395] The most severe outcomes are expected when particle size is fine (<10 μm)[131,396] and when exposure is chronic or severe.[604] Mineral structure and heavy metal content may also affect outcomes.[387,390,395] Health outcomes can include pneumoconiosis, cancer, systemic poisoning, hard metal disease, irritation and inflammatory lung injuries, allergic reactions, and irritation or infection of skin.[396,397] Silicate dust in exceedance of air quality standards has been associated with increased hospital admissions and mortality in Iceland and Spain.[605,606]

Crushing and grinding of rock in limestone operations can increase particulate pollution in the immediate vicinity, but with considerable variability depending on wind direction and operational layout.[607,608] Air pollution rapidly improved with distance from the source,[396] falling below detection against background levels within 100s of meters.[607] There is very limited evidence that granitic quarries produce less dust than limestone ones, but comparison of additional sites with fully analogous operational requirements is needed.[607] Transportation dust is more difficult to measure[609] and is believed to be most significant on the quarry site itself, where it re-entrains dust into the air.[610] Meta-analysis suggests that on-site hauling alone can raise dust concentrations well past hazardous US air quality thresholds.[396] Transport from mine to application site will also increase airborne particulate concentrations, especially within a 30m distance from the haul road or where dirt roads must be used.[393,394]

EW may pose inhalation hazards during in-field application, particularly where fine, silica-containing powders are spread under dry or windy conditions.[15,131,383] These risks can be reduced through established dust-control and occupational-safety measures, including avoiding excessively fine particle-size distributions where compatible with CDR performance, moisture conditioning, enclosed handling or dust collection during processing, enclosed tractor cabs, appropriate personal protective equipment, and avoiding spreading during high-wind or low-humidity conditions.[20,131,387,391,392] These measures are discussed more broadly in Section 3.5.2, including baseline air-quality assessment, safeguard thresholds, setback distances, monitoring, and adaptive management.

Enhanced Weathering may also affect air quality indirectly through changes in emissions of reactive nitrogen species. Increases in soil pH associated with EW have been hypothesized to reduce soil emissions of nitric oxide and nitrous oxide[19] Reduced nitric oxide emissions could in turn lower the formation of ground-level ozone. However, other analyses suggest that nitrogen oxide emissions associated with transportation of rock material have the potential to net elevated ground-level ozone concentrations.[385]

Mining and crushing of rock can result in hazardous noise pollution where additional rocks are processed (i.e., where EW does not utilize existing waste streams). Although noise pollution is classified separately in Figure 28, it is closely linked to the same upstream activities that generate particulate air-quality risks. Elevated noise levels during mining and blasting activities are well documented, with demonstrated impacts on both mine workers and surrounding communities.[392] Stone grinding and crushing operations in particular have been identified as high-intensity noise sources, with measured sound levels reported in the range of approximately 100 dBA.[611]

Noise impacts may be mitigated through operational and engineering controls, including limiting the use of explosions or restricting blasting to designated working hours.[392] Structural interventions, such as the construction of high-inertia foundations, may reduce vibration and associated noise transmission.[611,612] Additional mitigation options include insulating or physically isolating worker areas to reduce occupational exposure.[611] Overall, air-quality and noise impacts are likely to be most important near points of material extraction, processing, transport, and application, and should therefore be evaluated through site-specific exposure assessment and the mitigation and monitoring safeguards outlined in Section 3.5.2.

7.2.4 Impacts on soil heavy metal concentrations and toxicity

Enhanced Weathering will inevitably introduce trace metals contained in feedstocks to soils. Many metals are important micronutrients but can be toxic at higher concentrations. Metal release is therefore one of the key environmental concerns associated with EW, and may lead to adverse effects on plant growth, aqueous ecosystems, and food safety in some conditions (e.g., refs. [15,131]; Figure 28). To assess risks, the key question is to what extent feedstock derived metals increase *bioavailable* metal pools, mobilise to waters, or transfer into biomass at agronomically and CDR-relevant application rates. Potential impacts strongly depend on feedstock metal content, which can vary widely. Of key concern are feedstocks strongly enriched in metals compared to typical soil background levels, including ultramafic feedstocks, rich in Cr and Ni, and some industrial by-products (e.g., refs. [106,109,129,248,567]). If such high-metal feedstocks would be unsuitable for

widespread agricultural deployment of EW, this would substantially constrain feedstock availability and may limit its global CDR potential.[130,415]

Experimental results are highly variable but, overall reassuring at the tested rates, with limited reports of breaching of environmental regulations. A systematic, semi-quantitative assessment conducted here (Figure 28) indicates that concerns about heavy metal accumulation are more frequently discussed in a negative context for both soils and biota. However, this signal is driven largely by reviews and synthesis studies, whereas primary laboratory and field data do not consistently corroborate these concerns and instead show more balanced or neutral outcomes. With a few exceptions, available data from field studies do not show a significant accumulation of metals in soils,[444,475] likely reflecting high background soil concentrations, low application rates, and/or the use of low-metal feedstocks. Many soil core and mesocosm studies do find significant accumulation of metals in soils, but this rarely translates into proportional increases in bioavailable or toxic levels.[98,171,222,223,404,464] In fact, several studies report no significant effect or even a decrease in bioavailability as metals are immobilised due to a pH increase and/or secondary mineral formation.[83,453,613] This effect is particularly well established for exchangeable Al in acidic soils, where the liming effect of feedstock dissolution raises pH and reduces Al availability and toxicity.[47,388,513,614,615]

This pH effect on the bioavailability of most metals can also explain the lack of significant metal accumulation observed in biomass.[108,386,404,444,453,458,616] Other studies do report elevated biomass metal contents linked to silicate rock additions, but these generally do not surpass regulatory guidelines.[171,443,617] To mitigate impacts from high-metal feedstocks, phytoremediation using hyperaccumulating plants has been proposed.[150,179] Increased porewater or effluent metal concentrations can occur following feedstock applications, particularly for Ni and Cr after olivine additions, with concentration peaks sometimes occurring shortly after application.[179,184,222] Reported Ni concentrations occasionally approach or exceed regulated groundwater limits for experiments using olivine.[98,223]

A key limitation of most experimental studies, however, is their relatively short duration and reliance on single feedstock applications. Projections suggest that repeated annual rock powder applications can lead to substantial accumulation of metals over time, potentially exceeding environmental thresholds even for relatively low-metal feedstocks like basalt.[129,617] Although elevated soil metal contents may not pose risks due to immobilisation, discontinuation of liming or EW practices could lower pH and remobilise previously accumulated metals (e.g., ref. [69]).

Additionally, at the scales required for meaningful global CDR, EW is also likely to introduce additional, more subtle risks. Beyond direct toxicity, metal inputs may alter micronutrient availability and ecosystem functioning, potentially affecting, for example, soil microbial communities (cf., Section 7.2.1) or downstream aquatic systems (cf., Section 7.2.2). Multiple sources therefore emphasize that repeated applications, high application rates, and high feedstock metal contents, could push metals into more biologically relevant pools and ultimately food chains,[171] urging for monitoring of metals in field deployments (e.g., refs. [131,383]) and caution, even where short-term toxicity is not observed.[618]

Overall, until clear data-informed and EW-specific guidelines and regulations are in place, a pragmatic deployment strategy is to couple site- and feedstock-specific risk assessments with conservative initial application rates and systematic monitoring of soil, aqueous, and plant tissue metal concentrations. These measures align with the broader mitigation and safeguard framework outlined in Section 3.5.2, including feedstock screening, baseline characterization, predefined environmental thresholds, adaptive management, and contingency planning where thresholds are exceeded.

7.3 Agronomic effects

Agronomic co-benefits are among the main reasons why EW has attracted interest as a CDR strategy in managed landscapes. By supplying base cations and other nutrients, increasing soil pH, and altering soil chemical and physical properties, EW feedstocks can influence crop performance, nutrient cycling, and overall soil functioning.

The semi-quantitative assessment of agronomic side effects conducted here (Figure 28) indicates a strongly positive overall impact of EW across multiple domains, including system resilience, crop productivity (c.f., Section 7.3.1), and soil quality. This signal is particularly pronounced in experimental studies, where positive outcomes consistently outweigh neutral and negative responses, suggesting that these co-benefits are not merely conceptual but are supported by empirical evidence. For soil quality specifically (incorporating parameters discussed in more detail in Sections 7.3.3–7.3.5), the compiled dataset shows a clear dominance of positive effects. Across EW studies, 48 out of 81 report exclusively positive outcomes, compared to only 10 exclusively negative and 23 neutral cases. This pattern is also evident within the experimental literature: in field studies, 6 out of 8 report exclusively positive effects with no exclusively negative outcomes; in laboratory studies, 9 out of 20 are exclusively positive compared to 3 exclusively negative; and in model studies, 13 out of 20 are exclusively positive compared to 3 exclusively negative. A similar, and in some cases stronger, signal is observed in the conventional fertilisation literature, where 19 out of 20 field studies and 14 out of 20 laboratory studies report exclusively positive effects, with very few exclusively negative outcomes. For agronomic impacts more directly (Figure 28), the positive signal is even stronger, as discussed below.

Taken together, these results indicate that improvements in soil chemical properties—such as pH buffering, replenishment of weathering-derived nutrients, and enhanced cation availability—are consistently observed across study types, while adverse effects remain comparatively rare and context-dependent. Overall, there is strong empirical evidence that EW can improve soil quality and deliver tangible agronomic co-benefits alongside CDR. At the same time, the magnitude and direction of these responses remain strongly dependent on soil conditions, climate, and crop type, as well as on feedstock properties and application strategies. The following sections therefore assess the current evidence for the agronomic effects of EW in more detail, including impacts on crop yields, NUE, soil pH, and broader soil chemical and physical properties.

7.3.1 Impacts on crop yields

The idea to boost agricultural productivity with ground rocks dates back to the 19th century [40], and was studied extensively throughout the 20th century with a focus on fertilizing intensively

weathered or depleted soils, especially in the tropics.[619,620] Van Straaten's foundational reviews comprehensively addresses research on rock fertilisation in agriculture prior to the integration of EW.[620,621] Consistent with this historical body of work, the semi-quantitative assessment conducted here (Figure 28) shows strong overall support for agronomic benefits associated with EW, with 129 positive impacts reported compared to 11 negative and 26 neutral outcomes. Importantly, this pattern is also evident within the experimental literature: across field and laboratory studies, positive-only responses dominate (e.g., EW: 13/13 field and 11/17; rock flour as fertiliser: 25/31 field and 32/41 laboratory studies), while exclusively negative findings remain rare. Together, this indicates that agronomic co-benefits of EW are not merely conceptual but are consistently supported by empirical evidence across study types and conditions. This is supported by recent modelling studies that have proposed that EW feedstocks may function as a form of rock fertilisation, increasing global crop yields by supplying essential macro (P, K, Ca, Mg) and micronutrients.[22,64,106] Pot, greenhouse and field trials have further evidenced these potentials and the pitfalls of supplying rocks to crops since the latest agronomic review.[47]

The agricultural effects of EW are influenced by the target crop, background soil composition and nutrient status, as well as by rock type, particle size, application rate, climate, and potential co-applications. In the data compile here, treatments targeting total biomass or crop yield most commonly show modest positive responses, but responses vary substantially across studies and do not scale simply with application rate or feedstock type (Figure 29). This should not be interpreted as evidence that dose effects are absent where application rate is varied while other factors are held constant. Rather, within this dataset, many higher-application-rate studies were conducted on agricultural soils that may already have undergone pH correction or nutrient management, potentially limiting additional yield responses. Conversely, where background soil composition or nutrient status is limiting, rock powders can strongly increase total biomass or crop yields even at relatively low application amounts. Many feedstock types remain understudied, particularly across gradients in application rate, soil type, crop type, and background nutrient status.

Beyond the semi-quantitative synthesis in Figure 28 and some of the response-ratio data shown in Figure 29, individual studies illustrate that rock powders can substantially increase crop yields or total biomass where they alleviate specific soil constraints. For maize (*Zea mays*), dacite additions in mesocosm studies produced 1.4-fold higher yields relative to controls,[622] while gneiss powder improved growth in greenhouse trials.[600] Field trials report yield increases ranging from 1.1- to 1.7-fold.[388,444,556,623] In wheat (*Triticum* sp.) greenhouse studies, carbonatite increased shoot and root biomass 1.4-fold compared to lime,[624] while finely milled feldspars and other silicates can increase potassium uptake and biomass.[169,625] However, responses are not universally positive (Figure 29), producing no yield increase or even slight decreases compared to untreated controls.[613,626] Outside agriculture, positive long-term woody biomass increments have also been reported in forestry, with yield response ratios of 1.2 to 1.7.[70,514,627,628]

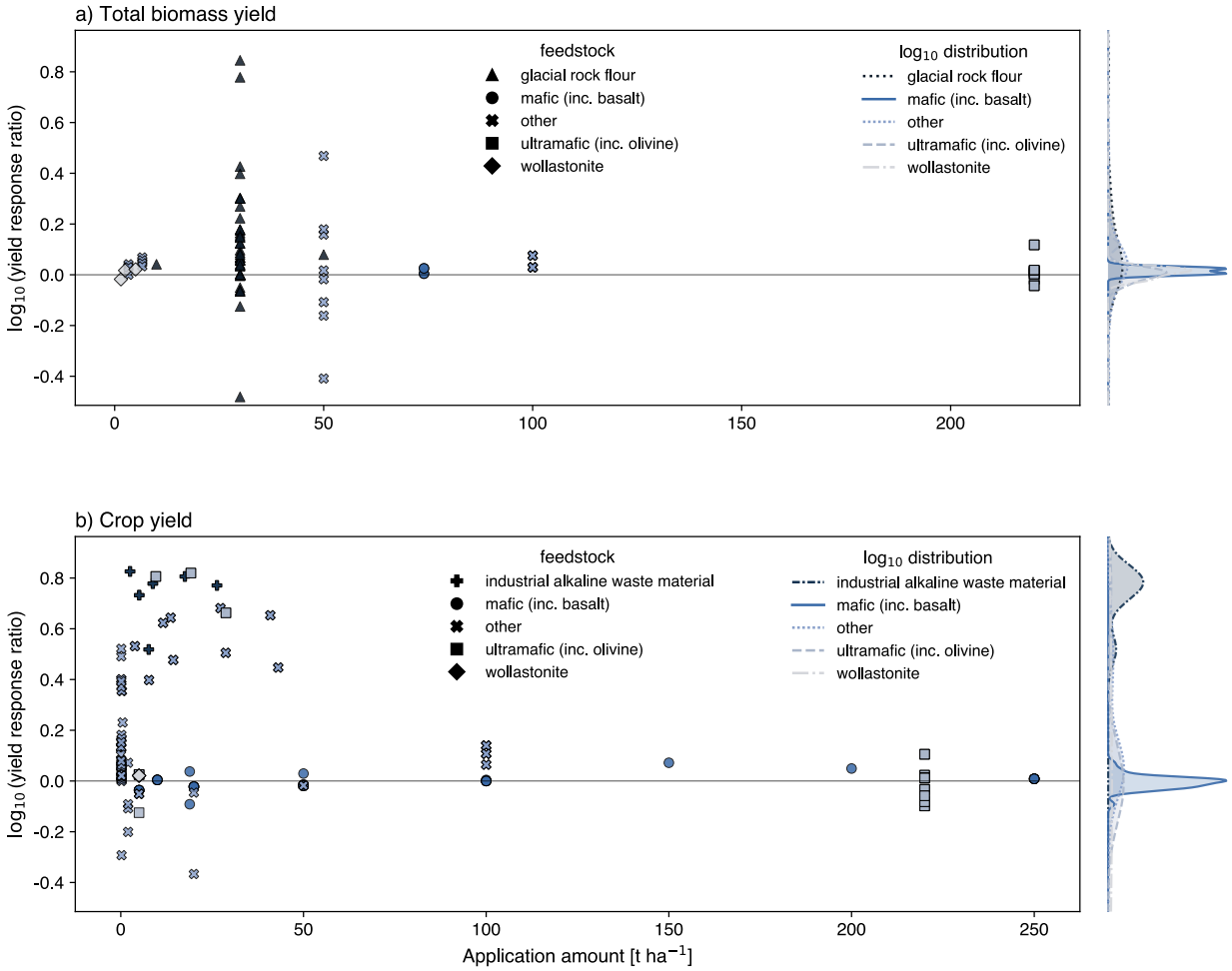

Figure 29: Total biomass and crop yield Enhanced Weathering (EW) responses. (a) Total biomass yield and (b) crop yield, both across studies and plotted as log_{10} response ratios as a function of EW feedstock application amount (t ha^{-1}). Feedstocks (glacial rock flour, mafic (inc. basalt), other, ultramafic (inc. olivine), and wollastonite are differentiated by shape. Log_{10} distributions are shown at right for both panels, with feedstock indicated by colour and line type.

Overall, the available evidence indicates that EW can provide clear agronomic benefits, particularly where rock powder additions alleviate existing soil constraints such as low pH or nutrient limitation. In some settings, EW could partially substitute for liming and mineral fertilisation while also supporting crop yields and biomass production due to nutrient release as well as improvements of NUE (see below). However, these benefits depend critically on whether the minerals present dissolve at agronomically relevant rates. To address this, candidate EW feedstocks should be routinely characterised by XRD and XRF and assessed for short-term acid-neutralising capacity and release of agronomically relevant nutrients, particularly Ca, Mg, K, and P.[132,629] Such characterisation would improve prediction of dose-response relationships with soil nutrient status[630] and crop productivity,[631] and could ultimately support the integration of EW amendments into national crop fertilisation advisory services.

7.3.2 Impacts on nitrogen use efficiency

Nitrogen availability, nitrogen retention, and NUE are all key variables in agriculture, as they have direct impacts on plant growth and yield, farming costs, and the downstream ecosystem. Increasing global NUE rates to their maximum potential has been suggested to be an important step in the path towards achieving global food security.[632] Crop yields generally increase with increased N inputs and optimal N application rates have been increasing over time, even when taking fertiliser and environmental costs into consideration.[633] Nitrogen losses are directly tied to N inputs, and can be lost in the form of gases, which contribute to GHG emissions or decreased air quality;[305] or lost as leached N into waterways, causing drinking water contamination and eutrophication.[634] It is therefore important to pursue other solutions to directly and indirectly mitigate N losses such as by increasing N retention and increasing NUE. While it is unlikely that EW would supply N to soil directly, as silicate rocks contain N in very low concentrations,[15,635] changes to soil chemistry or structure could have indirect effects on N cycling, availability, and NUE.

Only two studies appeared to measure NUE's response to EW specifically and found that NUE was significantly higher in soils treated with basalt compared to control soils.[353,444] Additionally, Conceicao et al., (2022)[623] found that EW treated soils resulted in higher accumulation of N in shoots of maize. Enhanced Weathering also has been shown to alter root gene expression to facilitate increased transport of soil N into the plants.[444] Another indicator that EW may increase NUE is its effects on plant growth; in 90% of instances where yield/biomass was measured in response to EW, EW significantly increased plant growth.

A few studies measured soil N content: one found that soil N increased in response to EW with a potassium feldspar feedstock,[636] whereas another study found that ammonium-nitrate decreased in EW treated mesocosms in the short term.[83] EW has been found to increase soil pH (see also next section), which may increase adsorption and subsequent N retention in soils, reducing losses via leaching and causing N to cycle more tightly. Only one study reported leaching losses but found that EW significantly reduced leaching losses of applied NH_4.[637] Similarly, EW was found to increase water use efficiency, which can also contribute to reductions in N leaching.[69,617,638]

Outside literature provides more context on nitrogen cycling with respect to soil pH. Nitrogen mineralisation is positively correlated with soil pH,[639] and soil pH has been shown to be positively correlated with rates of nitrification.[640,641] Higher soil pH generally results in higher rates of, and more complete, denitrification; however, the existence of a pH optimum is contentious.[642]

Overall, N was infrequently measured in the EW literature and warrants further attention in future studies. Nevertheless, the limited available evidence suggests that improvements in NUE may be one of the mechanisms through which EW could contribute to yield gains, alongside the more direct release of nutrients such as Ca, Mg, K, and P from dissolving feedstocks. Liming provides a useful analogue because, like EW, it increases soil pH and has been found to increase yields, SOC stocks,[42,43] and NUE in some studies.[643,644] These patterns suggest that EW-induced changes in soil pH, nutrient retention, and microbial N cycling could influence crop productivity, but this pathway remains much less directly tested than nutrient release or yield responses themselves. Future EW studies should therefore measure N pools, N losses, plant N uptake, and NUE alongside crop yield to determine whether improved N cycling is a consistent agronomic co-benefit of EW.

7.3.3 Impacts on soil pH

The potential for EW to replace liming as a method of soil pH management has been widely credited as a co-benefit of EW (e.g., refs. [16,106,645]). These arguments are supported by decades of agronomically-focused laboratory, greenhouse, and field trials that have demonstrated increases in soil pH following rock powder applications (e.g., refs. [625,637,646,647]). However, the exact effects on soil pH are wide-ranging across experimental treatments (Figure 30a), generally ranging from no significant effect (e.g., refs. [231,514,648]) to increases up to several pH units (e.g., refs. [112,114,179]). Soil pH responses are often variable across soil types and show a non-linear response to mineral application rates, with <50 t ha^{-1} applications often sufficient to achieve increases in soil pH (e.g., refs. [103,114]).

This variable soil pH response to EW appears strongly related to initial pH conditions (Figure 30b). In particular, EW and other rock powder applications appear most capable of neutralizing acid soils, with many studies reporting increases from acidic (pH <4-5) to neutral (pH ~7) soil pH following application (e.g., refs. [103,114,234,384,647]). As pH rises, dissolution rates for most minerals (cf., Section 3.2.2), including major silicates in basalt, decelerate and thus the soil pH response is likely to be smaller at higher initial pH values. Alongside direct pH controls on reaction rates, soil buffering capacity can influence the magnitude of soil pH response to EW. This buffering is driven by processes including ion exchange, which substitutes cations released during EW (e.g., Ca^{2+}) for exchangeable acidity, the pH dependency of these exchange sites on soil materials like iron oxyhydroxides and organic matter, and aluminium buffering in low-pH soils (e.g., refs. [68,368,647,649]).

Synergies and challenges reconciling the agronomic and geochemical aspects of EW also occur in the modelling of soil pH. Many models exist that robustly simulate soil properties such as pH in agricultural environments, but further development is needed to simulate the geochemical processes driving mineral dissolution and CO_2 sequestration in these models.[157,403] Conversely, RTMs that simulate chemical affinity and mineral dissolution and precipitation kinetics under semi-realistic to realistic hydrologic conditions typically do not simulate soil pH directly, although robust soil pH simulations in RTMs are emerging.[232]

From a global perspective, acid soils are estimated to comprise ~30% of the world's current land area, in some cases constraining agricultural production from arable land.[650] Modelling in the U.S. "corn belt" suggests that EW could reduce the fraction of acidic soils (pH < 6.5) from the current 70% to between 40% and 5% by 2070.[22] Silicate applications also show promise beyond agricultural settings. For example, the application of silicate minerals to forest soils acidified by "acid rain" deriving from industrial pollution during the 20th century has shown the capability of restoring ecological health by increasing base cation saturation and reducing soil acidity.[70,368,369,514,651] The widespread deployment of EW needed to achieve sufficient CDR could thus have significant benefits for soil pH across ecosystems. Overall, soil pH correction is likely one of the clearest agronomic pathways through which EW can improve crop performance, particularly in acidic or poorly buffered soils, but the magnitude and persistence of this benefit depend on feedstock reactivity, application rate, soil buffering capacity, and initial pH.

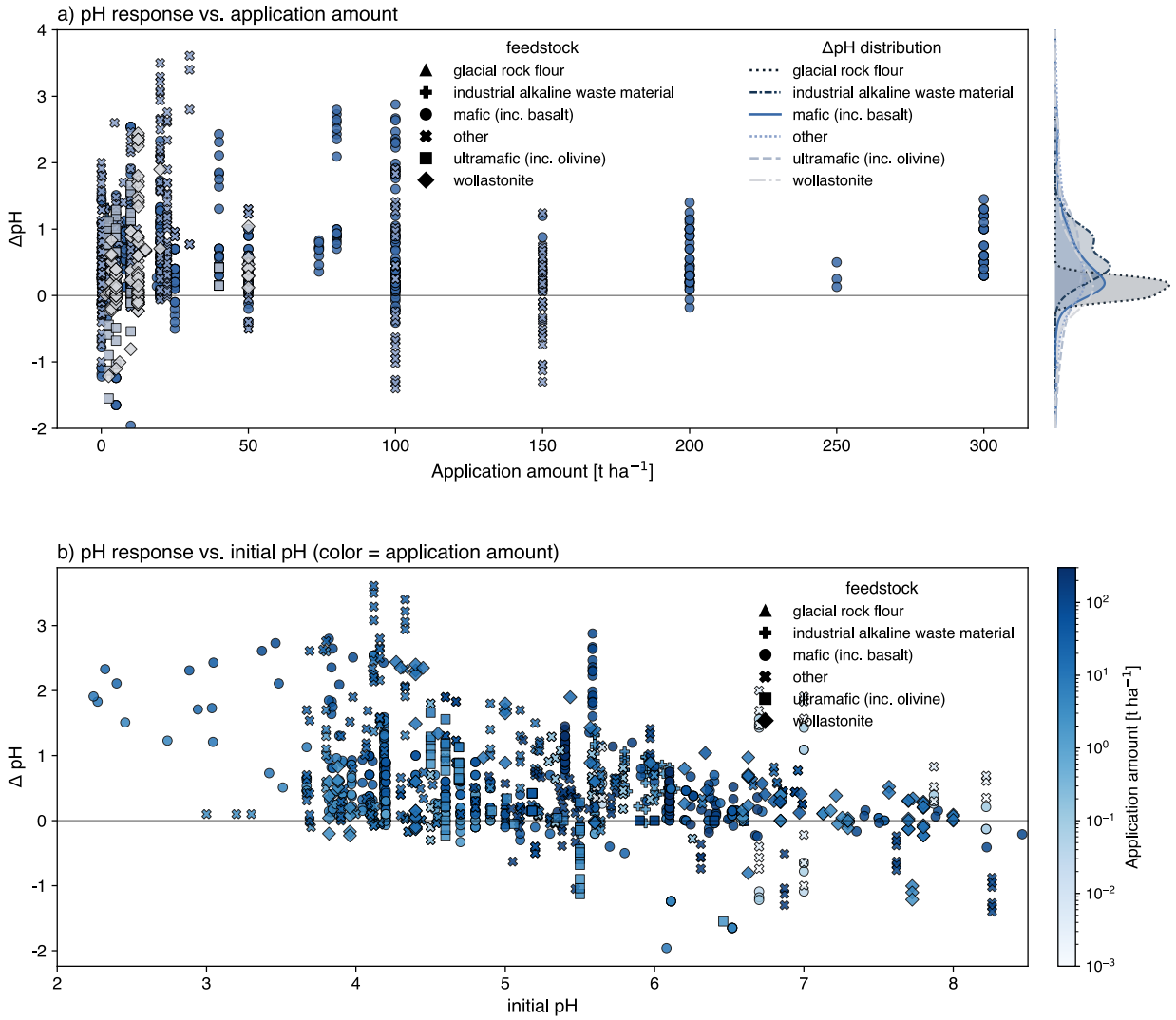

Figure 30: Change in soil pH due to Enhanced Weathering (EW) interventions as a function of the application amount (a) and as a function of the initial soil pH (b).

7.3.4 Impacts on cation sorption in soils

Base saturation and CEC are central soil fertility metrics because they influence nutrient availability, acidity buffering, and the retention of agronomically important cations such as Ca^{2+}, Mg^{2+}, K^{+}, and Na^{+}. In the context of EW, these same soil chemical properties are important not only as agronomic co-benefits, but also because they influence the fate of weathering-derived cations. Together, BS and CEC largely determine the extent to which soils can retain, at least temporarily, the cations released during mineral dissolution[68]—with associated CDR losses (cf., Section 3.4.2). Soils with greater exchange capacity and available exchange sites (low BS) may retain more Ca, Mg, K, and Na in exchangeable pools, delaying their export in solution, whereas soils with limited retention capacity and/or high BS may allow more rapid leaching and greater downstream losses. Because of this, changes in BS and CEC are central for understanding both

agronomic responses to EW and the timing of realised CDR, including the lag times discussed in Section 3.4.6.

7.3.4.1 Base cation saturation

Base saturation is a measure of the percentage of a soil's CEC occupied by "base" cations (typically K^+, Na^+, Ca^{2+}, Mg^{2+}) as opposed to exchangeable acidity (H^+, Al^{3+}).[215] Base saturation is a commonly used metric in evaluating soil health because soil biota utilize exchangeable base cations as nutrients whereas soluble aluminium is harmful to many plants, and higher BS provides greater buffering capacity to maintain soil pH against acidity inputs.[215] Through the release of major cations such as Ca^{2+} and Mg^{2+}, rock powder applications including EW can increase BS of soil (e.g., refs. [91,164,369,514,646,651]). Our complied data shows the magnitude

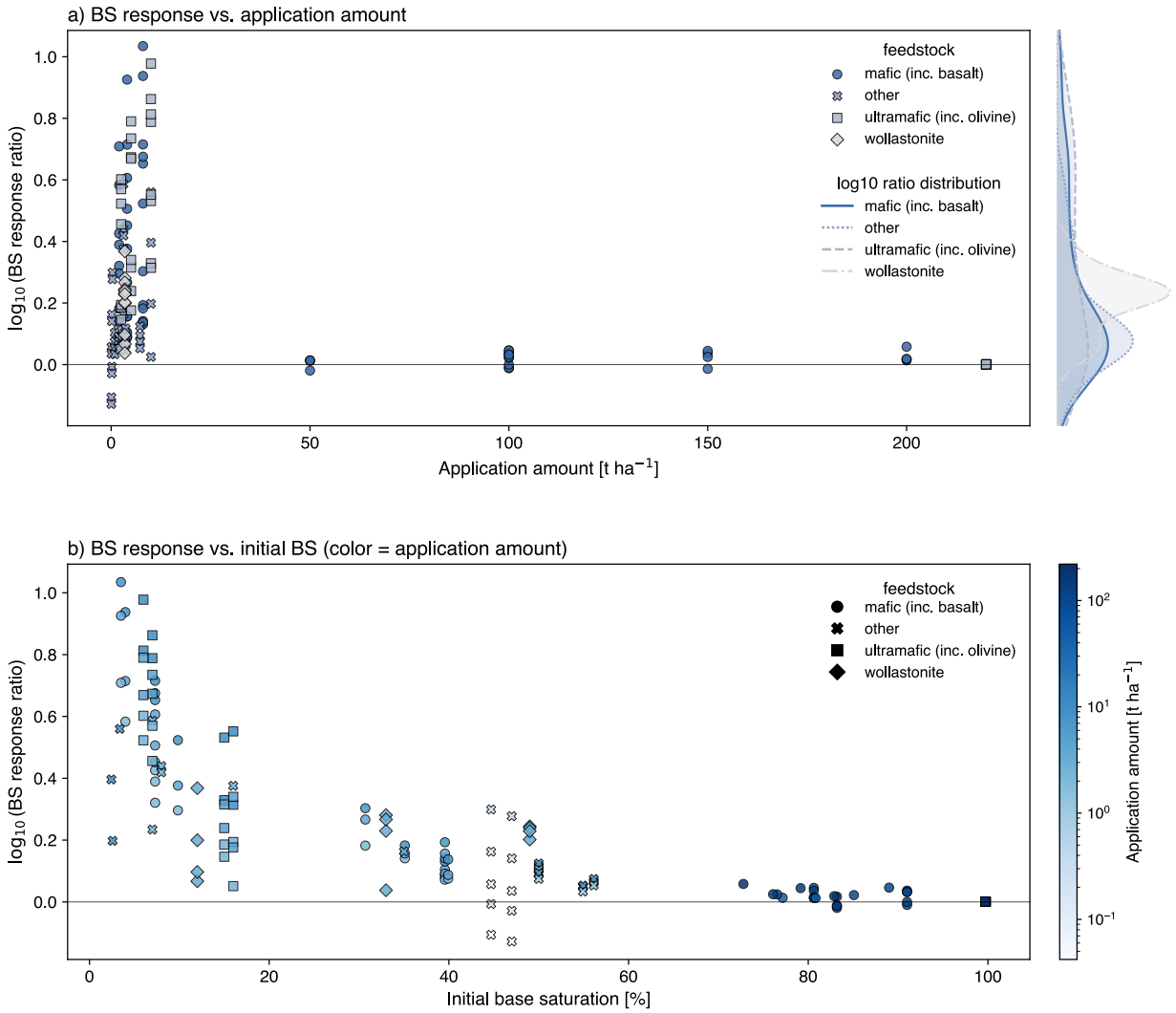

Figure 31: Change in base saturation (BS), expressed as the log ratio of final versus initial BS percentages, as a function of a) application amount (tons ha^{-1}) and b) initial BS (%).

of response in BS to mineral applications is strongly dependent on the initial conditions of the soil, with the large responses observed in typically acidic soils with low initial BS and minimal responses in soils with high initial BS (Figure 31). This effect is intuitive because soils with a high initial BS will have very little exchangeable acidity (H^+, Al^{3+}) to substitute for the cations released by mineral dissolution, and thus BS is not likely to substantially change.

The application rate response (Figure 31a) should be interpreted cautiously. In the compiled dataset, many studies with application rates >50 t ha^{-1} were conducted on soils with already high initial BS (>70%), where large additional increases are not possible (as 100% is the maximum fraction of exchangeable sites that can be occupied by cations). The apparent pattern in Figure 31a therefore likely reflects, at least in part, the distribution of initial soil conditions across the compiled studies. Nevertheless, BS responses are unlikely to scale linearly with application rate alone, because they also depend on initial BS, CEC, soil buffering capacity, and feedstock dissolution.

7.3.4.2 Cation exchange capacity

Cation exchange capacity is a measure of the total amount of negatively charged sites in a soil that can reversibly adsorb and exchange cations (e.g., K^+, Na^+, Ca^{2+}, Mg^{2+}).[215] It is a key control on soil fertility and geochemical buffering because it governs the retention, mobility, and bioavailability of nutrient and weathering-derived cations, thereby influencing both plant uptake and leaching losses.[215]

Mineral applications have often been observed to increase CEC of soil.[164,646,647] Across the compiled data, the exact change in CEC tends to be highly variable, with occasional decreases observed, and displays a slight tendency to increase with increasing application rates up to ~100 t ha^{-1} (Figure 32a). Relative changes in CEC are typically greatest when the initial CEC of the treated soil is low (Figure 32b), mirroring the trends observed for BS.

These increases may be driven by increases in soil pH impacting the charge distribution of variable charged soils, the formation of secondary phases such as clay minerals, or greater retention of organic matter, as clay minerals and organic matter both provide major sources of exchange sites in soils.[215,369,637] Conversely, the occasional decreases in CEC could reflect the loss of SOM observed in some soil experimental treatments (cf., Section 7.4.2). Alternatively, if not accounted for, decreases in bulk soil CEC may also arise from the addition of low-CEC mineral feedstocks, which can dilute the exchange capacity of native soils prior to, or independent of, subsequent weathering reactions. In applications where CEC increases, a greater availability of negatively-charged sites can promote greater retention of weathering-derived cations, which benefits soil fertility but may increase the time lag between mineral applications and realized carbon sequestration (cf., Section 3.4.6).

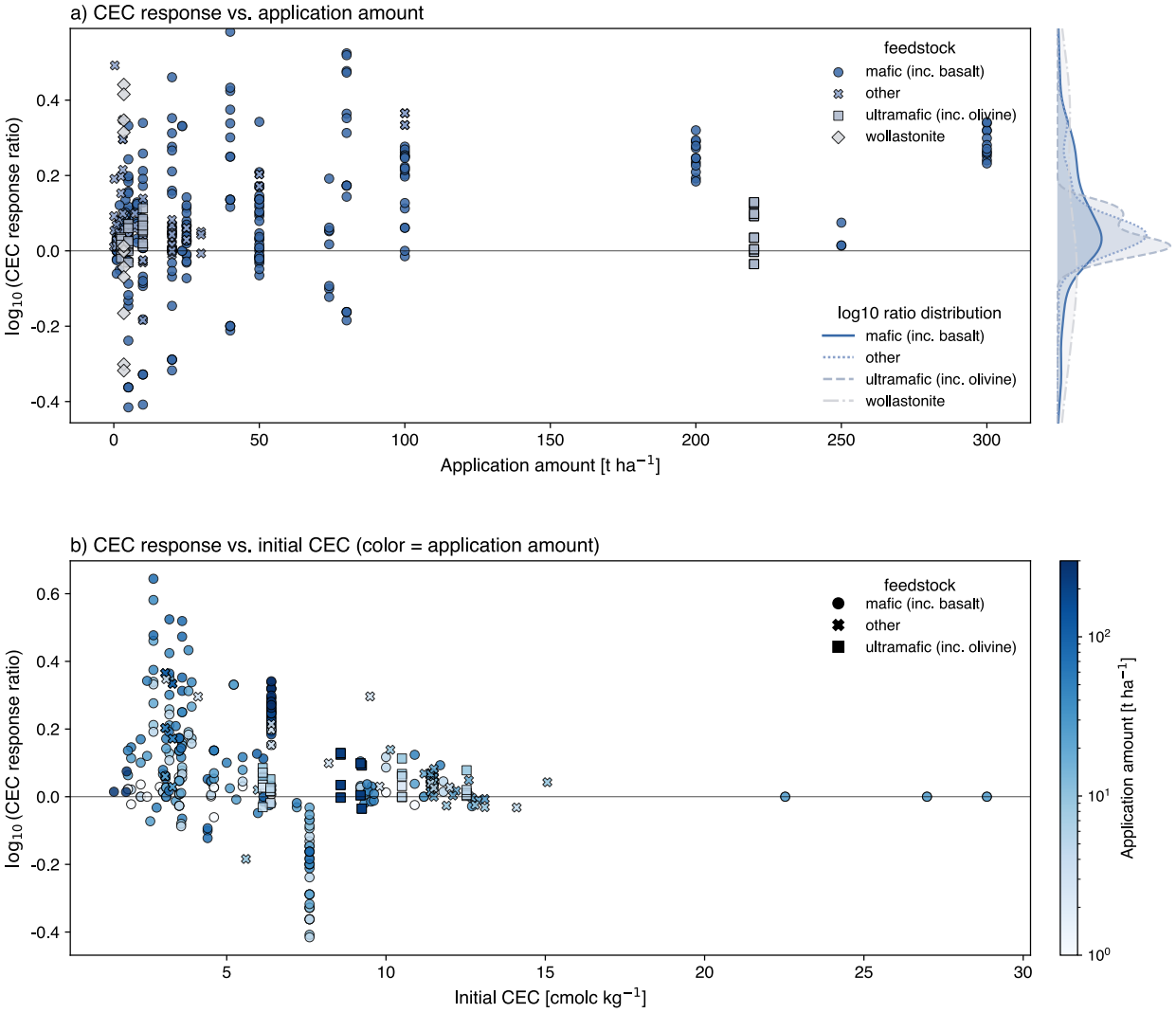

Figure 32: Change in cation exchange capacity (CEC, cmolc kg^{-1}), expressed as the log ratio of final versus initial CEC values, as a function of a) application amount (tons ha^{-1}) and b) initial CEC.

7.3.5 Impacts on soil physical properties

Chemical processes during silicate mineral weathering, including the formation of secondary clay or carbonate minerals, together with the physical addition and subsequent dissolution of finely ground minerals, can alter soil physical properties. These changes are often seen as beneficial, with several studies investigating mineral addition as a means of soil stabilisation, improved water retention, or altered soil hydraulic conductivity.[168,519,652,653] However, altered soil hydrology from processes including the introduction of large volumes of mineral or subsequent precipitation of swelling clay minerals could act as a bottleneck on weathering rates and the movement of EW products in some cases.[64,69]

Soil stabilisation from EW and similar rock powder additions can be a promising co-benefit, particularly in poorly formed or coarse-grained soils where the formation of soil aggregates and other stabilising structures can promote the retention of organic matter, reduce vulnerability to

erosion, and enable the rehabilitation of functional soils in impacted environments.[96,653,654] EW and other mineral additions may be beneficial to soil aggregate formation in agricultural or manufactured soils, driven by interactions between the primary and secondary minerals involved in EW and SOM.[96,653] The formation of carbonate minerals as the cations released from mineral dissolution react with DIC can also produce cementing effects and promote stabilisation as precipitated carbonates fill pore space and bind soil particles together.[652] In addition to impacts on aggregate stability, if rock additions are coarser or finer textured than native soil, they may alter the distribution of pore sizes or total porosity.[519]

The physical effects of EW can be beneficial or harmful to the retention of water in soils, with these diverging effects often driven by baseline soil texture (e.g., sandy vs. clay-rich soils) and potentially impactful on the rates of weathering. Soil water retention is shaped in large part by the hydraulic conductivity of the soil, with coarser grained soils often possessing higher hydraulic conductivity and thus shorter fluid residence times compared to fine-grained soils. Due to their high hydraulic conductivity, sandy and other course-grained soils typically drain water rapidly to the detriment of plant growth and potentially the time available for water-rock interaction during EW, and the formation of secondary mineral phases during EW can reduce hydraulic conductivity and improve water retention.[63,168] Conversely, further decreases in the permeability and hydraulic conductivity of fine-grained soils (e.g., clay or silt-rich soil) during EW can impede drainage, and the fine-grained rock powder applied during EW could itself throttle the hydraulic conductivity of a soil.[63,64,69,168] These physical changes are relevant to the geochemical processes driving weathering, as the lower hydraulic conductivity and longer fluid residence times of low permeability soils can reduce the influx of reactants, decelerate rates as weathering reactions approach equilibrium, or promote the formation of secondary minerals.[62] However, the impact of EW on soil water retention has been considered by relatively few studies, and evidence for the alteration of soil water retention is mixed.[173] Overall, soil physical responses to EW may represent either agronomic co-benefits or constraints on weathering efficiency, depending strongly on initial soil texture, hydrology, feedstock particle size, and the extent of secondary mineral formation.

7.4 Climate-relevant side effects

Beyond its agronomic and environmental co-benefits, EW directly interacts with multiple components of the terrestrial carbon cycle and broader climate system. Carbon Dioxide Removal is typically framed in terms of alkalinity generation and bicarbonate export, but a range of additional processes—including SIC formation, changes in SOC dynamics, and alterations in non-CO_2 GHG fluxes—can modify both the magnitude and permanence of net CDR. These processes operate across different timescales and compartments and may either reinforce or partially offset the intended CDR effect. The following sections assess how EW influences key carbon pools and GHG fluxes, and the implications for accurately quantifying its climate mitigation potential.

7.4.1 Impacts on soil inorganic carbon

Weathering releases base cations and alkalinity and increases soil porewater pH, thereby raising the saturation state of secondary minerals, including carbonates. As a result, SIC, primarily present as calcite ($CaCO_3$) and other carbonate minerals in soil horizons, can increase during EW where aqueous cations such as Ca^{2+} and Mg^{2+} combine with DIC. The precipitation of paedogenic

carbonate can provide a relatively durable solid phase carbon sink (cf., Section 3.3.3). However, carbonate precipitation also releases one mole of CO_2 per mole of carbonate formed, making it less efficient than storage as dissolved alkalinity and complicating CDR accounting frameworks that assume weathering-derived carbon remains in solution as bicarbonate.

The semi-quantitative synthesis of the literature (Figure 28) further supports this conceptual expectation, indicating that increases in SIC stocks (though sometimes also interpreted to include export of dissolved C) are reported more frequently than decreases, but are far from ubiquitous. Across more than 150 studies assessing SIC responses (incl. aqueous processes), positive outcomes dominate (≈70%), with a smaller fraction reporting negative (≈8–10%) or mixed responses (≈20–25%). This pattern is broadly consistent across both EW-focused studies and work on rock amendments applied as fertilisers. Importantly, these findings complement the dominant role of alkalinity export as the primary and more consistently observed inorganic carbon storage pathway in EW systems, with paedogenic carbonate formation representing a secondary, more variable sink that depends on local geochemical and hydrological conditions.

Mirroring this semi-quantitative assessment, experimental data on SIC concentrations range from clear decreases[108,352,444] to substantial increases.[178,246,384,453,485,655] Increases are reported more frequently than decreases, and their magnitude is generally larger, as reflected by overall larger positive than negative log response ratios (Figure 33). Nevertheless, SIC formation is not a universal response to EW. Where it does occur, its formation can represent a major carbon flux and may dominate over dissolved alkalinity export, making accurate quantification of SIC changes essential for CDR accounting. Soil inorganic carbon formation should primarily be expected where soil porewaters are driven toward carbonate supersaturation (cf., Section 3.3.3). This depends on initial soil chemistry, pH, DIC availability, hydrology, and feedstock stoichiometry, particularly the supply of Ca, rather than on application amount alone. For example, Ca-rich feedstocks such as wollastonite may be especially conducive to SIC formation where other conditions favour carbonate supersaturation.[103,112,170,178,246,384,451] In such settings, SIC accumulation may provide a useful MRV target for quantifying solid phase carbon storage (cf., Section 4.1.2). At the same time, this spatial specificity means that SIC-based MRV is likely applicable only in some deployment contexts, rather than as a universal approach across EW systems. This is an important limitation for MRV, but it also implies that carbonate precipitation losses should not be assumed to occur uniformly across all EW deployments.

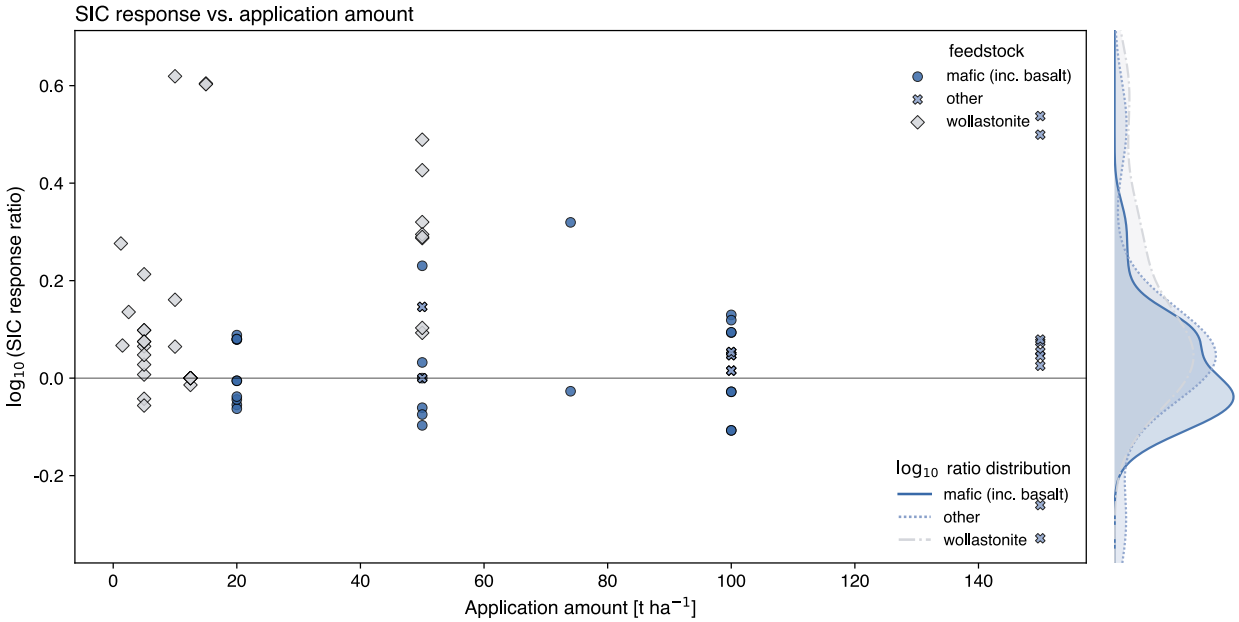

Figure 33: Soil inorganic carbon (SIC) responses on a log_{10}-scale, as a function of application rate (tons ha^{-1}) for different feedstocks (mafic (incl. basalt), wollastonite, or other, differentiated by shape and colour). The log_{10} ratio distributions are shown at right, with feedstock indicated by colour and line type.

Changes in SIC are likely temporally variable (cf., Section 3.3.3.1; Figure 14), with several studies suggesting that timescales longer than typical experiments (e.g., >365 days) may be necessary to observe SIC formation following EW deployment,[108,243] although some long-term studies still do not observe measurable SIC formation (e.g., ref. [444]). Changes in SIC also appear related to the depth of sampling, with SIC formation particularly relevant in deeper soil layers.[103] Most studies show a minimal influence of biota on SIC formation,[91,439] potentially reflecting that precipitation rates are primarily limited by cation release from mineral dissolution rather than the relatively abundant DIC produced by soil biota.[184] However, the presence of plants accelerated SIC formation in other studies.[656]

Paedogenic carbonate formation and thus increases in SIC is likely in part reflective of climate and particularly the water balance (e.g., precipitation inputs minus outputs via evaporation and transpiration) of the environment. Multi-site comparisons have found SIC accumulation is greater in more arid environments where concentrating effects of evapotranspiration and slow leaching of weathering products may increase the potential for carbonate precipitation.[95,234] These observations mirror overall trends in SIC in soils globally, with greater SIC typically observed in the soils of more arid environments.[286] A hydrologic influence on SIC formation is also particularly important for laboratory or greenhouse experiments emulating EW in field soils, as a failure to robustly simulate the leaching or accumulation of weathering products may lead to over- or underestimates of SIC formation compared to natural environments, respectively.

7.4.2 Impacts on soil organic carbon

Enhanced Weathering may affect SOC dynamics in multiple ways, through coupled physicochemical and biological processes that operate on different timescales. Given that SOC stocks and fluxes often exceed the inorganic carbon flux generated through EW, even modest SOC responses could materially affect net climate outcomes.[94]

The semi-quantitative synthesis of the available literature supports this picture of variability (Figure 28), while suggesting a slight predominance of positive SOC responses overall. Across field and laboratory studies, positive responses are somewhat more frequent (≈55–65%) than negative ones (≈20–30%), with a substantial fraction of studies (≈10–25%) reporting mixed or neutral outcomes. In contrast, model-based studies and review syntheses more consistently report net SOC gains or expectations thereof (≈70–80% positive outcomes), reflecting an expectation that longer-term increases in plant productivity and MAOM formation outweigh initial losses. A similar, albeit more limited, pattern is observed in studies of rock amendments applied as fertilisers. These divergent outcomes reflect the interplay of competing processes that operate on different timescales and respond differently to environmental and management conditions.

This qualitative pattern is broadly consistent with the formal meta-analysis of Xu et al. (2025)[93], who analysed 74 experimental studies of crushed rock amendments, including both silicate feedstocks and carbonate amendments such as lime. Their synthesis reported modest but significant increases in SOC and in both mineral-associated and particulate organic carbon pools, with mean effects of 3.8%, 6.1%, and 7.5%, respectively, whereas dissolved organic carbon and SIC did not show significant overall responses. Xu et al. (2025)[93] further found that SOC responses were associated with changes in exchangeable Ca, microbial biomass, soil structure, and climate, and that effects varied with application amount, experimental duration, and region. In their global cropland simulations, positive SOC responses were concentrated mainly in warm and humid low-latitude regions, while weaker or negative responses were projected for some colder or drier high-latitude regions. These results indicate that mineral amendments can enhance SOC accumulation in some settings, but that effects are highly context-dependent and should not be assumed to occur uniformly across feedstocks, soils, climates, application rates, or timescales.

Conceptually, EW can be expected to (initially) stimulate SOM decomposition through increases in soil pH, nutrient availability, and microbial activity. This may lead to increases in soil CO_2 efflux following silicate application (e.g., refs. [112,114]), but this is not always observed (e.g., refs. [91,224]). Moreover, the increase in SOM decomposition may only be short-lived. Indeed, several experiments have shown that EW can alter SOM turnover and stabilisation pathways over time, potentially leading to reduced decomposition or enhanced stabilisation rather than sustained SOC losses.[92,101,439] Others have found complex interactions between EW and organic amendments, including increased total soil C with EW, but less SOM when EW was co-applied with organic amendments.[528] The variability in SOM and SOC responses to EW application is apparent across feedstock types and application rates, with both positive and negative responses commonly observed (Figure 34).

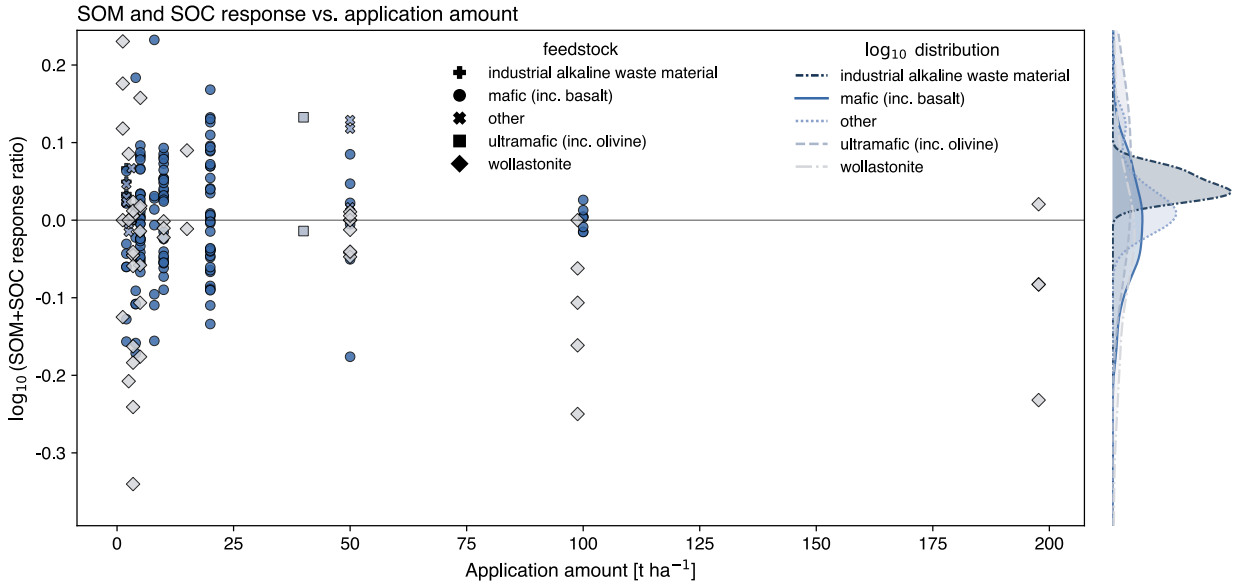

Figure 34: Soil organic matter (SOM) and soil organic carbon (SOC) combined response to Enhanced Weathering (EW). Log_{10} combined response ratios of SOM and SOC are shown plotted vs. application amount (t ha-1). Feedstocks (glacial rock flour, mafic (inc. basalt), other, ultramafic (inc. olivine), and wollastonite are differentiated by shape. Log_{10} distributions are shown at right for both panels, with feedstock indicated by colour and line type.

Over longer timescales, EW may enhance SOC stabilisation via MAOM formation and aggregate-mediated physical protection. Silicate dissolution releases divalent cations (Ca^{2+}, Mg^{2+}) that promote cation bridging, clay flocculation, and aggregate formation.[527] Mineral weathering also generates secondary Fe- and Al-(hydr)oxides with high sorptive capacity, enabling strong mineral–organic associations.[526,657] Recent EW studies support these mechanisms, reporting increased macroaggregate-associated SOC,[658] incorporation of weathering-derived cations into oxide- and organic-bound fractions,[439] and enhanced SOC storage in Fe-/Al-rich secondary mineral pools.[92] These findings suggest that EW has the potential to shift SOC from labile pools toward more persistent MAOM fractions. On the other hand, Sokol et al. (2024)[113] found reductions in MAOM in the upper soil layer two years after rock amendment, stressing the need to further investigate EW impact on MAOM formation.

While empirical evidence on EW effects on SOC remains limited, agricultural liming provides a useful analogue. Like EW, liming effects on SOC reflect competing processes: stimulation of microbial activity and organic matter mineralisation versus improved soil structure and mineral protection, and increased plant productivity and carbon inputs.[659] In their meta-analysis, Wang et al. (2021)[42] found an average positive effect of liming for moderately acidic, SOC-poor mineral soils, but not for other soil categories. However, organic soils and organic surface horizons more often exhibit SOC losses following liming, reflecting enhanced microbial activity and limited mineral protection.[659,660] Hence, the direction and magnitude of SOC responses depend strongly on initial soil pH, SOC content, ecosystem type, and soil properties. Such dependencies are likely to also apply to EW, highlighting the need for careful evaluation of EW impacts on SOC across diverse soil types and environmental conditions.

Current EW literature on interactions with SOC has several key limitations. Most studies are short-term and limited to bulk SOC measurements, which are likely to obscure moderate effects due to natural soil heterogeneity. Evidence for MAOM formation remains scarce and few studies explicitly link secondary mineral formation to SOC stabilisation. Moreover, priming effects, microbial carbon use efficiency, and SOC responses across soil depth profiles remain to be investigated. Finally, interactions between EW-induced changes in soil hydrology and aggregation have been rarely quantified, despite their relevance for SOC dynamics.[224] Addressing these gaps will require longer-term field experiments, detailed characterization of SOC dynamics across pools and depths, and explicit coupling of biogeochemical, mineralogical, and microbial measurements to robustly assess EW impacts on SOC stocks and stability. Generally, SOC responses should be reported separately from inorganic EW-derived inorganic CDR, because organic and inorganic carbon pools and related CDR differ in formation pathways, durability, reversal risk, and MRV requirements.

7.4.3 Impacts on non-CO_2 gasses

Gases other than CO_2 emitted from agricultural systems are relevant to climate change and public health. Nitrous oxide (N_2O) and methane (CH_4) are potent and well-mixed GHGs and are the dominant GHGs emitted from agriculture. Agriculture, Forestry, and Other Land Use comprised 22% of global GHG emissions in 2019[661] and N_2O and CH_4 emissions comprise 69% and 41% of global anthropogenic emissions (in CO_2-eq), respectively.[662] In addition to GHGs, agricultural practices also comprise 80% of global ammonia (NH_3) emissions[663] and 10% of global nitric oxide and nitrogen dioxide (NO_x) emissions,[661] which can have implications for harmful air pollutants such as ozone (O_3) concentrations and particulate matter ($PM_{2.5}$).[522] In the literature synthesis conducted here, N_2O was the most frequently measured non-CO_2 gas, followed by CH_4, NH_3, and NOx. The measured impacts of EW on ground level ozone, dinitrogen (N_2), and secondary inorganic nitrate and ammonium aerosols appeared just once in the literature.

In the majority of papers in our database, EW decreased N_2O emissions from soils. The primary mechanism proposed for this reduction is the rise in pH in response to EW and related changes in N cycling.[83,106] A higher soil pH increases the emissions ratio of N_2 to N_2O by favouring more complete denitrification, leaving less N_2O to be emitted in the process.[22] However, the reduction in N_2O has also been attributed to enhanced NUE and microbial activity changes.[664] In a handful of instances, EW had no effect on N_2O emissions. One paper that found no effect on N_2O emissions was a 15-year watershed scale application of wollastonite,[70] while another found significant decreases in N_2O emissions in the first round of the study but found no statistically significant decrease during the second growing season.[83] The reason for the lack of observed effect in the second growing season is unclear; however, it may be the result of a microbial priming effect following the initial application of rock dust that does not persist over longer time scales.[665] This highlights the need for long term studies of N_2O emissions in response to EW.

The majority of papers compiled here reported that EW either decreased CH_4 emissions or increased CH_4 uptake by soils. This occurrence has been attributed to the stimulation of iron-reducing bacteria, by the release of iron in silicate rocks, at the expense of methanogens; or an increase root biomass and subsequent oxygen transport from the plant to root by enlarging aerenchyma gas channels, which stimulates methanotrophy and consumes CH_4.[589] In a few

instances, EW was found to either increase or have no effect on CH_4 emissions. Papers attributed any increases in CH_4 emissions to the release of trace metals and a subsequent increase in methanogens.[601] More research is needed to understand the mechanisms underlying the effect of EW on CH_4, and whether trace metals are driving changes in microbial composition.

Enhanced Weathering increased NH_3 emissions in two instances and decreased NH_3 emissions in one instance. Increases in emissions of NH_3 were determined to be driven by pH increase, which suppressed N_2O and NO emissions but moderately enhanced NH_3volatilization.[522] On the other hand, zeolite amendments were found to decrease NH_3 emissions from manure, a major source of NH_3 emissions, by providing more negative binding sites for the NH_3 adsorption.[666] Some papers concluded that EW could decrease NO emissions, attributed to pH-driven changes in nitrification and denitrification. Such reductions could provide air-quality benefits by lowering ground-level O_3 formation and secondary inorganic nitrate and ammonium aerosols. However, Edwards et al., (2017)[385] caution that additional transportation activities from EW could indirectly cause an increase in NOx emissions and raise ground level O_3 concentrations.

Overall, the available evidence suggests that EW can reduce some non-CO_2 GHG emissions, particularly N_2O and CH_4, but responses are context-dependent and must be evaluated on a CO_2-eq basis before concluding whether total GHG emissions decrease.

8 Life Cycle Assessment of EW Implementation

Life cycle assessment provides the framework for evaluating the environmental performance of EW across its full supply chain and defined system boundary, rather than considering CDR in isolation.[667] For EW, this typically includes burdens associated with feedstock production or sourcing, comminution, transport, and field application, and in some studies these burdens are translated directly into net CDR or CDR efficiency. This perspective is critical because EW is materially and logistically intensive: upstream emissions from rock processing and transport can substantially erode the climate benefit of weathering, and under some modelled conditions can even eliminate net removals altogether. Robust LCA is therefore central to assessing whether EW can scale credibly in practice, as performance depends strongly on feedstock choice, siting, transport logistics, and energy supply. Where evaluated more broadly, LCA also reveals important non-climate trade-offs, including health, toxicity, land-use, and other environmental impacts relevant to responsible deployment.[209,250,422,425,426,428,433,668–670]

The literature compiled here includes variables related to GHG emissions and energy demand associated with EW implementation. We identified 38 studies (9.8% of the database literature) that provide quantitative and/or qualitative information relevant to assessing the current state of LCA in the EW literature.[15,18–20,22,23,75,171,178,184,185,209,234,240,250,408,411,420–423,425–430,433,449,539,540,543,559,668–672] Of these, 24 explicitly focus on the sustainability and/or carbon efficiency of EW. This section synthesizes reported estimates of GHG emissions and energy use across the full EW life cycle as well as across individual implementation stages, including mining, comminution, transport, and spreading. Figure 35 summarises the reported ranges discussed below. In addition, we review LCA-relevant methodological choices in the literature, the relative contribution of different life cycle stages, and logistical considerations associated with different feedstocks and deployment contexts. Importantly, nearly all of the evidence synthesized here is derived from modelled scenarios or experimental deployments rather than from large-scale, real-world EW deployments, and should therefore be interpreted as indicative of current understanding rather than definitive operational performance.

8.1 Greenhouse Gas Emissions and Energy Consumption

Across the peer-reviewed EW literature within the database, GHG emissions from EW implementation (not considering CDR rates) range from 1.1 to 215 kg CO_2-eq t_{rock}^{-1} (mean = 51.6, median = 33.15).[18,75,185,209,426,428,539,670] Grinding and transport accounted for the largest contributions, averaging 41% and 39.4% of total EW life cycle GHG emissions, respectively. Some studies also report energy consumption, ranging from 646.2 to 1400 MJ t_{rock}^{-1} (mean=987.2, median=826).[423,425,426] Longer transportation distances and smaller particle sizes generally lead to larger GHG emissions and energy usage.[185,428] Additionally, studies with estimates on the lower end of these ranges assumed more sustainable energy sources (e.g., solar, wind, hydropower).[75,411]

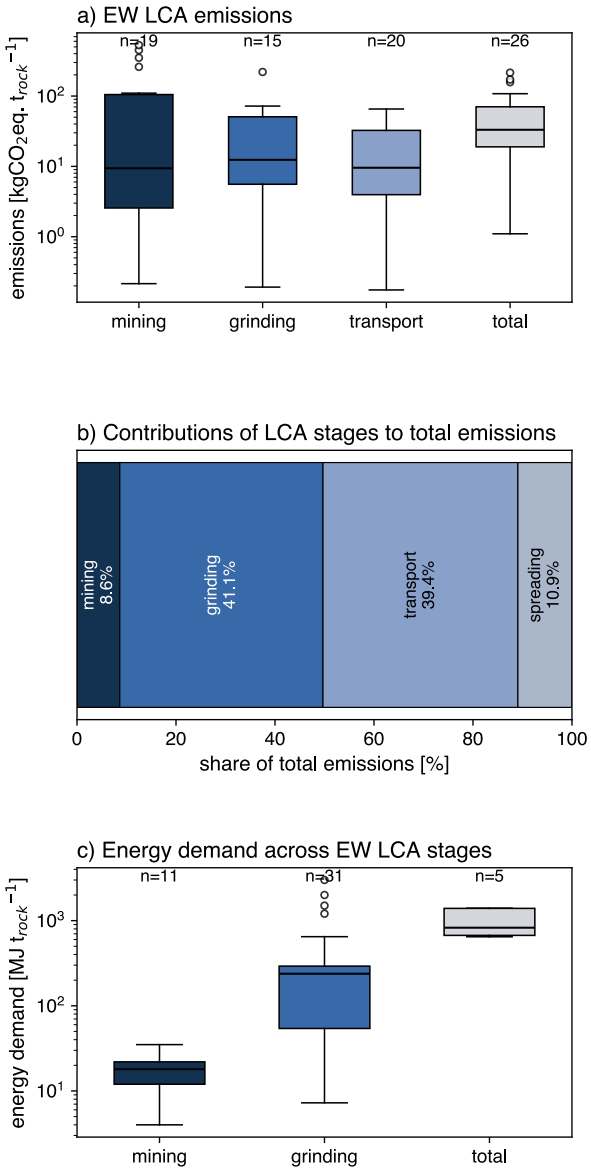

Figure 35: Life cycle greenhouse gas (GHG) emissions and energy requirements associated with Enhanced Weathering (EW) implementation. (a) Distribution of reported GHG emissions from key EW implementation stages across the literature, expressed as kgCO_2-equivalent per tonne of rock applied (central line = median). (b) Mean contribution of individual life cycle stages to total reported EW emissions across studies. (c) Distribution of reported energy demand associated with EW implementation stages, expressed as MJ per tonne of rock.

8.1.1 Mining

Mining represents the first industrial stage in the EW supply chain and therefore constitutes a foundational component of life cycle environmental performance. Quantifying emissions and energy demand associated with rock extraction is essential for evaluating the net climate benefit of EW deployment, as these upstream burdens contribute directly to the overall carbon efficiency of the pathway. Across the literature compiled here, reported GHG emissions associated with EW

feedstock mining span more than three orders of magnitude, ranging from 0.2 to 530 kg CO_2-eq t rock^{-1} (mean = 98, median = 9.4).[18,20,209,408,421,427,429,559,669,670] Corresponding estimates of mining energy demand range from 4 to 35 MJ t rock^{-1} (mean = 18.1, median = 18.0).[15,18,408,422,425,426,429,669] These values reflect substantial variation in assumptions regarding mining practices, energy supply, and feedstock characteristics across studies. Studies evaluating basalt extraction generally report lower mining emissions than analyses focused on olivine or generic "crushed silicate" materials, reflecting differences in extraction intensity and associated industrial processing.[559] At the lower end of the reported emission range, several assessments explicitly assume renewable energy sources, which substantially reduces the mining-stage footprint.[411,429,517] Overall, these results highlight that although mining typically contributes a smaller share of life cycle emissions than comminution or transport, its magnitude remains sensitive to feedstock choice, extraction intensity, and the carbon intensity of the energy system supplying mining operations.

8.1.2 Comminution

Comminution—encompassing crushing and grinding of rock feedstocks—is widely recognised as one of the most energy-intensive stages of EW deployment (Figure 35c) and therefore represents a critical determinant of overall LCA emissions and CDR efficiency (on average 41% of LCA emissions). Across the literature compiled here, twenty studies report GHG emissions or energy demand associated with comminution of EW materials. Reported comminution emissions range from 0.19 to 220 kg CO_2-eq t rock^{-1} (mean = 38.8, median = 12.39).[18,209,408,427,429,540,543,669,670,672] Corresponding estimates of energy demand span 7.25 to 2000 MJ t rock^{-1} (mean = 420.8, median = 237.9).[18,23,178,184,209,408,422,425,426,428,671] As illustrated in Figure 36, smaller particle sizes are generally associated with larger GHG emissions and energy expenditure within this subset of LCA-relevant datapoints, reflecting the additional mechanical energy required to achieve finer grain sizes. Although reducing particle size increases mineral surface area and can therefore enhance weathering rates and theoretical CO_2 removal potential, the associated increase in comminution energy demand can offset these benefits if not carefully optimized. This trade-off highlights the importance of balancing dissolution kinetics and operational energy demand when designing EW systems. In contrast to mined rock feedstocks, studies evaluating mine tailings and other industrial waste materials often assume little or no additional grinding prior to application.[126,410] These materials may therefore offer opportunities to reduce the energy and emissions associated with comminution while simultaneously contributing to waste remediation and resource recovery.

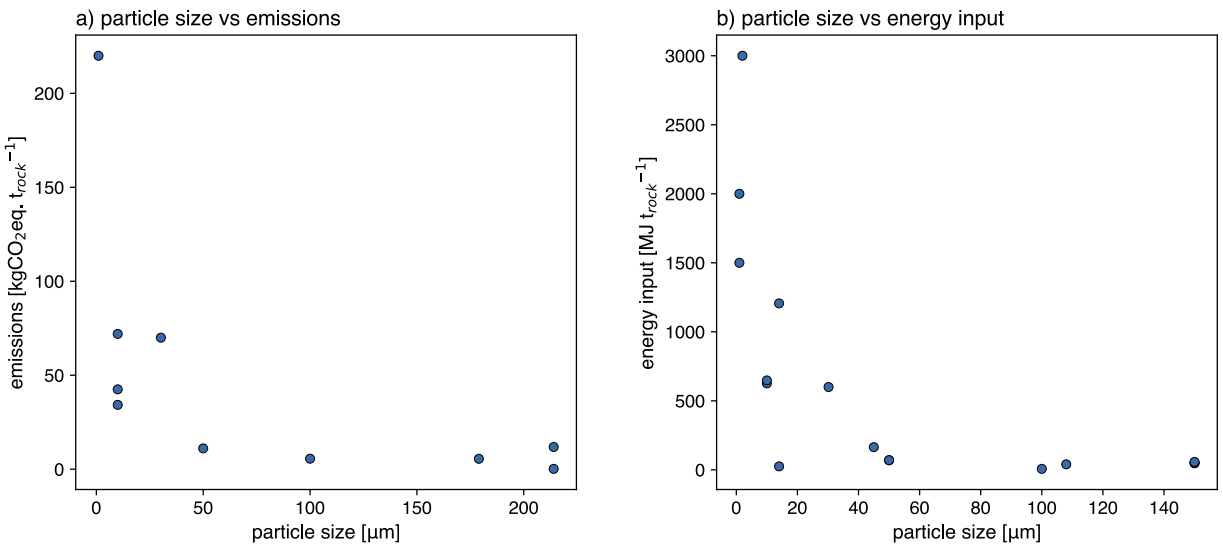

Figure 36: Particle size relationships in Enhanced Weathering life cycle assessments (LCAs). (a) Particle size versus reported comminution-related emissions (kg CO_2-eq t rock^{-1}). (b) Particle size versus reported energy input for comminution (MJ t rock^{-1}). Particle size data combine values reported either as p80 or as mean particle.

8.1.3 Transportation

Transport of rock feedstocks from extraction sites to application areas represents another key component of the EW supply chain and can contribute substantially to overall life cycle emissions depending on transport distance and modality. Across the literature compiled here, thirteen studies report GHG emissions or energy demand associated with transporting EW materials to deployment sites. Reported transportation emissions range from 0.175 to 65.4 kg CO_2-eq t rock^{-1} (mean = 19.23, median = 9.56).[18,75,185,209,411,427,428,517,669,670,672] When normalized by transport distance, emissions range from 0.0157 to 0.177 kg CO_2-eq tkm^{-1} (mean = 0.121, median = 0.0567).[18,75,185,209,427,428,433,540,669]

Across the studies assessed here, transportation contributes on average 39.4% of total EW life cycle GHG emissions, although individual assessments indicate that transport can dominate system impacts under certain conditions. For example, transportation may account for up to 90.4% of total EW life cycle emissions in scenarios with long transport distances or inefficient logistics.[428] Transportation distances modelled in the collected literature range from 10 km to 1650 km, with most studies assuming diesel truck use. Multiple studies identify rail and barge shipping as potentially more efficient methods for transportation.[18,19,185,408,421,428,544,668,672] Collectively, these findings indicate that spatial planning and logistics optimization—particularly minimizing transport distances and selecting lower-emission transport modalities—represent critical factors in reducing the life cycle emissions associated with EW deployment.

8.1.4 Spreading

The final operational stage of EW deployment involves spreading finely ground rock materials across agricultural soils. Compared to upstream processes such as comminution or transport, relatively few studies explicitly quantify the emissions or energy demand associated with this

stage.[209,411,428,517] Available assessments consistently indicate that spreading contributes only a minor fraction of total EW life cycle impacts relative to other stages (Figure 35b).

Most studies assume that EW materials can be applied using existing agricultural infrastructure similar to fertiliser or agricultural lime application. Typical implementation scenarios therefore involve standard farm machinery such as tractors equipped with broadcast or conventional lime spreaders.[23,178,426,428,429] Because these technologies are already widely used in agricultural systems, spreading operations are generally assumed to require limited additional infrastructure and relatively modest energy inputs. As a result, spreading is typically treated as a minor contributor to life cycle emissions in EW assessments, although additional empirical data from field-scale deployment would help further constrain these estimates.

8.2 Additional Considerations for EW LCA

Despite the growing body of EW research, explicit treatment of life cycle emissions remains limited. Across all literature compiled for this study, approximately 90% of studies do not quantify embodied GHG emissions or energy demand associated with EW implementation. This gap represents a major limitation for assessing the net CDR performance of the pathway, as upstream energy and material inputs can substantially influence overall climate outcomes. Systematically incorporating embodied emissions and energy use into EW assessments is therefore essential for producing robust estimates of net CDR and for improving the transparency and credibility of EW deployment claims. Building on the LCA-relevant literature reviewed in this section, the following sections highlight several methodological and conceptual considerations that warrant greater attention in future EW LCAs.

8.2.1 LCA Methodology

Transparent and standardised methodological reporting is essential for ensuring that LCA are comparable, reproducible, and interpretable across studies. Only a small subset of EW studies provide detailed documentation of their methods in compliance with standardised LCA procedures. For example, only five studies explicitly define a clear system boundary for their quantifications, including cradle-to-grave,[209,425] cradle-to-gate,[422,428] and cradle-to-storage approaches.[668] A few studies report inventory databases, including ecoinvent,[209,250,422,425,426] and some of these studies also document LCA software used for analysis—OpcenLCA,[426,428] Simapro,[209,250] GaBi (now Sphera).[425] Six studies report their choices of LCA methods, including TRACI,[250,429] ReCiPe,[425,426,430] and IPCC.[428]

The limited transparency in methodological reporting across the EW LCA literature represents a significant constraint for both scientific interpretation and policy relevance. Without clearly defined system boundaries, inventory sources, and impact assessment methods, it becomes difficult to compare results across studies, reproduce analyses, or evaluate the robustness of reported net CDR estimates. Greater adherence to established LCA standards, such as the ISO 14040 series,[673,674] will therefore be critical for improving methodological consistency and comparability within the field. In addition, future EW LCAs should more systematically

incorporate end-of-life carbon fate within their system boundaries to ensure alignment with emerging MRV frameworks for CDR.[667] Strengthening methodological transparency in this way will be essential for enabling credible cross-study synthesis and for informing robust evaluation of EW deployment at scale.

8.2.2 Other Environmental Impacts and Co-Benefit Assessment

The existing EW literature only sparsely reports impacts across environmental categories beyond climate change (GHG emissions). Where such assessments are included, they consider additional impact categories such as human health impacts associated with industrial processes[209,422,425,426] and potential effects from rock dust dispersal during application.[19] Other reported impacts include resource depletion—such as land transformation and water use[209,422,426]—as well as broader ecosystem effects including ecotoxicity and acidification.[209,422,425,426] Ecotoxicity assessments are particularly relevant for ultramafic rocks and certain industrial materials that may contain elevated concentrations of trace metals.[15,129,130] More broadly, a recent review of LCA approaches for CDR highlighted the lack of quantitative evaluation of co-benefits within the CDR LCA literature, identifying this as a critical gap given the potentially important role of co-benefits in shaping stakeholder acceptance of CDR technologies.[667] A recent study proposes a methodological framework for integrating the co-benefits of EW, including reduced fertiliser requirements and N_2O emission reductions, into LCA and techno-economic analysis.[544] The study revealed significant, region-dependent impacts of co-benefits on the overall environmental and economic performance of EW applications. Expanding future EW assessments to include a wider set of environmental impact categories and systematically evaluating various co-benefits across different regional contexts will therefore be important for developing a more comprehensive understanding of the environmental performance of EW deployment.

9 Synergies and antagonisms with other CDR approaches and regenerative practices

Enhanced Weathering is rarely deployed in isolation within real-world land systems. Instead, it is typically implemented within agricultural, forestry, or managed ecosystem contexts where multiple CDR processes could potentially operate simultaneously. As discussed in Section 3.1.1, EW occupies a distinctive position among CDR approaches because it combines durable inorganic carbon storage with land-management co-benefits, including nutrient supply, soil pH buffering, and improvements in soil fertility. These properties create opportunities for EW to interact with other CDR pathways that rely on biological carbon cycling, particularly biomass-based and soil-based organic carbon sequestration strategies. In many cases, these interactions may enable complementary or synergistic outcomes, allowing multiple forms of carbon storage to occur within the same landscape.

Building on this comparative profile, EW can be viewed as a durable, low-reversal-risk inorganic sink that can complement biologically mediated CDR approaches and regenerative practices. In managed systems, EW may also replenish base cations and other nutrients rather than exporting them via biomass removal, which is particularly relevant where biomass harvest for BECCS or other Biomass Carbon Removal and Storage (BiCRS) pathways increases fertilisation needs.[675] In contrast to organic amendments such as compost or manure that can acidify soils via nitrification, EW can raise soil pH (see also Section 7.3), a potential agronomic co-benefit, particularly in strongly weathered tropical soils where acidity constraints are common. Because EW can often be implemented on existing managed lands, its land-use and land-surface impacts are generally expected to differ from large-scale afforestation or dedicated biomass systems, which are frequently highlighted for land-use trade-offs and regional hydroclimate effects. Enhanced Weathering also supplies nutrients such as silicon that are not always effectively returned via biochar or crop residues.[47] Finally, because EW relies on hydrologic export, and soil carbon saturation is not a concern, this allows for high cumulative rates of carbon storage over time, which may provide portfolio complementarities with biomass-derived sinks that may experience plateaus or gradual losses; thereby supporting strategies that stabilise net CDR over longer timescales.[676] At the same time, EW's MRV readiness lags behind several established approaches (e.g., PyCCS, BECCS), and deployment can face system-level constraints including quarry expansion and comminution impacts,[677] logistics and quality assurance needs,[19,82,132] and environmental concerns (also see Sections 3.5 and 7.1).[131]

The integration of EW with other CDR approaches introduces additional biogeochemical, ecological, and operational complexities. Interactions between mineral weathering, plant productivity, microbial activity, and soil chemistry can influence both the magnitude and permanence of carbon sequestration across different pools. Such interactions may enhance total carbon removal in some systems but can also introduce trade-offs, including shifts in nutrient cycling, altered soil carbon dynamics, or challenges for MRV. This section therefore examines how EW interacts with other CDR strategies and regenerative land-management practices, focusing on potential synergies, trade-offs, and co-benefits across biomass-based CDR, soil carbon sequestration, and emerging hybrid approaches that combine biological, mineral, and technological processes.

9.1 Biomass Carbon Removal and Storage

Biomass Carbon Dioxide Removal and Storage refers to terrestrial-based methods that utilize photosynthesis for the extraction of CO_2 from the atmosphere, storing carbon in trees, crops, residues, and soils. Most BiCRS methods aim to increase organic carbon stored in vegetation and soils. To extend the duration for which the carbon remains sequestered biomass can be used in construction,[678] to produce BECCS and long-term sequestration in geological formations[675,679] or it can be converted to pyrogenic carbon (PyC, biochar) via pyrolysis.[680] Complimentary to natural regeneration, BiCRS is a human intervention and can influence global biogeochemical cycles.[679] Conventional methods include afforestation, reforestation, agroforestry, conservation agriculture, and peatland or wetland restoration and contribute more than 99.9% to current CDR (2.2 $GtCO_2$ yr^{-1}). Novel methods such as biochar application and BECCS contribute 1.3 $MtCO_2$ yr^{-1}.[2] Biomass CDR offers a versatile set of options for both temporary and long-term CDR, while different methods provide distinct sets of co-benefits in the field of ecosystem services, resource use efficiency, and energy supply.[675,679] Combining EW with BiCRS allows "stacking" the CDR potential per area – enabling a higher efficiency of terrestrial CDR methods.

9.1.1 BiCRS synergies

Current research suggests that the combined application of EW with BiCRS could generate a range of beneficial effects that surpass the impacts of either technique alone. These potential effects span improvements in soil fertility, sequestration of both organic and inorganic carbon, microbial processes, crop performance, and ecosystem resilience, although empirical evidence remains emergent and sometimes inconsistent.[69] Several studies indicate that a combined deployment of EW with biochar results in greater total CDR than either amendment alone, primarily through organic carbon stabilisation by biochar and inorganic carbonate precipitation driven by EW,[69,477,681,682] although some studies have only shown additional CDR.[443,676] Suggestions of accelerated inorganic carbon sequestration through enhanced paedogenic carbonate formation and microbial-driven mineral weathering processes still need to be proven.[683–685] Nevertheless, multidimensional CDR pathways can be realized via inorganic and organic carbon, short- and long-term realization of CDR, low and high permanence and risk of reversal, and several ecosystem and hydrological services. This may increase CDR portfolio resilience and value in the SOM while making soil engineering of degraded land more efficient.[76,101,514] Multi-amendment strategies using the same area may reduce the spatial footprint needed to meet specific CDR targets, offering advantages in scenarios with land use constraints.[681] In drought-affected degraded or otherwise marginal soils, these combined approaches have demonstrated promising results, suggesting wide applicability[598,686] while providing a chance to mitigate possible CO_2 emissions through a EW-BiCRS negative feedback.[477,598,676] Moreover, the higher public acceptance of BiCRS compared to EW alone could facilitate social and policy support for integrated deployment.

Improvements such as MAOM formation, SOC stabilisation, and biological induced mineral dissolution have been observed in combined applications of EW and biochar that surpass simple additive effects[108,404,443,617] and thus strengthen the resilience of soils and soil fauna against climate extremes and pests. Another benefit of biochar is the improvement of soil structure, water retention, and porosity, allowing prolonged weathering during dry seasons and optimizing water drainage in fine grained soils.[76,676]

Although agroforestry has been recognised for its contributions to global food security[687] and its capacity for sequestering atmospheric CO_2 both above and below ground[688], the combination with EW for CDR has not been addressed in long-term laboratory or field studies so far. A 2-year field study by Medeiros et al. (2024)[689] showed the improvement of soil quality, biodiversity, and microbial activity from spreading powdered gneiss on coffee plants, but the impact on CDR remains unclear. Studies of EW in matured forest plantations show increased CDR rates by reduced soil CO_2 fluxes and increased tree biomass.[686] In general, degraded soils in forests benefit from elevated soil pH and, supporting sustained tree health and productivity.[675,682]

9.1.2 BiCRS trade-offs

Despite the promising potential for synergies, the combination of EW with BiCRS presents a range of uncertainties and will likely increase complexity, arising mainly due to the high variability and context dependence of outcomes of each single approach. The effectiveness of biochar heavily depends on feedstock type, pyrolysis conditions,[690] and soil initial conditions;[691] agroforestry and afforestation are subject to climatic and environmental changes; and results from EW are influenced by rock mineralogy, grain size, and soil texture.[69,675,692,693] Further, long-term field trial data regarding the integration of EW and BiCRS are still missing[675] as are explorations into how EW impacts the risk of reversal due to droughts and wildfires.[2] Since weathering is slow, EW might not counterbalance the removal of nutrients via biomass for BECCS.[675] Combining EW with afforestation, reforestation, BECCS, or wetland restoration will not solve the competition with food production and other land use.[561,580,682] Both, EW and biomass removal for BECCS and biochar require logistics and may buildup to large volumes for transport expanding costs and energy expenditure, potentially offsetting some climate mitigation gains.[209,694] Also, the application of both rock powder and biochar holds risks from dust and wind blow out,[695] while spreading rock powder after tree planting might be a logistical challenge.[675] In agroforestry, access to spreading equipment might be limited as agroforestry practitioners are mainly smallholder farmers.[687]

Since excessive application of both biochar and EW can cause undesirable shifts in soil pH, affecting nutrient availability, nutrient solubility, and decomposition of SOM due to increase microbial activity[69,91,112] their combined use introduces further complexities and uncertainties to soil chemical interactions. An EW-driven increase in soil pH may influence biochar's priming effects, microbial activity, and nitrogen cycling, potentially leading to reduced mineral dissolution rates and altered soil chemistry.[528,683] The influence of increased alkalinity on organic matter decay rates can compromise long-term carbon storage ambitions and may accelerate organic matter decomposition.[98,112,343] Potential GHG emissions add another layer of uncertainty. Counterbalancing CO_2 release from organic matter decomposition via EW may fail due to slow weathering rates and large differences in the CO_2 fluxes and differences in the CDR potential per ton of material.[676] Although MAOM formation requires minerals, the application of both amendments simultaneously may lead to short-term reductions in MAOM-C attributed to microbial mineralisation of labile C inputs[528] and some treatments exhibited net negative stocks of C and N in MAOM over time, suggesting enhanced microbial decomposition.[528,696]

When nutrient release from EW leads to enhanced biomass production, additional plant uptake of cations will decrease alkalinity and lower the inorganic CDR provided by EW. At the same time, these cations will contribute to CDR via BiCRS but it remains unclear if it can counterbalance the lost alkalinity from soil pore water.[220] Similarly, the release of cations from biochar will interfere with alkalinity produced by EW and may produce a higher inorganic CDR than from EW alone. This is problematic for MRV and CDR accounting as the cations from biochar are biogenic (i.e., part of the short-term nutrient cycle in the soil) and do not provide an additional carbon sink.[676] Monitoring the combined effects and verifying additionality and long-term durability of different forms of carbon storage at once remains complex, necessitating robust long-term, multidisciplinary research and extensive field studies that are carefully designed to address either inorganic and organic CDR on different timescales.[104]

9.1.3 BiCRS co-benefits

Enhanced Weathering can improve soil nutrient availability and ecosystem functioning, leading to increased plant productivity, soil health, and overall BiCRS potential. The release of Ca, Mg, K, P, Si, Fe, and Zn to the rhizosphere by EW improves the nutrient supply and availability. The improved plant growth and health can lead to greater ecosystem resilience and multifunctionality.[106,487,561,580,675] This can be further realized in forests where increased soil moisture supports the access of water to interact with minerals[69,697] and certain trees like bamboo may immobilise heavy metals by accumulating them in their roots.[698] In an agronomic context, EW shows positive effects including reduced fertiliser needs, improved soil structure, higher pH, and elevated microbial activity, which collectively can improve or restore soil health and productivity resulting in higher BiCRS.[47,76,675,682,683]

While EW-BiCRS combinations aim for an increase of biomass, the use of biochar plays a special role in storing both organic and inorganic carbon in the soil. Pyrogenic carbon capture and storage (PyCCS) is the approach that uses pyrolysis in an oxygen limited environment to convert biomass into biochar with high concentrations of pyrogenic carbon (PyC).[680] Biochar can be used in multiple ways including in water filtration, build materials, as reducing agent in the metallurgical industry, in anaerobic digestion, and in animal feeding.[685,699,700] The most common practice is in agriculture where biochar is charged with nutrients via co-composting with manure or urine and returned to the soil as a soil amendment.[701] With this, biochar provides multiple benefits with agronomic relevance such as yield increase, and improvement of plant physiology, soil biology, and soil remediation.[693] Due to high CEC and sorption sites,[702] biochar provides sustained nutrient retention acting as an agent to buffer nutrient release and soil pH changes from EW but also may be able to retain heavy metals released during EW.[69,76] Further, biochar enhances soil physical conditions by improving structure, water-holding capacity, and soil drainage.[97,675,676] The combination of rock powder and biochar may accelerate the development of soil structure and aggregates, including MAOM.[99,101,683,691,703,704] Biochar can also supply Ca^{2+} and Mg^{2+}, which support carbonate formation, while microbial activity stimulated by biochar can lead to increased CO_2 production and CO_2 partial pressure in the rhizosphere which accelerates weathering.[683,705] Practical synergies come from the opportunity to include sequential or simultaneous application, or co-processing approaches such as composting biochar with rock powders or blending rock powders into nutrient-poor biochars.[706,707]

9.2 EW in combination with regenerative and conservation agriculture

The combination of EW with methods aiming to increase SOC emerges as a promising approach for the integration of CDR into agricultural practices. Rock power can be applied in systems already managed according to principles of conservation or regenerative agriculture such as cover cropping, diversified crop rotations, polyculture, reduced/no tilling, mulching and the addition of organic amendments allowing both organic and inorganic carbon sequestration.[99,101,477,681,708] While these sets of techniques deliver multiple agronomic benefits, such as reduced pest pressure or enhanced water retention – they all can support SOC sequestration. The technical CDR potential of SOC ranges from 3.3 to 6.8 $GtCO_2\ yr^{-1}$.[709] However, the sustainable CDR potential is expected to be lower due to economic, social and soil biogeochemical constraints.[709,710] Due to its inherent molecular heterogeneity and site-specific degradation dynamics, the mean residence time of SOC ranges from years to decades.[711–713] Further, SOC based carbon sinks have a higher risk of reversal through tillage, drainage, land-use change, and global warming.[714,715] Still, SOC build-up offers multiple co-benefits including improved soil fertility, water management and ecosystem resilience.[709,715,716]

9.2.1 Regenerative and conservation agriculture synergies

Organic amendments such as compost, manure, and crop residues stimulate the soil-microbial activity which can accelerate the weathering of silicate minerals mediated by the exposure to organic acids and elevated CO_2 partial pressure in the soil.[94,439] Further, both organic amendments and rock powder can ameliorate limitations in nutrient supply. Overcoming such limitations can prompt ecosystem responses such as enhanced plant growth and SOM formation – and proportional carbon sequestration.[69,76,234,528,703,717] Potentially toxic trace elements (PTTEs) released from EW can be immobilised through sorption and complexation to amendments (including biochar) which can further benefit ecosystem performance.[675]

Organic amendments can locally decrease the soil pH due to the release of organic acids and CO_2 from mineralisation. Both can accelerate EW and buffer a pH shift.[47,96,170,477] A high soil pH favours the precipitation of carbonates from the soil solution which can increase soil carbon sequestration as paedogenic carbonates.[96,170] Carbonates, in turn, can stabilise SOC.[47] Alongside the release of Ca^{2+} and Mg^{2+} rock weathering produces reactive secondary clay minerals and iron oxides that bind particulate organic matter (POM) and microbial necromass. This leads to the formation of persistent MAOM which is of higher recalcitrance than free POM and can increase the carbon storage duration.[99,101,106,703,712,718] Additional physical protection of SOM is provided by the formation of soil aggregates involving minerals and organic matter.[99]

Regenerative agriculture practices and organic amendments can improve soil physical properties, including water retention and hydrological functioning. Because mineral weathering requires sufficient water availability, practices that maintain soil moisture, reduce runoff, or increase water retention may enhance EW efficiency, particularly in drought-prone regions.[76,719] Conversely, irrigation, managed runoff, or desalination-supported water supply may also increase weathering potential where water availability limits reaction progress. Mineral amendments themselves may further improve water-holding capacity in coarse-grained soils, creating a potential feedback in which improved soil moisture conditions support continued weathering.[96,170,676] Deeper rooting

(due to better soil structure, improved plant performance, or rotations with complementary root morphology) and increased aeration of the soil may improve the access to CO_2 for EW as well.[76,675] Generally, greater plant productivity can enhance root respiration, delivering more CO_2 towards the weathering process in the soil while increasing carbon storage in roots and above ground biomass.[69,76,477,675]

9.2.2 Regenerative and conservation agriculture trade-offs

Several potential trade-offs were recently addressed by the scientific community – however, uncertainties remain due to the lack of long-term field studies and a strong influence of soil type, climate, and hydrology on both the responses of EW and SOC.[76] The largest concern so far is the occurrence of a priming effect from the stimulation of microbial activity mediated by nutrient supply and soil pH optimization accelerating soil respiration via the mineralisation of organic matter and proportional CO_2 release.[101,477,696] This may temporarily offset SOC gains, especially when mineral dissolution or nutrient release is rapid.[100,111,112] On the other hand, a pH spike from EW may harm microbial communities[47,177] and temporarily reduce soil respiration.[686] Still, high uncertainties remain in the absolute quantification of positive and negative priming effects as short-term experiments do not capture the longer-term feedbacks.[179]

When rock powder is incorporated by tillage, the associated physical disturbance of topsoil and shallow root systems may counteract some mitigation benefits of no-till or reduced-disturbance practices.[477] Monitoring SOC responses and EW-derived inorganic CDR is already challenging when each is assessed separately, because both involve high spatial variability, cost constraints, uncertain storage permanence, and complex biogeochemical feedbacks.[104,113,475,676,720] Combining SOC-focused practices with EW may therefore exacerbate MRV challenges and costs, as organic and inorganic carbon pools differ in mechanisms, permanence, and accounting requirements. A combined application further increases system complexity potentially reducing MRV reliability. Large amounts of cations may be bound in metalorganic ligands via organic acids or incorporated in SOC.[343,721,722] These cations do not charge balance DIC and cannot contribute to CDR any longer. Although these cations still facilitate CDR via BiCRS and SOC, their contribution per mole of released cation from EW is difficult to account for precisely.[220] From an agronomic perspective, nutrient release from EW may not be fast enough or balanced with plant or microbial demand to substitute for fertiliser application.[675] The drainage of water in an agricultural soil can have a major impact on weathering rates and export of alkalinity from soils and can be impacted by soil and rock power grain size.[63,97,676]

9.2.3 Regenerative and conservation agriculture co-benefits

Despite the uncertainties, the combination of EW with SOC enhancing agricultural practices shows evidence of co-benefits for both inorganic and organic CDR pathways. A key benefit is a greater total carbon sequestration per land-unit without requiring land-use changes.[47,101,106,113,234,477,528,681,703] The stabilisation of SOC by the formation of soil aggregates and MOAM can improve the soil structure and hydrological parameters - improving the overall soil fertility and resilience.[100,101,113,686,703] Improved soil moisture management due to organic and inorganic amendments can maintain not only EW but also biological activity in drier seasons.[96,97,170,676,723] Additional supply of geogenic nutrients from EW can lead to higher crop

yields and a reduced need for mineral fertilisers.[47,76,100,104] The rehabilitation of degraded soils by the combination of EW with SOC build-up not just supports food production under the pressure of rising food demand and global warming but also contributes large-scale ecosystem restoration.[76,118]

9.3 EW in combination with other methods and CDR approaches

9.3.1 Mineral-enriched biochar

The co-pyrolysis of biomass and minerals was recently suggested as an avenue to enhance CDR efficiency by increasing pyrolysis carbon yield and PyCCS recalcitrance.[724] Likewise, rock powder can be added to biomass pre-pyrolysis to produce rock-enhanced (RE) biochar. Rock-enhancement is a novel approach to combine inorganic and organic CDR pathways in a single composite material.[725] Several studies have suggested that the addition of minerals rich in alkali- and alkaline earth metals (AAEMs) is effective in enhancing the yield and stability of PyC.[726–728] This may also have impacts on the long-term buildup of SOC.[729] Rocks suitable for EW typically contain these AAEMs in high concentrations – however they remain largely un-reactive during pyrolysis.[725] RE-biochars may contribute more to CDR by promoting faster and more effective mineral weathering, which enhances the formation of stable secondary carbonates.[724,726] Recent laboratory experiments have shown higher weathering rates of co-pyrolysed minerals through an increase of their specific surface area and a closer mineral-water interaction facilitated by biochar's high water holding capacity.[730] The exposure of serpentine-rich minerals to high temperatures resulted in a thermal activation and the formation of olivine minerals resulting into significantly higher CDR from increased Mg^{2+} release.[731] The biochar–mineral interfaces established during co-pyrolysis promote the formation of organo-mineral complexes that stabilise SOM and PyC.[730] Further, phyllosilicates such as montmorillonite or vermiculite augment this mineral stabilisation potential.[732,733] RE-enhanced biochar may improve the habitat quality for soil-microbial communities which can generate synergies with EW, leading to enhanced nutrient availability and mineral transformation, thereby strengthening both inorganic and organic soil carbon sinks.[724,726]

Nonetheless, several limitations warrant consideration. Mineral additions may not consistently increase the fixed carbon yield but some mineral additives can alter pyrolysis dynamics unfavourably or accelerate post-application oxidation.[726,728] Co-pyrolysis outcomes are not uniformly synergistic and both reduction and increase of the potential carbon sink have been reported depending on the mineral feedstock.[730,731] Moreover, mineral inputs originating from industrial by-products can introduce contaminants, including heavy metals.[725,734] The presence of "pyrogenic coating" (secondary char coatings) on mineral particles following co-pyroylsis was described by Meyer zu Drewer et al. (2025)[725]. This process may be exploited to design functionalized RE-biochars but could also reduce mineral-water interaction.[725] Despite these concerns, the deployment of RE-biochars presents a resilient dual-pathway by combining recalcitrant PyC with forms of inorganic carbon sequestration.[725,728] The combination of two materials in one homogeneous aggregate of high bulk-density and reduced dust evolution bears practical advantages in agriculture.[676]

9.3.2 Biomineralisation and biotechnological approaches

Biomineralisation offers a complementary mechanism to EW. Hereby, biological processes accelerate mineral dissolution and carbonate precipitation through microbial or enzymatic catalysis.[682] Biotechnological strategies employing carbonic anhydrase enzymes or engineered microbial strains can further accelerate silicate dissolution and carbonate mineralisation, thereby enhancing the CDR potential of EW on an economic viable scale.[735–740] Microbial processes applied to mine tailings or industrial residues can simultaneously sequester CO_2 and remediate hazardous waste streams.[739,741–743] Microbial biomass produced in these systems may additionally serve as a feedstock for fuels or bio-based chemicals.[744] Still, scalability of these systems is limited by the sensitivity of microbial activity to environmental conditions such as moisture, temperature, and CO_2 availability.[745] Biomineralisation systems require energy and nutrient inputs for cultivation and process control, potentially reducing net carbon benefits.[746] Moreover, the use of untreated industrial or mining waste material poses the risk of releasing toxic elements,[747] and engineered microorganisms raise biosafety concerns. When integrated with EW, biomineralisation processes may increase weathering kinetics, accelerate carbonate precipitation, and improve contaminant stabilisation, offering a combined remediation and CDR strategy. Joint deployment could be especially effective in industrial settings, mine sites, or engineered soil systems, where reactive mineral surfaces from EW can be paired with biological catalysts to significantly enhance carbon sequestration efficiency.[682] Beyond these direct effects, yield enhancements associated with EW in agricultural and forest systems may also increase biomass availability per unit land area, thereby improving the efficiency of biomass-based CDR pathways such as BECCS, biochar production, or biomass burial. In this way, EW can indirectly amplify the CDR potential of bio-based approaches by increasing organic feedstock supply without requiring additional land expansion.

9.3.3 Phytomining and soil remediation

Phytomining presents a biologically mediated pathway for metal recovery from soils and can build onto the geochemical changes induced by EW.[179,675] Hyperaccumulator species can extract substantial quantities of heavy metals such as Ni and Cr from soils concentrating Ni up to 3% in biomass and up to 20% in biomass ash.[524,748] EW of ultramafic and mafic rocks can mobilize Ni and Cr (cf., Sections 7.2.2 and 7.2.4), which creates opportunities for consecutive phytomining which simultaneously mitigates metal accumulation in soils.[150,749] Though generally more rare than for Ni, several hyperaccumulator species are capable of Cr uptake,[750–752] supporting multi-metal extraction and remediation. Enhanced Weathering also increases the availability of other essential nutrients, thereby stimulating plant biomass production and CO_2 uptake, which in turn increases the realized yield-potential of hyperaccumulators.[75] High-biomass accumulating species such as willow or bamboo may serve a dual purpose as being both phytomining crops and BECCS feedstocks, enabling carbon-negative energy production and metal recovery from combustion ash.[753–755] These interactions are mirrored in emerging concepts such as carbon-negative Ni production ("NegativeNickel").[749] If metals accumulate primarily in roots, extraction requires disruptive harvesting that may disturb SOM and reduce SOC stocks.[755] Nevertheless, 99% of mobilised metals may remain unavailable for plant uptake (ref. [184]; cf., Section 7.2.4) while increased soil pH following EW may further reduce metal bioavailability and species-specific uptake patterns can leave certain metals unaddressed, such as Co or Cs.[756] Despite these constraints, by removing metals mobilised during weathering, phytomining reduces the risk

of long-term metal accumulation that could otherwise limit EW deployment.[150,756] Nutrient release induced by EW enhances plant productivity, and coupling with BECCS enables metal recovery alongside negative-emissions energy creating synergies and connections between resource recovery, CDR, and sustainable land management.

Similar to phytomining, EW can also be integrated as a soil remediation strategy to address heavy metal contamination through stabilisation of toxic metals by reducing their solubility and bioavailability through adsorption and precipitation processes[423,681] while enhancing the stabilisation of both organic and inorganic carbon pools.[104] However, in certain soil conditions, EW may mobilise specific metals, necessitating careful selection of mineral amendments or vegetation to ensure safe immobilisation.[423,681] The increased nutrient availability associated with EW can alter plant community composition, and the long-term ecological consequences remain insufficiently studied.[423,681]

9.4 Analytical and systemic comparison

Beyond the physical combination of EW with other CDR methods, the integration into model-based mitigation scenarios provides a broader understanding of its role within CDR portfolios. In climate-mitigation modelling, EW is typically deployed alongside other CDR approaches as an independent method rather than through physical integration with other methods. Strefler et al. (2021)[78] shows that EW appears within multi-method climate scenarios, contributing through its own sequestration pathway while operating in parallel with other CDR approaches. The modelling of CDR approach interactions is still an emerging frontier; climate mitigation and net-zero scenario frameworks increasingly emphasize CDR approach portfolios rather than single-method strategies, treating CDR approaches as a coordinated suite, supporting portfolio-style deployment of EW.[413,682,757,758]

Assessments of CDR implementation also highlight policy mechanisms such as agricultural subsidy schemes that could co-incentivize EW with other CDR approaches.[413,759–761] Separating CDR from emission-reduction target mixed portfolios in which EW complements technologies such as CCS can strengthen mitigation outcomes.[413,762] For example, EW can address diffuse agricultural CO_2 emissions while geological storage manages concentrated industrial emissions.[104] Still, such combinations may require infrastructure including MRV and technical capacity that are not uniformly available.[423] EW, due to its lower technological readiness and associated uncertainties, may benefit from combinations with other methods, like BECCS or DACCS, to bolster effectiveness, scalability, and public acceptance.[541]

10 Public perception, climate justice, and stakeholder engagement

Research on EW has largely focused on its geochemical mechanisms, CDR potential, prospective costs and environmental impacts, and life cycle considerations. However, the feasibility and legitimacy of large-scale deployment will not ultimately depend on technical performance or economic viability alone. As with other climate mitigation and CDR strategies, EW necessarily operates within and across salient social, political, legal and ethical contexts that shape whether, where, and to what extent it can be effectively implemented. Public and stakeholder perceptions, governance structures, and questions of justice therefore represent critical dimensions for assessing its deployment, particularly as EW moves from the stages of conceptual assessments and experimental trials toward real-world application at scales meaningful for potentially helping to mitigate climate change.

These considerations are especially relevant because many regions identified as highly suitable for EW—such as humid tropical agricultural systems—are located in parts of the Global South that have contributed relatively little to historical GHG emissions but may experience disproportionate impacts of both climate change and climate interventions.[1,5] The following sections review emerging research on public perceptions of EW, the relevance of ethical and climate-justice considerations, and the growing need for broad stakeholder engagement in shaping responsible research, governance, and potential implementation pathways. Also note that a range of related topics are discussed in Section 7.1 as societal "side effects".

10.1 Public Acceptance and Perceptions

Public acceptance is generally acknowledged as critical for EW development and deployment,[20,541,552,554,763] with the key role of farmers and local stakeholders also recognised.[104,575] Public stakeholders generally have a low awareness and understanding of EW.[2,21,423,569,573,764–766] In specific, participants have characterized EW as abstract, confusing, and difficult to grasp,[764,767] or even inducing feelings of "dread".[768] Most attention in the literature and public discourse, for instance on social media, focuses less on EW versus more familiar, ecosystems-based methods as well as those such as DACCS.[2]

Keeping in mind the nascent understanding of EW, there appears to be relatively more support than opposition.[2,569,764–766,769] Moreover, there seems to be stronger support for EW – and for CDR methods in general—among publics in the Global South.[569] Further research on public perceptions in the Global South is much needed, given the relevance of such regions (cf., Section 5.2.3) for EW deployment versus how little research has engaged with them.[555,569,571,770] In terms of factors that drive support, both trust in relevant actors and institutions, such as industry, scientists, and national governments, and prior knowledge of EW are found to matter.[765,766,771,772] Furthermore, those with stronger beliefs that science and technology will help solve climate change were more likely to support EW,[569,764] while those more concerned about tampering with nature were much less likely to do so.[764,766,772]. Concern about climate change had a more inconsistent, though often positive effect on EW support,[764,766] with some research finding this factor to matter less once others are included.[765,771] Rather, some studies suggest that it is more affective factors such as hope and/or worry about climate change which strongly (and positively) predict EW support.[572,773]

Lastly, while the relevance of socio-demographic factors is inconsistent, Sovacool et al. (2024a, 2024b)[570,774] analyses of data from a 30-country survey exercise identified greater EW support among younger groups (44 and under), male versus female, and among minority and indigenous groups, specifically those living in urban areas.

In any case, Cox et al. (2020)[573] have described such support as "conditional" given unfamiliarity of EW among participants and the variety of concerns which they raised—noting, for instance, that support is stronger for research and field trials than large-scale deployment.[2,569] Concerning a field trial for palm oil agriculture in Malaysia, Cox et al. (2025)[555] also found that workshop participants better understood the consequences and purpose of field trials vis-à-vis if EW were to be scaled-up. Opportunities for deliberation in groups, to ask questions to experts as well as information and real-time data about projects that were underway all positively influenced support.[555,574,775]

Still, the level of support for EW is usually less than for other CDR methods.[569,572,573,776–778] The concerns which publics referenced the most are: opposition to mining; interfering with nature and oceans in specific; the impact on rural and agricultural landscapes; and distributional (un)fairness and justice in the Global South.[541,569,571,574,581] Another crucial concern was the perceived failure of EW to address the root cause of climate change.[569,573,776] Indeed, Cox et al. (2020)[573] directly cited this "temporal dilemma" as reason for the greater support of research vis-à-vis large-scale deployment for many people.

Diverging views on the potential acceptability of EW also emerged when considered from an objective (usually informed by the natural sciences) vs. subjective perspective, where empirical methods were used to engage members of the public. As examples of the former, one evaluation suggested EW could achieve broad public acceptance owing to its low cost and infrastructure requirements and remediation potential,[779] and with social impact scoring used as a proxy for potential acceptance to suggest that the below-average population density near quarries could result in fewer social conflicts and greater likelihood of acceptance for further extraction activities.[133] In contrast, the social sciences literature has identified substantial concern about impacts on rural and agricultural landscapes.[574] Drawing on 44 focus groups in 22 countries around the world (one rural, one urban in each), Sovacool et al. (2025)[780] found greater concern about the need for more extractive activities in urban versus rural areas (at least in a European context); the latter were more likely to perceive EW as relatively natural, less expensive, and not needing to confront storage-related hazards.

In general, the European focus groups frequently wrestled with siting concerns, land-use conflicts, and the prospect of NIMBYism (Not In My BackYard-ism) in their discussions of land-intensive forms of CDR. Analysing these focus groups on a global level, Low et al. (2024)[571] identified a fault line for EW, where the balance of hopes and concerns broadly depended on whether it was envisioned primarily as an expansion or refinement of mining operations or rather in terms of additional benefits through agricultural management. Notably, a small though global North-South cross-cutting group of participants discussed co-benefits of EW for soil enhancement, also highlighting the need for further assessment of rock types, mining locations, and efficacy as a soil amendment (see also ref. [555]). While there was general discussion on the siting and logistics of potential mining operations in many focus groups, participants from countries with large mining

sectors, such as Australia and South Africa, also considered how EW could be used to recycle mining waste or clean up existing quarries. In this regard, there is some early evidence of EW discussions taking on nationally specific flavours—though it is again crucial to reiterate that these focus groups were much relatively less likely to engage with EW during their discussions and found sense-making to be much more difficult, that is, unless there was a specific individual or national familiarity with the extractive or agricultural sector.

10.2 Climate Justice and Ethics

Enhanced Weathering raises various ethical and justice-related considerations, emerging first and foremost from the greater promise for deployment in hot, humid climates and tropic regions across the Global South.[19,23,24,78,245] Where this creates opportunities for higher yields in food production or reduced need for costly fertilisers, local farmers and communities could benefit. Particularly if new mining and extraction were to be required, however, the placement of such activities in specific regions could create severe distributional injustice issues, even to the point of land-grabbing, dispossession, or cultivating "sacrifice zones".[560,565,583]

In a more general sense, EW, like other CDR technologies, raises justice issues regarding the displacement of mitigation efforts (e.g., ref. [6]), especially if poorer or lower-emitting regions needed to further reduce their emissions as a result.[78,581] EW could also promote intergenerational injustice, if future generations are left to bear most of the adverse consequences.[561,581] Looking at public perceptions of EW, environmental injustice and the issue of unequal burden-sharing between poorer and richer countries have consistently been underlined.[555,569,571,573,574,580,774] Particularly where EW was to be deployed in the Global South, publics raised concerns about exploitation and "environmental dumping"—even where this was initially framed in terms of job creation.[574,580] Publics across the Global South were, moreover, most negative about the risks that EW (and other CDR methods) would decrease the motivation to reduce CO_2 emissions.[569]

10.3 Stakeholder Engagement and Collaboration

The relevance of stakeholder engagement for developing and deploying EW has mostly been an implicit or abstract consideration, until recently. Some researchers gesture towards outreach and dialogue with local stakeholders and communities as a general principle and requirement, specifically noting farmers and mineral extraction as key actors;[21,22,131,161] others suggest the value of more theoretical work to foster a fairer allocation of benefits and costs among stakeholders, via multi-agent decision-making models.[420] Using a large expert-interview exercise, Sovacool (2023)[580] highlighted the concern that scaling-up of EW can lend itself to more consolidated, top-down governance, rather than a community-based approach.

Lessons from actual engagement with real-world stakeholders are starting to emerge, including through the greater involvement of social sciences researchers. Haque et al. (2024)[556], for instance, offers insights into the difficulties of engaging and coordinating with stakeholders in the context of field trials. Santos et al. (2023)[578] similarly stressed the usefulness of listening to farmers when it comes to suitable application rates. Meanwhile, qualitative research with local communities and stakeholders, in Malaysia and the United Kingdom, respectively, has established how conditions for EW implementation must be: tailored to local contexts (e.g., via job creation, local

infrastructure investments, local-scale monitoring), promote fair, open governance, and provide transparent and unbiased information about impacts.[555,775] While those involved in workshops raised several concerns and questions about EW (e.g., effectiveness removing CO_2, potential for environmental contamination, mitigation deterrence, and absence of remediation plans), the studies also found that acceptability could be promoted by providing opportunities for deliberation and to ask questions (to experts), information about projects that are underway, and presenting real-time project data. O'Sullivan et al. (2025)[775] also identified a broad preference for using public funding versus private financing on the grounds of being fairer, more transparent, and avoiding risk of mitigation deterrence. Using a "question-led innovation" approach, where public and local stakeholders could ask actionable questions to inform ongoing research efforts, Cox et al. (2025)[555] identified agreement between stakeholder interests (in Malaysia) and the current research on EW. Notably, questions focused on rock resources and their sources, including the need for new mining operations. The second-most discussed subject were benefits for local farmers and crop yields—more common than questions on health and/or environmental impacts, and regulation. Stakeholder concerns did however indicate knowledge gaps, notably on full life cycle CO_2 emissions, including sequestration, impacts on aquatic life or worker health, and socio-economic factors such as governance, cost and regulation. Pointing to the fraught history in the local context, Cox et al. (2025)[555] highlighted underlying concerns about resource mining impacts and carbon goals being over-emphasized vis-à-vis conservation and biodiversity.

11 EW in Integrated Assessment Models

The scale of deployment of EW will depend, among other factors, on climate mitigation ambition, the availability of required resources, and its cost competitiveness relative to other CDR options. Integrated assessment models (IAMs) are forward-looking optimization frameworks used to derive cost-optimal mitigation pathways while representing interactions between the energy system, land use, and the broader economy. By linking these sectors within a single modelling framework, IAMs allow exploration of how technological developments, policy choices, and biophysical constraints influence future mitigation portfolios. This section first outlines how EW is represented in IAMs, focusing on assumptions regarding feedstocks, weathering processes, deployment limits and cost (Section 11.1), and summarises the insights emerging from scenario analyses (Section 11.2).

11.1 Integration in IAMs

The integration of EW in IAMs is still emerging and currently limited to a small number of models and studies. The Sixth IPCC Assessment Report (AR6) identified two IAM-based analyses including EW,[661] using the REMIND,[78] and the En-ROADS models.[781] Since then, EW has also been implemented in GCAM,[782] GET,[717] and EPPA,[418] and additional IAMs are exploring EW scenarios in preparation for the 7th IPCC Assessment Report. Additional scenario-modelling approaches focus either specifically on EW or on broader CDR portfolios.[19,22] While these analyses similarly consider factors such as cost, energy demand, and land constraints, they typically rely on different modelling frameworks and represent a narrower portion of the coupled energy–land–economy system.[783–786]

Table 6 provides an overview of key implementation differences across IAMs. All IAMs represent basalt as the primary feedstock and assume a maximum sequestration potential of approximately 0.3–0.33 tCO_2 per tonne of basalt under complete dissolution. Existing IAM studies do not explicitly constrain deployment by global basalt resource availability or mining capacity, allowing expansion beyond current mining levels. Instead, the theoretical deployment limit is primarily determined by the land area available for application. Implementations based on Beerling et al. (2020)[19] and Strefler et al. (2018)[23] restrict deployment to croplands located in sufficiently warm and humid climate zones. Under constant cropland area assumptions, this framework yields a weathering potential of around 4.9 GtCO2 yr^{-1}.[23] In contrast, Chiquier et al. (2025)[418] find a cropland-based removal potential of 7.7 $GtCO_2$ yr^{-1} in 2100 in one EPPA scenario, reflecting different assumptions regarding cropland availability and climatic suitability. Beyond croplands, the GET model additionally allows basalt application in forests, albeit at higher costs and energy requirements for rock spreading. This substantially increases the modelled cost-effective CDR potential through greater land availability and through simulated increases in biomass growth that contribute additional biotic CDR.[717]

The models take different approaches to geographic differentiation as well as the influence of climate conditions and grain sizes on the weathering. GCAM and EPPA employ an aggregated approach, taking the transport distance of 300 km covering 80% of climate-suitable cropland from ref. [23], and applying an upper bound on the annual spreading of rock on fields derived from

Table 6: Overview of EW representation in common IAMs.

IAM	REMIND	GCAM	GET	EPPA
Earliest Paper	Strefler et al. (2021)[78]	Fuhrman et al. (2023)[787] Morrow et al (2023)[782]	Gaucher et al. (2025)[717]	Chiquier et al. (2025)[418]
Key references used	Strefler et al. (2018)[23]	Beerling et al. (2020), Strefler et al. (2018)[23]	Strefler et al. (2018)[23], Renforth et al. (2012)[18], own analyses	Strefler et al. (2018)[23], Renforth et al. (2012)[18], own analysis
Cropland application	Yes	Yes	Yes	Yes
Forest application			Yes	
Biotic CDR considered			Only for forest application	
Weathering explicitly modelled	Yes	No	Yes	No
Application rate limits on cropland	15 kg m^{-2} (cumulative)	Implicit in the supply curves	15 kg m^{-2} (cumulative)	2 kg m^{-2} yr^{-1}
Grain size (µm)	20		20	10
Weathering rate	Depending on warm or temperate climate	Implicitly region-specific through supply curves from Beerling et al (2018)	1-26% p.a.	NA
Agricultural co-benefits		option available, not used in studies		
Maximum potential	Theor. max. ~4.9 $GtCO_2$ yr^{-1} based on Strefler et al (2018). In high ambition mitigation scenarios maximum around 4 $GtCO_2$ yr^{-1}	Theoretical cropland potential like GCAM & REMIND. Incl. forest application & biotic CDR in 1.5°C Scenario with low overshoot: up to 12.4 $GtCO_2$ yr^{-1}	7.7 $GtCO_2$ yr^{-1} by 2070, 3.2 $GtCO_2$ yr^{-1} in a constrained scenario	
Total cost range	~200 USD2015 tCO_2^{-1} based on Strefler et al (2018) ~ 50% thereof region-unspecific investment and operation and maintenance cost.	Based on region-specific cost curves from Beerling et al (2020) with cost of ~50-400 USD tCO_2^{-1}. Short-term cost markup of 138 USD tCO_2^{-1}based on estimate range in Strefler et al (2018) that is faded out until 2050.	Cropland application: 43-132 USD tCO_2^{-1} Forest application: 146-364 USD t^{-1} basalt. Energy cost make up 20-40% thereof, depending on carbon price. Including the biotic CDR effect, removal cost of 20-166 USD tCO_2^{-1}.	265-340 USD2021 tCO_2^{-1} Relatively high likely due to higher energy requirements for grinding to 10 µm, as non-energy cost are taken from Strefler et al (2018).

underlying weathering assumptions. The weathering process is not explicitly modelled. REMIND and GET, on the other hand, explicitly model weathering and limit the amount of non-weathered rocks per ha at any point in time (Table 6). These differences imply different temporal links between application of rocks and the CO_2 removal accounted for in each year. They also result in differences in the timing of cost incurred for rock spreading and the accounting of associated CO_2 removal compensation.

Electricity demand for basalt mining, crushing, and grinding, as well as fuel demand for rock transport, are typically based on estimates from Strefler et al. (2018)[23] or Renforth (2012)[18]. Associated costs and CO_2 emissions are calculated endogenously within the models and contribute to regional cost differences (see also Section 6 on costs). Assumptions on transport distances and investment, operation, and labour costs are generally derived from average or upper-bound estimates reported in Strefler et al. (2018)[23] (Table 6). Cost dynamics associated with increasing deployment differ across models. In REMIND, for example, transport distances are represented explicitly, leading to rising costs as deployment expands to more distant croplands. The model also prioritizes areas with higher weathering rates within each region, resulting in increasing marginal removal costs as deployment expands into areas with less favourable weathering conditions. In the more aggregated representation used in GCAM, increasing marginal costs per tonne of CO_2 removed are represented through region-specific EW supply curves derived from Beerling et al. (2020)[19], which approximate the expansion of deployment into areas with progressively less favourable weathering conditions.[787]

In summary, despite differences in how weathering dynamics and application constraints are represented, current IAM implementations of EW remain relatively simple and are largely shaped by a small set of foundational studies that are commonly used as benchmarks for theoretical CDR potential.[18,19,23] Across the models summarised in Table 6, EW deployment in ambitious mitigation scenarios with global cooperation reach several gigatonnes of CDR per year. Cropland-only implementations generally approach their maximum removal limit with roughly ~3–7 $GtCO_2\ yr^{-1}$, while models that include additional land areas or biotic feedbacks can yield higher values (Table 6). Differences relative to the technical potential ranges discussed in Section 5.2 primarily reflect differing assumptions about what is technologically achievable, together with the fact that many IAM benchmarks predate recent recognition of geochemical losses and feedback processes that may reduce realised CDR (Section 3.4).

11.2 Performance of EW in IAM scenarios

The existing IAM scenario literature that includes EW remains relatively limited. Across the studies identified in this review, ten IAM scenario analyses include EW in their modelling frameworks.[78,416–419,717,781,787–789] Most of these studies examine EW as part of broader CDR portfolios, with only one study explicitly focusing on EW as the primary technology of interest.[717] Even where EW is represented in IAMs, it is therefore not treated as a standard technology like BECCS and remains much less extensively covered than DACCS.[32] This is also reflected in scenario design decisions, for example the exclusion of EW alongside DACCS in scenarios framed as “limited CDR”.[789]

At present, most IAM implementations do not explicitly represent all geochemical limitations or feedbacks on primary weathering rates and subsequent CDR or only include a subset of the

processes now recognised as relevant (see also Sections 5.2.1 and 3.4). These include rate-dependent effects such as increasing CDR losses at higher application rates due to secondary phase precipitation, declining weathering rates associated with increasing porewater saturation, limits on the availability of rock powder (Section 5.2.1). In addition, several EW-related impacts discussed in earlier sections—such as side effects from large-scale mining, water requirements for slurry preparation, or MRV challenges—are typically not represented explicitly beyond indirect assumptions such as delayed technology availability.[418] Consequently, some of the higher CDR rates modelled in IAM scenarios may depend on assumptions about weathering efficiencies that remain uncertain under real-world geochemical constraints.[161,172] Potential agronomic co-benefits[47] (cf., Section 7.3) are likewise generally not represented, apart from an experimental implementation in GCAM treating basalt as a fertiliser[782] and the inclusion of biotic CDR for forest application in GET. Nevertheless, these frameworks remain highly valuable for exploring the role of EW within broader CDR portfolios, and their representations are expected to improve as empirical understanding of how EW unfolds in soils continues to advance and updated projections of technological potentials these models rely on become available.

In comparison to other CDR options, in IAMs, EW has intermediate cost and potential (Table 6),[78,788] making its deployment sensitive to the mitigation ambition[717] and, at least regionally, EW cost assumptions.[419] The availability of EW increases total CDR deployment.[78,717,787] The deployment can be sufficiently high to reduce mitigation cost through substitution of more expensive mitigation options and/or allowing delay of mitigation through the additional removal capacity, which depends on the scenario design.[78,717] Deployment tends to increase with less favourable conditions for other CDR technologies, for example when geologic CO_2 storage or bioenergy supply are limited, even in regions where the theoretical potential is small.[417,419]

In high mitigation ambition scenarios, EW can approach the models' theoretical deployment limit (Table 6) and scales rapidly. For example, it approaches the maximum potential by 2040 in global and regional GCAM studies.[419,787] A notable exception is Apeaning et al. (2025)[789] with a global deployment of only 0.75 $GtCO_2\ yr^{-1}$ for a 1.5°C scenario. Although not discussed in the paper, the reason may be their scenario's regionally differentiated mitigation ambition which increases with gross domestic product (GDP). The climate conditions cause the tropical regions of Latin America and Asia to have a higher deployment potential, which includes countries with lower GDP and historic emission responsibility.[78,787] Gigaton-level deployment may thus depend on global cooperation and could require large international finance transfers, raising equity concerns.[78]

The studies with gigaton deployment also critically reflect on the upscaling in relation to logistical challenges, reflecting tensions relating to requirements for technical potentials as discussed in Section 5.2.1. Although EW can use infrastructure and equipment that already exist today,[22,788] large-scale deployment would imply significant modification of material supply chains and expansion of basalt mining efforts beyond the current size of the coal industry.[78,418,419] Potentially increasing costs from the upscaling of mining are not reflected in the cost assumptions, nor are other social and legal barriers represented.[22,788]

Finally, IAMs allow a systems perspective on EW's electricity and fuel demand. Beerling et al. (2020)[19] restrict deployment to 3% of countries' final energy production, which indirectly limits EW's impact on the energy system in GCAM.[787] Nonetheless, the sub-regional distribution can

lead to much higher local shares.[419] Beyond the individual technology perspective, EW energy requirements also need to be considered in the context of the entire CDR portfolios deployment. For example, Afrane et al. (2025)[416] show scenarios in which the total CDR electricity demand is up to 20% in some regions, with DACCS as the main driver. The feasibility and impacts of such demand in the context of regional socio-economic conditions and renewable energy potentials has so far not been covered extensively in IAM research.

12 Technology readiness level

Technology readiness level (TRL) is a stage-based framework used to describe how far a technology has progressed from basic concept to operational deployment.[790] In practice, TRL assessments ask whether a method has only been theorized, validated in the laboratory, tested in relevant environments, demonstrated at pilot scale, or proven under real operating conditions. For CDR, TRL matters because lower-readiness methods generally carry greater uncertainty in cost, performance, scalability, environmental effects, and governance needs;[3,790] this is especially important where policy, procurement, or carbon-crediting systems may move faster than the underlying evidence base. Technology readiness should not be interpreted in isolation: methods at similar nominal TRLs can differ strongly in supply-chain maturity, MRV feasibility, and regulatory readiness.

Enhanced Weathering is generally characterized as an early- to mid-stage CDR method overall, most often around TRL 3–4 (out of 9, including by the IPCC), with one synthesis placing it at TRL 4–5.[3,541,583,790] Technology readiness of EW is typically judged to be below afforestation/reforestation, BECCS, DACCS, and biochar in overall deployment maturity, while still ahead of some ocean-based approaches.[3,541,790] This overall ranking is consistent with the fact that EW has been demonstrated in laboratory studies and small field trials, but not yet through broadly replicated, transparently monitored, large-scale deployment. The same attached assessments also suggest that EW compares favourably to some other novel CDR options in terms of conceptual simplicity and use of familiar industrial processes, but less favourably in terms of proof of durable, quantified net removals under routine operating conditions.

A useful distinction when evaluating the readiness of EW lies between the maturity of its physical supply chain and that of its carbon accounting and monitoring systems. The infrastructure required to implement EW is largely derived from well-established industrial and agricultural practices. Core operational steps—including quarrying, crushing, mineral processing, transport, and field spreading—closely resemble activities already performed at large scale in the mining, aggregate, and agricultural liming sectors rather than requiring entirely new technological systems.[19,106,118,538,582] In many regions, existing liming equipment, established logistics networks, and proximity between rock resources and agricultural land could enable relatively rapid operational scaling where favourable geographic conditions and rock sources exist. At the same time, regional constraints such as transportation capacity, supply-chain coordination, and infrastructure availability may still limit deployment in some contexts (see also Section 8).[17,428] Overall, these observations suggest that the physical processes involved in producing, transporting, and applying rock materials are not the principal factors limiting the overall TRL of EW.

Instead, important readiness challenges currently arise in the domain of MRV as well as the limited availability of verified field data. Recent assessments highlight that robust methods for quantifying CDR from EW remain under development, and that standardised accounting approaches suitable for large-scale implementation are still emerging (see also Section 4).[325,383] More broadly, recent assessments emphasize that MRV approaches for EW are still being developed and tested, reflecting the complexity of linking mineral dissolution, carbon uptake, and long-term carbon storage across heterogeneous environmental settings.[345,505,541] While robust MRV systems are widely recognised as essential for establishing credibility and enabling the responsible deployment of CDR approaches,[541] MRV methodologies remain at different stages of development across

pathways. For EW, the main challenge lies in reliably quantifying net CO_2 removal under field conditions, where carbon uptake depends on coupled geochemical and hydrological processes that operate across heterogeneous soils, climates, and management regimes, limiting both conceptual clarity and availability of large-scale, verified EW projects.

Taken together, these considerations indicate that the TRL of EW is uneven across the deployment chain. While many supply-chain elements already exist and could in principle be mobilised at scale, key uncertainties remain in demonstrating durable, verifiable CDR under real-world conditions. Progress toward higher TRL will therefore depend less on further demonstration of basic engineering feasibility and more on large-scale field deployments, improved monitoring frameworks, and transparent, independent evaluation of net CDR outcomes. This distinction is important when comparing EW to other CDR approaches. Relative to methods such DACCS or BECCS, EW may rely less on entirely new capture hardware, but it currently faces greater challenges in directly quantifying and verifying removal outcomes in situ.

13 Policy frameworks for scaling EW

There is a general consensus that CDR as a climate solution requires policy support to scale[791–793] and that each CDR solution, including EW, requires additional context-specific assessment and support.[794,795] A holistic approach to EW policy will likely include i) improved estimates for global and local removal potential, ii) targets for EW embedded in larger climate policy measures, iii) appropriate and scalable quantification tools built on robust data sets, iv) updated environmental regulations to prevent harm (e.g. release and bioaccumulation of toxic trace metals) and v) large-scale subsidies for deployment where on-farm benefits (e.g. crop yield improvements) do not fully cover the cost of feedstock, spreading, and measurement.

Accurate estimates of CDR via EW must underpin policy decision-making. Existing literature provides global estimates of current and future EW potential under a range of climate scenarios, with varying degrees of geographic specificity.[19,75,245] More recent efforts have attempted to provide granular estimates of removal potential for specific pathways in specific jurisdictions,[796] though these efforts rely on simple assumptions about feedstock availability and have been hampered by limited field trial data on EW outcomes, from which one could extrapolate potentials for larger scale deployment. Efforts have also been made to improve integration of CDR into IAMs (cf., Section 11). Improved estimates are particularly important as policy discussions move from high-level multilateral institutions (e.g. IPCC assessment reports) to national, sub-national, and municipal governments focused on implementation.

Adoption of CDR technologies into policy is often driven by pre-existing climate laws and goals. The first step is often setting a net-zero target. According to the International Energy Agency, 105 national governments have made net-zero pledges, 30 of which are codified in law. An additional 61 countries have included net-zero pledges in non-binding policy documents such as national climate plans.[797] The second step in adopting CDR policy is defining residual emissions at the time of net-zero, which would be offset by removals. Recent reviews of long-term climate strategies suggest that residual emissions are poorly defined or entirely excluded from national strategies.[798,799] Where residual emissions are included in national strategies, they are often balanced by large amounts of land-sector removals, with limited use of novel CDR methods like EW.[800] Integration of novel removals into national strategies, using refined estimates of removal potential, will likely precede broad EW policy development and implementation. Recent work has highlighted the utility of separate targets for CDR, alongside targets for emissions reductions and conventional removals, to unlock public innovation support, reduce the risk of mitigation deterrence, and enable tracking of progress against multi-decadal goals.[541,788,801]

To accurately assess progress towards removal targets, including jurisdiction-specific EW targets, appropriate tools and standards are needed. Significant progress has been made in the quantification of EW in the last decade, as demonstrated by the rapid increase in academic publications and codified in carbon registry methodologies (cf., Section 4). Significant challenges remain given the inherent variability of soils, making it difficult and expensive to achieve the tonne-level precision often required for carbon crediting. It has been proposed that monitoring of EW in the context of large-scale agricultural policies, which would in turn be reported in national GHG inventories, need not require this level of precision. In this context, the ongoing IPCC 2027 Methodology Report on Carbon Dioxide Removal Technologies, Carbon Capture, Utilization, and Storage for National GHG Inventories is particularly relevant. The report is intended to provide

additional methodological guidance for estimating and reporting CDR and CCUS in national GHG inventories, including where existing IPCC Guidelines contain gaps or outdated guidance for emerging removal approaches. If EW is included, this process could provide an important basis for how countries account for EW removals in national inventories, although the final guidance has not yet been adopted.

To support such national-level GHG accounting, landscape-level estimates using soil and watershed data may be sufficient, in alignment with reporting requirements for other activities in the agriculture and land-use sector and with emerging proposals for aggregated monitoring.[802,803] These efforts can leverage existing soil data collection efforts, such as national soil health cards, and riverine water-quality monitoring systems, such as the US Geological Survey stream-gaging network. Regardless, in the near term, public support for research and development is needed to develop sufficiently robust and transparent datasets to assess CDR via EW across a range of soil types, climates, hydrological conditions, feedstocks, and crops to enable both carbon-market-based projects and non-market-based policies. The US, EU, UK, Germany, and Australia have publicly funded research programs on EW in 2026, but additional public support is needed.[21]

Recent efforts have highlighted the range of potential environmental impacts of EW on agricultural lands, both in soils and in downstream waters.[113,129–131,475] Particular emphasis has been placed on the potential for heavy metal accumulation in soil, with possible health impacts on animals and humans.[129] Alongside continued research on potential environmental impacts, updated environmental regulations are needed to ensure that large-scale EW deployments do not contribute to environmental harms and, ultimately, environmental injustice. In addition to regulatory limits, third-party certification and pre-deployment feedstock screening may help mitigate toxicity concerns by requiring material characterization, traceability, and assessment of potentially toxic elements before agricultural application. This is particularly important as an increasingly wide range of industrial waste materials have been proposed as potential feedstocks for agricultural EW.[410,415,478,582,804–807]

Near-term policy development for CDR, including EW, largely relies on the logic and language of carbon markets, drawing on both existing compliance market infrastructure (e.g. the European Union Emissions Trading Scheme) and activity in the emerging SOM for novel types of permanent CDR. For example, Canada and the US have proposed directly purchasing CDR credits with public funds to spur growth in the CDR market and the EU is in the process of developing a voluntary CDR credit certification scheme and registry.[21] A recent expert survey suggests that EW companies expect to rely on compliance markets for revenue through 2050.[808] However, recent meta-reviews of climate policies suggest that compliance markets are important but insufficient to achieve climate targets—a mix of policy mechanisms is needed.[793,809]. Enhanced Weathering, in particular, could benefit from integration into large-scale agricultural subsidies. Agricultural subsidies are popular and politically resilient around the world, totalling over 400B USD annually.[810] Carbonate-based EW (liming) is already subsidized in many countries and sub-national jurisdictions[802] and efforts have been made to incorporate silicate weathering into the same policies.[811]

Enhanced Weathering, perhaps uniquely among novel CDR solutions, also presents opportunities to tap into a range of promising policy mechanisms, including but not limited to landscape-level

monitoring infrastructure[280] (e.g. water quality programs) and agricultural subsidy regimes (e.g. liming subsidies).

14 Conclusion and summary

There is now substantial and converging evidence that EW can deliver meaningful CDR while simultaneously generating a range of co-benefits, particularly in agricultural and agroforestry systems. Across laboratory experiments, field trials, and modelling studies, EW has been shown to increase alkalinity generation, enhance carbon storage, and, in many cases, improve soil fertility and crop performance. Yet, despite this growing body of evidence, key quantitative metrics remain highly uncertain. Reported area-normalized CDR fluxes, regional and global potentials, and cost estimates frequently span orders of magnitude. This spread does not solely reflect unresolved scientific uncertainty or methodological limitations. Rather, it also captures the inherently context-dependent nature of EW: its performance, costs, and co-benefits vary systematically across climatic conditions, soil properties, feedstock characteristics, application strategies, and monitoring approaches. As a result, EW should not be understood as a single, uniform CDR pathway, but as a spectrum of system-specific interventions whose outcomes depend on the interplay of environmental, geochemical, economical, and social factors. These context dependencies have direct implications for policy design, target setting, and implementation strategies, which are explored in more detail in Section 13.

14.1 Research landscape and harmonisation challenges

The current evidence base for EW is both rapidly expanding and highly heterogeneous, comprising 241 peer-reviewed studies published between 1995 and April 2025, with particularly rapid growth since 2020. This literature spans a wide range of research approaches, including experimental laboratory and mesocosm studies (23%), modelling studies (21%), field experiments (10%), and a substantial share of reviews and perspective papers, reflecting both active knowledge generation and ongoing synthesis efforts. With respect to content, the majority of studies (65%) focus on quantifying CDR through weathering rates, alkalinity generation, and carbon budgets, while environmental impacts, agronomic effects, and techno-economic considerations are investigated less frequently. Across this diverse body of work, a range of feedstocks are investigated, with a clear dominance of mafic ($n = 86$, including basalt) and ultramafic ($n = 53$ including olivine/dunite) feedstocks, with smaller contributions from wollastonite, industrial by-products, and other materials. This diversity reflects both the flexibility of EW as a concept and the wide range of environmental and operational contexts in which it is being explored.

At the same time, this breadth introduces a set of persistent challenges that currently limit comparability and quantitative synthesis across studies. A central issue is the lack of consistent feedstock characterization,[132] with many studies reporting insufficient or non-standardised information on composition, classification, particle size, or surface area, despite these being key controls on weathering rates. Even where such parameters are reported, they are described in highly inconsistent ways. For example, particle size distributions are reported using an exceptionally wide range of metrics and conventions, with more than 80 distinct units, measures, or reporting approaches identified across the literature. These span fundamentally different representation types, including mean or percentile diameters, cumulative mass fractions at arbitrary thresholds, and discrete size-class bins defined using a wide variety of sieve ranges. As a result, studies that ostensibly investigate similar materials often cannot be directly compared.

More broadly, key experimental parameters are often reported inconsistently or incompletely, further limiting comparability across studies. A recurring issue is the reporting of application amounts and sometimes CDR data: particularly in laboratory and mesocosm experiments, feedstock additions or CDR findings are frequently expressed only as mass per treatment (e.g., kg per pot) without sufficient information on experimental footprint, making it difficult to derive area-normalized application or CDR rates necessary for comparison. In other cases, applied volumes are reported without corresponding material densities, or repeated applications are described without clearly specifying frequency, duration, or cumulative totals.

Similar limitations arise in the reporting of hydrological conditions. In many studies, not all components of the water balance are documented, and water inputs are often described only partially, for example as watering regimes intended to maintain constant soil moisture, or as volumes per treatment, without reporting total water fluxes over time or lacking experiment dimensions necessary for conversion. This makes it difficult to compare hydrological forcing across experiments or relate results to field-relevant conditions. Where water inputs are reported, they are rarely expressed in standardised, scale-independent units such as depth-equivalent fluxes (e.g., mm or mm yr^{-1}), which would provide a more consistent basis for cross-study comparison. Furthermore, the temporal structure of water inputs is seldom resolved: event frequency, intensity, and intermittency are rarely documented, despite their strong influence on infiltration dynamics, residence times, and geochemical reactions.[252,371]

Such reporting gaps apply to initial soil properties, which are essential for interpreting EW outcomes and explaining variation across studies. Soil texture, pH, CEC, BS, SOC/SIC content, bulk density, and initial exchangeable cation pools strongly influence mineral dissolution, cation retention, alkalinity export, agronomic responses, and SOC dynamics. Yet these parameters are not consistently reported, are often measured at different depths, or are described using non-standard units and methods. This limits the ability to identify drivers of variability across experiments and constrains more detailed meta-analyses aimed at disentangling the effects of feedstock, soil, climate, hydrology, and management. More consistent reporting of baseline soil properties, including sampling depth and analytical methods, would therefore substantially improve the interpretability and comparability of EW studies.

A further harmonisation challenge is that some studies report CDR estimates that combine inorganic EW-derived CDR with changes in SOC stocks. This can make it difficult, and in some cases impossible, to distinguish the contribution of EW from changes in SOC or biomass. Such combination is problematic because inorganic and organic carbon pools differ fundamentally in formation pathways, monitoring approaches, storage timescales, durability, and reversal risks. We therefore recommend that inorganic CDR from EW and organic carbon responses should always be reported separately, even where both are included in a broader net climate-impact assessment.

Together, these issues mirror the inconsistencies observed for grain size and feedstock characterization and point to a broader lack of standardisation in how key process variables and CDR accounting boundaries are documented. Addressing these gaps through clearer and more consistent reporting standards will be essential for enabling robust cross-study comparisons, improving mechanistic understanding, and constraining the CDR potential of EW.

14.2 Quantifying CDR potential

A rapidly growing body of MRV approaches has emerged to quantify CDR from EW, spanning solid, aqueous, and gaseous phases, as well as isotopic tracers.[325,383] These approaches differ substantially in their system boundaries and the loss processes they account for, which makes direct comparison across studies or approaches challenging. At the same time, this diversity reflects the fundamentally diverse nature of field conditions EW operates in and provides a valuable opportunity to interrogate different parts of the CDR and loss pathway with targeted methods. Rather than converging on a single "best" MRV approach, the field is likely to evolve toward a suite of complementary methods, each optimized for specific components of the initial CDR and subsequent loss and transport cascade as well as deployment contexts.

Across the empirical and modelling literature, area-normalized CDR fluxes from EW span several orders of magnitude, reflecting both true system variability and differences in study design. Across 511 compiled flux estimates from 61 studies, the median flux is 0.84 $tCO_2\ ha^{-1}\ yr^{-1}$ but estimates vary over multiple orders of magnitude. Across experimental studies ~84% of fluxes are positive, ~34% are <0.1 $tCO_2\ ha^{-1}\ yr^{-1}$, 25% are 0.1-1 $tCO_2\ ha^{-1}\ yr^{-1}$, and 33% are between 1-10 $tCO_2\ ha^{-1}\ yr^{-1}$. However, many of the larger estimates do not fully account for loss processes that reduce net CDR after initial weathering occurs and/or reflect conditions that would be unrealistic to achieve in the field. Importantly, these fluxes overlap with—but are often lower than—values assumed in large-scale projections of CDR potential[19,24,245] even though not all estimates fully account for loss processes or transient early dissolution pulses. Nevertheless, emerging evidence increasingly shows that EW can deliver measurable CDR and clear co-benefits in some settings, while producing limited or negligible CDR and agricultural benefits in others. Enhanced Weathering should therefore no longer be framed simply as a question of whether it can work in the field, but rather where, when, and how it can be made to work most effectively. The field should be—and increasingly is [19,24,93,245]—pivoting toward identifying the environmental, mineralogical, agronomic, and management conditions that govern EW outcomes, and toward optimizing deployments to maximize durable CDR alongside agronomic and SOC/SOM co-benefits. Importantly, agronomic benefits provide a major opportunity to align climate mitigation with farmer incentives, offering a pathway for scaling EW that is not dependent on CDR value alone.

When scaled globally, the literature is beginning to converge on a maximum technical CDR (c.f., Section 5.2) potential in the range of ~0.2–2 $GtCO_2\ yr^{-1}$. Empirically constrained approaches suggest lower-bound estimates of ~0.22–0.24 $GtCO_2\ yr^{-1}$ [14,240], while process-based models project ~0.5–2 $GtCO_2\ yr^{-1}$ and cumulative potentials of 64-217 $GtCO_2$ over 75 years, depending on assumptions regarding application rates, land availability, and model implementation.[19,24,245] At the same time, emerging evidence on transient and long-term losses of initial CDR, e.g. via formation of secondary phases, strong acid weathering, or cation exchange,[68,172,216,325,439,445] is not yet consistently incorporated across models. In addition, many assessments assume sustained large-scale deployment without accounting for resource limitations and cumulative application limits. This is particularly evident in earlier landmark assessments deriving higher potentials,[17,23] which provided important first-order estimates under simplified assumptions but were developed prior to the current understanding of key feedbacks and loss processes in the EW context. Together, these factors suggest that even current "best estimates" of maximum technical potentials may remain optimistic relative to fully constrained system behaviour.

The representation of EW in IAMs remains limited but is expanding, with only a small number of IAMs currently including EW (e.g., REMIND, GCAM, GET, EPPA). Across these models, EW is typically implemented using simplified and harmonised assumptions, often drawing on a small set of foundational studies.[18,19,23] Deployment is generally constrained by land availability, particularly cropland, while other factors such as feedstock availability, supply chain capacity, and some geochemical feedbacks are represented in a more aggregated or implicit manner. As a result, IAM-derived deployment levels—commonly reaching several gigatonnes of CO_2 removal per year in high-ambition scenarios that sometimes extend beyond agricultural lands—tend to reflect upper-bound estimates under idealized assumptions, rather than fully constrained real-world outcomes. At the same time, IAMs consistently identify EW as an intermediate-cost, intermediate-potential CDR option whose deployment increases when other CDR pathways are limited, highlighting its potential role within diversified mitigation portfolios. Continued integration of emerging empirical insights on weathering dynamics, resource constraints, and system-level interactions will further improve the representation of EW in IAM frameworks and help refine projections of its long-term contribution, and this represents critically important future work.

Activity in the VCM provides an early indication of how EW is transitioning from research to deployment, albeit at modest scale. As of early 2026, cumulative EW credit sales amount to ~0.8 $MtCO_2$, representing approximately 1–2% of total CDR credits sold since 2022 (based on CDR.fyi data). With its rapid scaling, the VCM it is already functioning as an important testbed for EW implementation, providing early insights into operational feasibility and MRV development, and can be an important tool for learning where EW does and does not work—provided data is shared transparently.[530]

14.3 Technology readiness, costs, and LCA

Enhanced Weathering is generally classified as an early- to mid-stage CDR approach, with overall readiness around TRL 3–5.[3,541,583,790] While core supply-chain components such as mining, comminution, transport, and spreading are based on mature and widely deployed technologies, a primary bottleneck lies in demonstrating durable, quantifiable CDR under real-world conditions. In particular, MRV frameworks and large-scale field validation remain underdeveloped, and few deployments have yet achieved robust, transparent verification of net removal. As a result, advancing EW readiness will depend less on engineering innovation and more on scaling field deployments, improving monitoring systems, and generating reproducible, independently verifiable evidence.

Across 25 studies (166 datapoints), EW CDR costs span more than an order of magnitude, with harmonised central estimates ranging from approximately USD 34 to more than USD 1,000 tCO_2^{-1} (inflation adjusted to 2024 USD) and a median around USD 190–200 tCO_2^{-1}. While favourable cases exist, the literature broadly supports a central tendency of roughly a few hundred dollars per tonne. These estimates are broadly consistent with current prices on the VCM (CDR.fyi data).[529] Cost variability reflects both real differences in deployment configurations and inconsistent assumptions regarding CDR efficiency and system boundaries. In the academic literature, costs are found to be primarily driven by grinding, transport, and application logistics, while suppliers report that MRV costs contribute on the order of USD 15–70 tCO_2^{-1} in early-stage deployments.[577]

Life cycle assessments indicate that EW is a materially and energetically intensive CDR pathway, with upstream emissions ranging from ~1 to >200 kg CO_2-eq per tonne of rock applied and energy demands of ~600–1,400 MJ t^{-1}. Nevertheless, unlike many other novel CDR approaches, EW does not inherently depend on large dedicated renewable energy inputs, although access to low-carbon energy can substantially improve its life-cycle emissions. Grinding and transport typically dominate emissions, together accounting for ~80% of total life cycle impacts on average. These upstream burdens can substantially reduce—or in some scenarios eliminate—net CDR, highlighting that EW performance depends critically on logistics, particle size optimization, and energy sources. At the same time, most studies do not comprehensively account for life cycle emissions, indicating that net CDR estimates remain incompletely constrained.

At scale, EW deployment based on basaltic or other primary rock feedstocks is unlikely to be supplied solely from existing waste streams or currently available surplus material. While some industrial alkaline residues may provide locally important near-term opportunities, large-scale basalt-based EW would likely require substantial expansion of quarrying, mining, comminution, and associated transport infrastructure. Consequently, LCA and cost studies that assume abundant low-cost waste or by-product feedstocks may not provide sufficient constraints for assessing the costs, emissions, permitting requirements, and deployment timelines of primary-rock EW at scale. In this context, limited data on suitable feedstock reserves, current and expandable production capacity, quarry development, permitting processes, and the time required to bring new extraction capacity online may represent a major bottleneck for EW scalability in the mid-term.

14.4. Co-benefits

Across the literature, EW is consistently associated with positive agronomic outcomes, particularly in terms of crop yields, nutrient use efficiency, and resilience. Evidence from both EW-focused studies and a broader body of agronomic rock-flour applications[47] indicates that yield responses are overwhelmingly positive, with relatively few negative or neutral outcomes reported: out of 166 studies, 129 report positive and 26 neutral responses, with only 11 negative responses. These benefits are commonly attributed to the supply of base cations (e.g., Ca, Mg, K), improved soil pH, and enhanced nutrient availability, particularly in acidic or nutrient-limited soils.[47] In some cases, yield increases translate directly into economic benefits for farmers, reinforcing the potential for co-benefits to offset deployment costs, offering an alternative avenue for scaling.[556,812]

Responses of SOC to EW are variable and less well constrained, but emerging evidence suggests that they may be substantial. Reported SOC responses range from negative to positive, with increases often linked to enhanced plant productivity, greater biomass inputs, and changes in microbial activity and nutrient cycling.[19,47,94,101,439] At the same time, EW can stimulate SOM decomposition through increased pH, nutrient availability, and microbial activity, leading in some cases to transient increases in CO_2 efflux,[112,114] although such responses are not consistently observed.[91,224] Over longer timescales, EW may promote SOC stabilisation through MAOM formation and aggregation, mediated by weathering-derived cations and secondary minerals.[92,526,527,657,658] While these processes point to the potential for significant organic carbon gains, losses of SOC could partially or fully offset inorganic carbon sequestration, and in extreme cases turn EW systems into net carbon sources.[113,172,325] Evidence from literature on agricultural

liming suggests that initial pulses of SOC mineralisation can occur following mineral amendments but are often counterbalanced by longer-term increases in plant productivity and organic carbon accrual.[42,43,659] Whether similar trajectories hold for EW remains insufficiently constrained and likely context dependent. This highlights the need for careful, long-term monitoring of SOC responses to ensure that EW delivers net climate benefits and does not inadvertently undermine them. In this context, parallel monitoring of inorganic EW CDR and soil SOC dynamics becomes essential, underscoring the potential value of co-application and co-crediting frameworks on the same land.

14.5 Synergies with other CDR approaches

Enhanced Weathering is rarely deployed in isolation and instead operates within managed land systems where multiple CDR processes co-occur, creating opportunities for synergistic interactions. As a geochemical CDR strategy that stores carbon predominantly in stable inorganic reservoirs, EW occupies a distinctive position within the broader CDR portfolio by combining durable carbon storage with natural weathering reactions embedded in managed land systems and associated agronomic co-benefits.[7,15,19] By supplying base cations and nutrients (e.g., Ca, Mg, K, Si) and buffering soil acidity, EW can enhance plant productivity, rooting depth, and microbial activity, thereby increasing biomass inputs and soil carbon formation.[47,64,561] At the same time, because it can be implemented on existing agricultural lands without requiring dedicated biomass production, EW can complement rather than compete with biomass-based CDR approaches, enabling "stacking" of inorganic and organic carbon storage within the same system.[7,813]

Beyond additive effects, several mechanisms suggest the potential for true synergies between EW and regenerative or biomass-based approaches. Organic amendments and increased root respiration can elevate soil CO_2 concentrations and organic acid production, accelerating mineral dissolution, while weathering-derived cations can promote stabilisation of SOC through MAOM formation and aggregation.[92,439,526,527,657,658] In turn, improved soil structure, water retention, and nutrient cycling can sustain both biological productivity and weathering reactions over longer timescales.[76,676] These agronomic benefits can also translate into increased biomass production, potentially reinforcing biomass-based CDR pathways such as BECCS or BiCRS generally. Together, these interactions can enhance total CDR per unit land while increasing system resilience. However, such synergies remain highly context dependent and introduce additional system complexity, particularly with respect to nutrient cycling, carbon partitioning, and attribution of CDR across pathways. Realizing their full potential will therefore require integrated system design and long-term evaluation of coupled inorganic and organic carbon dynamics.

14.6 Socio-economic impacts, public perception, and governance considerations

Socio-economic factors are central to the feasibility of EW deployment. Evidence points to potential economic co-benefits, particularly in agriculture, where improved nutrient supply and soil conditions can increase yields and reduce fertiliser demand,[180,444,556] with reported revenue gains of approximately USD 300 ha^{-1} in smallholder systems. At larger scales, EW could generate employment across mining and agricultural sectors (~60,000–80,000 jobs under high deployment

scenarios in the US alone; ref. [22]) and contribute to multiple SDGs, although outcomes are highly context dependent.[84,560]

These benefits are offset by important distributional and environmental challenges. High-potential deployment regions are concentrated in the Global South,[19,24,245] raising concerns about unequal burden-sharing, extractive pressures, and "sacrifice zones" linked to mining.[560,565,583] Economic gains may also be unevenly distributed across value chains.[422] In addition, environmental and health risks—particularly dust exposure and potential heavy metal mobilisation—require careful monitoring and management.[129,131,209,426]

Public support remains conditional and linked to low awareness,[573,765] with greater acceptance for research than large-scale deployment and key concerns focused on mining impacts, fairness, and governance. Compared to other CDR approaches, support for EW is generally lower than for more familiar land-based biological options such as afforestation or soil carbon sequestration, but often comparable to or higher than support for DACCS, reflecting EW's combination of engineered deployment, geochemical CDR, and links to managed land systems. Support increases with trust, knowledge, and opportunities for engagement.[555,574,775] Overall, deployment will depend on aligning economic incentives with transparent governance, stakeholder engagement, and equitable distribution of risks and benefits.

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17 Conflict of interests

JEL maintains un-exercised options in the carbon registry Isometric awarded during his employment there from 2023-2024. MD currently works for UNDO Carbon, but at the time of contribution to this systematic review held no conflicts of interest. RBN has served as a paid reviewer of project applications submitted to Frontier Climate for ERW funding, and as a paid technical consultant for Cascade Climate, Mati Carbon, and Crew Carbon. RBN also collaborates on an EW research project with UNDO Carbon, which has provided a research gift to support this work. PAR, CTR, and NJP serve as scientific advisors to CREW Carbon, a company that generates alkalinity in wastewater treatment plants using lime addition but does not work on terrestrial Enhanced Weathering in agricultural fields. MEV offers consulting services related to the general subject area of this research, but these consulting activities are not directly connected to the work presented in this manuscript. NJP and CTR were co-founders of the EW supplier Lithos Carbon but have no financial ties to the company. All other authors declare no competing interests.

18 Author contributions

Conceptualization: TJS, SFu
Data Curation: TJS, CD, TK, AL, TRee, JT, MBB, LAB, RDA, IF, PF, RG, NI, JEL, YL, BR, JMS, ES, SSET, RVDB, SZ, SKA, JSC, IC, ID, MD, XD, SFo, MG, KH, CK, AKl, JK, TL, SLi, SRM, EM, LN, HN, SP, EP, KR, RAR, MR, SWS, LS, ILS, TS, FS, XS, WT, LTL, AV, JW, BW
Formal analysis: TJS, CD, TK, AL, TRee, JT, MBB, IF, PF, NI, YL, SSET, RVDB
Funding acquisition: TJS
Investigation: TJS, CD, TK, AL, TRee, JT, LAB, JEL, JK, SWS, TS, XS, DACM, JRHM, YY, BZ
Methodology: TJS, CD, TK, AL, TRee, CF, SLü, TRep, SV, SFu
Software: TJS, BR, JMS, SSET, TRep
Supervision: TJS, CD, TK, AL, TRee, JT, MA, RHJ
Validation: TJS, CD, TK, AL, TRee, JT, RG, NI, BR, JMS, ES
Visualization: TJS, JK, SLB, CF, YK, DACM, JRHM, PR, JDS, YY, BZ, CTR
Writing-original draft: TJS, CD, TK, AL, TRee, JT, MA, MBB, LAB, RG, JEL, RVDB, SZ, IC, XD, JK, TL, SLi, EM, SWS, TS, FS, CMB, SLB, SC, TD, CF, MH, JH, IOH, BZH, KJ, AKh, CRL, DACM, JRHM, JMZD, RBN, PAEPVS, IMP, PAR, PR, JS, RS, JDS, SV, MEV, RMW, YY, BZ, CTR, NJP
Writing - review & editing: TJS, CD, TRee, MA, JK, EM, SP, SWS, CMB, SLB, SC, TC, CF, RHJ, YK, SLü, DACM, JRHM, RBN, CRP, PAEPVS, TRep, LT, MEV, YY, SFu

19 Data availability statement

The data used in this systematic review were compiled as part of a broader database effort on EW. The full database is still being finalized, and additional data not used in the present review are undergoing quality control. Because the present review uses only a limited subset of the wider database, we have not separated this subset from the broader data product, as doing so would fragment a database intended to be published as a single, coherent resource. The complete database will be made publicly available alongside a dedicated database publication once final quality control has been completed. Until then, the data underlying the analyses reported in this review are available from the corresponding author upon reasonable request.

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Supplementary material to

An Ecosystem of Carbon Dioxide Removal Reviews – Part 3: Enhanced Weathering

T.J. Suhrhoff[1,2*], C. Dietzen[3], T. Kukla [4], A. Lunstrum[5], T. Reershemius[6], J. Thompson[2], M. Almaraz[1], M. Bertagni[7], L. Bullock[8], R. D'Ascanio[2], I. Falkson[2], P. Frings[9], R. Gregg[2], N. Iff[10], J. Lambert[11], Y. Li[12], B. Rogers[13], J.M. Schneider[13], E. Swanson[2], F. Tao[14], S.S. Tsao[15], R. Van Der Bauwhede[16], S. Zhang[12], S.K. Anand[17], J.S. Campbell[18], I. Chiaravalloti[2], I. Davis[19], M. Dobson[19], X. Dupla[20], S. Foteinis[18], M. Guo[21], K. Harrington[22,18], C. Kent[2], A. Klemme[23], J. Kroeger[15], T. Linke[24], S. Linnekogel[25], S. Moller[1], E. Milliken[2], L. Ndlovu[26], H. Niron[27], S. Patel[26], E. Pihlap[28], K. Rees[29,30], R. Rioux[15], M. Rizzi[3], S. Shaheen[15], L. Steinwidder[27], I. Steeley[29], T. Sweere[31], F. Sun[15], X. Sun[12], W. Tatge[15], L. Tejedal Lemus[13], A. Vienne[27], J. Westphalen[32], B. Woollen[33], C.M. Baum[34], S.L. Brantley[35], S. Calabrese[17], T. Cyronak[32], T. Dorndorf[36,37], C. Fyson[36], M. Hagens[38], J. Hartmann[24], I.O. Holzer[39], B.Z. Houlton[40], R.H. James[19], K. Jensen[41], Y. Kanzaki[42], A. Khan[33], C. Levy[43], S. Lück[36], D.A.C. Manning[6], J.R.H. Manning[44], J. Meyer zu Drewer[45,46], R.B. Neumann[47], C.R. Pearce[48], P.A.E. Pogge von Strandmann[49], I.M. Power[21], P. Raymond[15,1], P. Renforth[18], T. Repke[36], J. Saiers[15], R. Santos[50], J.D. Smolen[51,52], L. Tan[2], S. Vicca[27], M.E. Vorrath[24], R.M. Webb[53], Y. Yao[15], B. Zhang[15], C.T. Reinhard[42], N.J. Planavsky[2,1], S. Fuss[36,54]

[1] Yale Center for Natural Carbon Capture, Yale University, New Haven, CT, 06511, USA
[2] Department of Earth and Planetary Sciences, Yale University, New Haven, CT, 06511, USA
[3] Globe Institute, University of Copenhagen, Copenhagen, Denmark
[4] Carbonplan, San Francisco, CA, USA
[5] School of Engineering and Applied Science, University of Pennsylvania, Philadelphia, PA, 19104, USA
[6] School of Natural and Environmental Sciences, Newcastle University, Newcastle upon Tyne, England, NE1 7RU, UK
[7] Department of Environment, Land and Infrastructure Engineering, Polytechnic University of Turin, Turin, Italy
[8] Department of Geological Resources for the Ecological Transition, Spanish Geological and Mining Institute, Spanish National Research Council, Murcia, Spain
[9] GFZ German Research Centre for Geosciences, Organic and Earth Surface Geochemistry, Telegrafenberg, Potsdam, 14473, Germany
[10] Department of Earth Sciences, Freie University Berlin, Berlin, Germany

[11] NYU Gallatin School of Individualized Study, New York, NY, USA
[12] Department of Oceanography, Texas A&M University, College Station, TX, 77843, USA
[13] Department of Earth System Science, Stanford University, Stanford, CA, USA
[14] Department of Informatics and Intelligent Systems, Institute of Energy and the Environment, The Pennsylvania State University, University Park, PA, USA
[15] School of the Environment, Yale University, New Haven, CT, 06511, USA
[16] Department of Earth and Environmental Sciences, Division of Soil and Water Management, KU Leuven, Leuven, Belgium
[17] Department of Biological and Agricultural Engineering, Texas A&M University, College Station, TX, 77843, USA
[18] Research Centre for Carbon Solutions, Heriot-Watt University, Edinburgh, EH14 4AS, UK
[19] School of Ocean and Earth Science, University of Southampton, Waterfront Campus, National Oceanography Centre, European Way, Southampton, SO14 3ZH, UK
[20] Department of Earth and Planetary Sciences, ETH Zurich, Zurich, Switzerland
[21] Trent School of the Environment, Trent University, Peterborough, Ontario, Canada
[22] Smith School of Enterprise and the Environment, University of Oxford, Oxfordshire, England, UK
[23] Institute of Environmental Physics, University of Bremen, Bremen, Germany
[24] Department of Earth System Sciences, University of Hamburg, Hamburg, 20146, Germany
[25] UK Centre for Ecology and Hydrology, Library Avenue, Bailrigg, Lancaster, LA1 4AP, UK
[26] Department of Chemical and Biomolecular Engineering, University of Pennsylvania, Philadelphia, PA, 19104, USA
[27] Biobased Sustainability Engineering (SUSTAIN), Department of Bioscience Engineering, University of Antwerp, Antwerp, Belgium
[28] Institute of Ecology and Earth Sciences, University of Tartu, Tartu, 50409, Estonia
[29] School of Biosciences, University of Sheffield, Sheffield, UK
[30] Leverhulme Centre for Climate Change Mitigation, University of Sheffield, Sheffield, United Kingdom
[31] Department of Earth and Planetary Sciences, Institute of Geochemistry and Petrology, ETH Zurich, Zurich, Switzerland
[32] School of Earth, Environment and Sustainability, Institute of Coastal Plain Sciences, Georgia Southern University, GA, 31419, USA
[33] Carbon Removal Standards Initiative, San Francisco, CA, 94114, USA
[34] Department of Business Development and Technology, Aarhus University, Aarhus, Denmark
[35] Department of Geosciences and Earth and Environmental Systems Institute, Pennsylvania State University, University Park, PA, USA
[36] Potsdam Institute for Climate Impact Research (PIK), Member of the Leibniz Association, Potsdam, 14482, Germany
[37] Geographic Institute, Humboldt University of Berlin, Berlin, Germany
[38] Soil Chemistry Group, Wageningen University and Research, Wageningen, 6700 AA, The Netherlands
[39] Environmental Studies Program, University of California, Santa Barbara, Santa Barbara, CA 93106, USA
[40] Ecology and Evolutionary Biology and The Ashley School of Global Development and the Environment, Cornell University, Ithaca, NY, USA
[41] Environmental Defense Fund, NY, USA
[42] Georgia Institute of Technology, Atlanta, GA, USA
[43] Science & Innovation, Carbon 180, Washington, DC, USA
[44] School of Chemical Engineering, University of Manchester, Manchester, M13 9PL, UK
[45] Ithaka Institute for Carbon Strategies, Goldbach, 63773, Germany
[46] Institute of Sustainable Energy Systems, Offenburg University, Offenburg, Germany
[47] Department of Civil and Environmental Engineering, University of Washington, Seattle, WA 98195, USA
[48] National Oceanography Centre, European Way, Southampton, SO14 3ZH, UK
[49] Mainz Isotope and Geochemistry Centre (MIGHTY), Institute of Geosciences, Johannes Gutenberg University Mainz, Mainz, 55128, Germany
[50] University of Guelph, Guelph, Canada
[51] Department of Earth and Planetary Sciences, Harvard University, Cambridge, MA, 02138, USA
[52] Department of Human Evolutionary Biology, Harvard University, Cambridge, MA, 02138, USA
[53] Sabin Center for Climate Change Law, Columbia Law School, New York, NY, 10017, USA
[54] Geography Department and IRI THESys, Humboldt University of Berlin, Berlin, Germany

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S1 Literature query

The following search query was used to retrieve candidate EW records from Web of Science and Scopus as the first step in the literature identification process. The query was designed to capture soil-based EW studies in the context of CDR, while excluding literature focused primarily on weathering and carbon cycling in the geological past. Retrieved records were subsequently filtered through an AI-based pre-selection step, manually screened against predefined inclusion and exclusion criteria, and supplemented with additional studies identified through expert knowledge and forward/backward snowballing where appropriate.

((TS= ((geoengineer*) AND (silicate OR olivine OR albite OR CACO3 OR liming)) OR
TS= ((silicate OR olivine OR albite OR CACO3 OR liming OR basalt*) AND ((("soil*" OR "cropland") AND ("CO2" AND (stor* OR sequestr* OR remov*))) OR (mitigat* NEAR/3 ("climate change" OR "global warming" OR geoengineer*)))) OR
TS= ((silicate OR olivine OR albite OR CACO3 OR liming OR basalt*OR mineral OR apatite* OR feldspar*) AND (agricultur* OR soil) AND (crush* NEAR/5 rock*)) OR
TS= ((("enhance*" OR "artificial*") NEAR/2 "weathering") AND ((carbon OR CO2 OR "climate change" OR "global warming") NEAR/3 (remov* OR sequest* OR storage OR sink OR mitigat* OR reduc*))) OR
(TS = (("enhance*" OR "artificial*" OR "chemical" OR "accelerate*" OR "mineral") NEAR/2 "weathering") AND TS = ((carbon OR CO2 OR "climate change" OR "global warming") NEAR/3 (remov* OR sequest* OR storage OR sink OR mitigat* OR reduc*))) AND
TS = (mineral* OR rock OR silicate)) OR
TS=((enhance* OR accelerate*)NEAR/4 ("carbon dioxide mineralization" OR "CO2 mineralization") AND (weathering)))
NOT TS = (glaci* OR ice* OR ordovic* OR Aptian OR Cenozo* OR Paleo* OR Mezoso*)

S2 Inclusion and exclusion criteria

The inclusion and exclusion criteria below were developed iteratively by the two reviewers who screened all abstracts retrieved through the literature search and automated classification process described in the Methods section of the main text.

S2.1 Inclusion

Include literature that is:

1. Specifically on *terrestrial* ERW in an soil context, include any type of paper, e.g. (but not limited to):
    a. Field trials and laboratory experiments
    b. Method development
    c. Modelling
    d. Natural analogues discussed in ERW context
    e. Climate justice, equity, etc.
    f. Reviews
    g. Policy/government
2. Also included are some types of non-peer reviewed publications such as:
    a. Editorials, comments, letter to the editor, etc.
    b. Doctoral (or MSc) dissertations
3. Include literature that uses **lab experiments** at ambient conditions that determine
    a. Mineral dissolution rates
    b. Weathering rates
    c. Carbonation rates if they explicitly link to ERW. This includes for example laboratory papers (or reviews thereof) that assess dissolution kinetics and discuss the implication of surface passivation for ERW.
4. That is about CDR in general and includes ERW to a significant degree. In this case **make sure to use the corresponding label.**
    a. Should be included if ERW makes up a significant part of the discussion/analysis.
    b. Should not be included if ERW is just mentioned in the introduction as one of many CDR options. Only include if ERW is actually discussed.
    c. If can't judge based on abstract include and it can still be excluded later
5. Mine tailings (or other waste feedstocks including but not limited to: cement, steel slack, etc.) literature *if it assesses/proposes the use of this feedstock on agricultural fields.*
6. Studies that assess downstream impacts in rivers and oceans, but *specifically in the context of ERW*.

S2.2 Exclusion

Exclude literature that is about:

1. The topic of the paper needs to be specifically on ERW in the context of CDR, papers that use ERW as a fertilizer are excluded (but labelled with a marker to identify later).
2. ERW in reactor designs, or to treat industrial flu gasses.
3. ERW in mine tailings or other feedstocks in their waste storage forms if not discussed in the context of applying them on agricultural fields.
4. "Ex-situ" carbonation of minerals/rocks outside of agricultural settings
5. OAE studies that do not specifically refer to terrestrial ERW as the source of alkalinity.
6. Papers that refer to natural rock weathering in streams, watersheds, soils, etc. without an "enhanced component" (unless it is discussed as a clear natural ERW analogue).

Note: Papers classified as "rock flour as fertilizer" were retained in the broader study where relevant to the review scope. These papers, together with additional studies identified through expert knowledge and snowballing, were included to inform agronomic responses and side effects, but were not labelled as CDR-focused studies unless they explicitly discussed EW in a CDR context.

S3 Code book

The codebook below was developed for the broader EW database project, whose scope extends beyond the systematic review presented here. Most codebook categories are not used directly in the review; nevertheless, we provide the full codebook for transparency and completeness, as it reflects the instructions given to individual data extractors. We note that not all categories described here will necessarily be included in the final open-access database. Only variables that pass internal quality-control criteria will be released.

S3.0 Introductory notes

Please read this section carefully before you start filling in the database**.**

You might be asking yourself: What is a code book? Essentially, code book is the fancy name for the instructions on how to enter data into a database in a quantitative literature review. The code book is important to make sure that the database is filled in consistently between different contributors … and we are many. So please read this document carefully to make sure we have an internally consistent and accurate database.

S3.0.1 Input data types

The database is designed to capture as much information on the quickly growing field of Enhanced Rock Weathering (ERW) as possible, while minimizing the amount of required input. The number of categories required to capture the literature on ERW is quite large, but at the same time no paper will require filling in more than a few percent of existing categories. For each paper you review, please leave categories that do not apply to that paper empty when filling out the database**.** If a category applies to a paper, there are several types of data that you can enter for a given field. In general, all acceptable answer types to every category are listed at the top of the database file sent to you.

General data types are

1. Mixed binary: (leave blank) – no – yes/explanation.
   We distinguish between leaving an entry blank and inputting "no" to account for the difference between a study that did not address the category (leave blank) compared to one that did, but found a negative result (enter "no"). For example, if heavy metal accumulation was not addressed, that field is left blank. If it was addressed but found to be negligible, heavy metal accumulation should be set to "no".
   Crucially, any input in these fields that is not "no" will always count as yes.
   Because we want to capture as much information as possible; some of these ask you to not only put "yes", but to add additional information in a specific and concise way. For example, category "5.1; reactive transport model"; if this study is not based on reactive transport models leave blank, if it is, enter something like "yes/1D reactive transport model considering 0.5m of soil depth". In general, try to add less than ~10 words unless truly necessary

2. Choose from a set of given answers/choices:
   In this case, sometimes it is required to "list all that apply" – i.e., from a list of given possible answers all relevant ones should be listed. For one line of data, this should happen as a list in a single cell, separated by commas. Do not add additional lines just for this (unless it marks the difference between experimental conditions, see further below). Please make a habit of using copy and paste to transfer given answer options into your input to minimize typos! If typos are inserted into some of these categories – though we will try to check for this – your inputs may not be detected in some of the statistical tests we are doing!
3. Data values, i.e., measurement/model results: Leave blank if not applicable or enter the number value as reported.
   As a rule of thumb, if a number value is expected the options specified will include "value", and if a text value (see mixed binary) is expected it will say "specify" – but this might not be entirely consistent so please just put what is appropriate!

We are trying to capture as much information as possible. Many papers that contain model or experimental results may describe multiple experiments or conditions that we want to capture. This means that some papers may require more than one line in the database. The reason that we only want one experimental condition per line is that this will make it much easier to query and analyze the database. For more information on what defines an experimental condition and how to fill out the corresponding categories see section 1.2.
If a single study has several lines, each line should still contain all relevant (meta-) data, which means that e.g. the basic paper info or soil baseline or feedstock data – all data necessary to capture all information regarding this experimental condition - need to be copied into each row. The reason is that when people are querying the database, each line of data needs to make sense independently and contain all necessary data.

There is also a comment category at the end of each section where you can add any information relevant that the categories were not able to capture, of if there is any information necessary to clarify and of the input data provided.

Please note that for dissertations, meta-analysis and reviews, and book (chapters), it is not necessary to insert experimental or modelling data because they usually will still be published in, or rely on, peer reviewed articles. For these types of publications, still fill out the basic information and discussion sections.
In general, you do not need to input data that is cited from other sources in the study you are looking it. Only insert new data that was generated by the study itself. If the study uses literature data to come up with new estimates, report the new estimate – for example if a study looks at different cost estimates of ERW, and based on the available literature estimates a realistic price as well as a range, report these parameters but not all the initial literature data used for this new meta-assessment.
When in doubt, reach out to Jesper what to fill in for a specific publication

When inputting data into the database files, please only start below line 37 (the area marked red in the code book file). The reason is that the lines above contain possible answers for some categories, and if you start inputting data higher up in the file, we may erroneously detect some of the default

answer possibilities as your input. Alternatively, you could delete all lines below line 2 and look at sample answers in a different file.

S3.0.2 Units and conversions

With a few exceptions, we do not ask you to do any unit conversions but put in data in the native units used in the study. As a result, each data value input category is comprised of two columns – the first one to enter the data value, and the second column for the unit. Please make sure to fill in both as without the unit the data is likely not usable.

Sections 7 and 9 are designed to be pre- and post-deployment data pairs. Some studies may not report pre-and post-deployment data but only the difference between the two. In this case you can fill the difference between pre- and post-deployment data in section 9, while choosing the appropriate unit containing "delta" – e.g., delta % or delta ppm and leave section 7 blank. Do this only if the pre-and post-deployment data are not reported! If they are reported, pre- and post-deployment data is preferrable and the difference can simply be computed later (i.e., it does not need to be manually added even if it is reported).

Except some trivial metric conversions (e.g., sample depth in m to cm, etc.), only two data fields require you do convert given units. The first is the GPS location which needs to be put in in decimal degrees - the reason is that putting in data in degrees, minutes, seconds, etc., is difficult to do correct in excel. You may use this website to convert to decimal degrees if a given study should use a different format. Make sure to enter a minus in front of the coordinates if they refer to the western or southern hemisphere.
The second conversion we require you to do is converting precipitation and irrigation data to a water column or water column per time (e.g., mm or mm/yr) – the reason is that this is the only way water throughput can easily be compared between experiments. Some experiments provide data e.g. in L/pot, or watering to keep soil moisture constant, etc. – these units are not useful to compare studies, and converting to water column values afterwards is difficult because experimental geometry may sometimes be hidden in the SI, etc. which is why we ask you do pay particular attention to this parameter when entering data.

Apart from some straightforward unit conversions, we ask that you do not calculate missing data fields, even if there is enough information to derive them. For example, take an experiment that gives the carbon dioxide removal (CDR) estimate in t/ha, and the length of the experiment as, e.g., 2 months. With these values, a CDR flux could be calculated in t/ha/yr – and there is a related CDR flux category in the database. But, in this case, we ask that you leave this category blank because the authors did not present the CDR flux in t/ha/yr, and calculating it implicitly assumes their findings across 2 months would hold over a year. In general, avoiding such calculations ensures consistency across database entries and fidelity to the reported results. Please only fill in the information the paper reports without performing additional calculations (other than the few unit conversions defined above) by yourself.

We are obviously aiming to fill this database as accurately as possible. So, if you have questions on how to fill out a certain section, please get back to Jesper. Jesper and Tyler will also be available for questions regarding some of the more data-intensive papers. Please take your time when transferring number values and double check that they are correct. Some studies will be given to several people, and each person will review at least one study that is reviewed by multiple people. This is not done to control everyone or out of distrust, but we will be using this data-overlap to assess the reliability of the database in a quantitative way.

S3.0.3 How we capture reported uncertainties

Most studies that report data also report associated uncertainties. To the extent feasible, we would like to capture these uncertainty estimates in the database as this will be crucial information for many potential uses.

The challenge is that some papers report absolute uncertainties on measurements, some report relative uncertainties, and others only report ranges. While the first two could be converted (which would be a lot of work when filling the database), the latter needs two separate data fields for each measurement (min and max). Hence, if we just wanted to capture all possible reported data, this would require 4 additional columns for each data column – which would simply make the database file too large.

The way we decided to handle this is to supply separate files for uncertainties. So in addition to the main database file that you're given to populate the database (**ERW_literature_main_round_database.xlsx**), you will be supplied with 4 additional files:

1. **ERW_literature_database_ main_round_absSD:** Use this file to enter absolute uncertainties (1SD) should they be reported in the paper. Absolute uncertainties are usually reported in the same unit as the given data category – please do so also when filling in this uncertainty file!
   Note that if the primary unit is percent, e.g. 50% +/- 5%, then the 5% is the absolute uncertainty, which you would enter here – or enter as "0.1" of 10% (5/50) in the relative uncertainty file (next).
2. **ERW_literature_database_ main_round_relSD:** Use this file to enter relative uncertainties (1SD) should they be reported in the paper. Relative uncertainties are given as a fraction of measurement values – e.g. "0.05" or as %, e.g. 5%. Whether uncertainty is reported as a fraction or as percent needs to be indicated at the beginning of this file, and then needs to be uniform throughout the whole file.
3. **ERW_literature_database_ main_round_min:** Use this file to enter the minimum of the range of reported values.
4. **ERW_literature_database_ main_round_max:** Use this file to enter the maximum of the range of reported values.

These files have the same structure as the main database, but only contain the data fields where uncertainties are relevant. It is important that you insert the same rows with identical identifying information (database sections 1 and 2) into the uncertainty files – i.e., the data structure (i.e.

number and naming of different experiments/database rows) must be identical to the main database. Although for *relative* uncertainties the same SD % in percent of the total value is often applied to many measurements/experimental conditions, we ask you to still use the same data structure as in the main data file to allow for easy cross referencing. To ensure this, we recommend filling in the main database, copying the basic information section to the uncertainty file (i.e., sections 1 and 2 of the database), and only then filling in the respective uncertainties. Often, relative uncertainties will require much less work so if they should be given, we recommend you only report them. For example, if all Ca measurements have a 5% uncertainty, you can copy the value "0.05" or 5 into all rows (and indicate whether you are reporting fractions or percent in the beginning of the spreadsheet!), whereas in the absolute uncertainty file you would have to report the 5% as an absolute value for each measurement.

A note on units: Absolute values that are reported should be in the same unit as the primary data in the main database. Nevertheless, the uncertainty files also contain the unit columns – primarily to make it easy for you to structure data and copy past them in the same way without having to take into account that there no additional columns in between data columns. If the uncertainty you are adding is in the same unit as the primary data in the main database file, you may leave the unit columns blank. Only insert in the situation where they are not the same. Similarly, when reporting relative uncertainties, information is entered in the basic information section of the database (whether in % or in fraction) and does not need to be inserted into each unit category unless a certain category differs from the overall choice indicated for the whole database.

More information on the related categories can also be found in section 1.3.

S3.0.4 Data files and naming

When inserting data into the excel files, please use a new file for the data of each study. Do not combine several studies into one file. This is going to make merging the database easier and reduces the risk of data loss on your side.

When you are getting your paper assignments, each paper comes with an assigned number in one of the following formats (with ### being a 3 digit number):

- terERW-###
- AcCoe-###
- AcNPR-###
- NAc_rev-###
- NAc_data-###
- fert-###
- FertAcNPR-###

When naming the completed database files containing the data of a study, please use only this number as the name for the file name, for example: terERW-001.xlsx. Do not add additional author name or year information, as this will make merging the database more tedious.

Do the same for the data files used to report uncertainties but add the relevant indication what kind of uncertainty is reported, e.g. terERW-001_relSD.xlsx.

S3.0.5 Code book and database structure

The structure of this document follows the structure of the database so that the Table of Contents may be used as a general overview of the database structure.
The structure of this document can loosely be defined as follows:

- Basic and identifying information on the study (sections 1-4)
- Information relevant to modelling studies and (semi-)quantitative reviews or social science studies (section 5)
- Information relevant to the specific experiment (or model; section 6)
- Modelling and experimental studies: pre-amendment data (section 7)
- Modelling and experimental studies: ERW deployment data (section 8)
- Modelling and experimental studies: post-amendment data (section 9)
- Modelling and experimental studies: CDR calculations and details thereof (sections 10 & 11)
- All studies: main findings & discussions (section 12 & 13)

This document does not contain instructions for *all* categories, but only the ones that require explanation. Most categories (like soil composition data) are self-explanatory. Similarly, inputs that are repeated for several types of data are only explained once. Hence the explanation may not be in the section where you need help. In this case use the search function within this document to locate it in a different section.

Lastly, and most importantly, thank you very much for being a part of this project! This project is a lot of work and would not be possible alone or in a smaller team. Each of your inputs is valuable and highly valued! We think that the outcome of this project – both the publicly available database covering all publications (not just the data-containing ones) and the meta-analysis paper we want to write – will be super valuable resources for the field of ERW. Particularly as it grows quickly, and particularly if we manage to keep the database updated after publication.

S3.1 Basic information

Mandatory basic information to identify the paper and specific data. This section is required for every study.

S3.1.1 Paper information

All authors: Please insert a list of all authors into the field "all authors" using "Initials (with dots, no spaces) Last Name", e.g. T.J. Suhrhoff. Separate authors with "; ".
Do not add "and" even if it is only 2 authors; e.g., "T. Reershemius; T.J. Suhrhoff"

Hyperlink: Please enter the DOI hyperlink. You can do this by adding "https://doi.org/" in front of the doi. If the entry has no DOI, make sure there is a link that works.

First author location: enter the location of the first affiliation of the first author as precisely as possible.

Peer-reviewed: Choose which of the suggested peer-review categories best describes a given study.

Keywords: Most articles provide keywords – copy all that apply, separate using a ";"

DOI/ISSN/ISBN: Always enter DOI if it exists; otherwise use anything else that is standardized. If not DOI, add the qualifier before the value (e.g.: ISSN 0257–8050)

is ERW the primary and only focus?: If ERW is the dedicated focus of this paper put "yes", if ERW is considered as part of a general CDR review, or if the study only looks as rock flour as a fertilizer without discussing CDR, put "no" – so this includes the body of older literature that looks at rock flour amendments to fertilize fields as well as general CDR assessments.

Coder: Put your own name.

Database paper number: Each study comes with a pre-assigned number (terERW-###, AcCoe-###, AcNPR-###, NAc_rev-###, NAc_data-###, fert-###, FertAcNPR-###) – please record this number here. This is very important as I plan to use this data categpry to merge the final database.

S3.1.2 Unique data and experiment identification

This section is really important such that every paper and experimental condition can be uniquely identified.

The first two categories are mandatory for every paper**.**

Unique data identifier: Enter first author, year, journal in this format: "Name_journal_year", e.g.: "Suhrhoff_FC_2024".
The database entries you receive should already have the journal in an abbreviated format, so just use abbreviation as suggested.
Note, that we will go through the database in the end and add "a, b,.." as appropriate if a first author published in the same journal in the same year several times.

 # data entry for this paper: The database initially has one row for each paper. However, some papers will require several rows, one for each experimental condition. If a paper only requires one row of input, put "1". If you add additional rows, increase the number as you add new rows (e.g., if your paper requires 16 rows, use integers 1, 2, … 16.).

Some of the papers will require several lines. This is because we want to capture data from different experiments or different time points. For example, because we have the "post-amendment" categories only once, an additional line must be added for data at several time points or for different experimental conditions (e.g. different plants, precipitation regimes, field sites, etc.). The following categories are designed such that each row of data can be uniquely identified based on these characteristics.
For all of these categories, please do not use spaces (" ") but an underscore instead ("_").

Name Experiment: Name the experiment as concisely as possible. This can be a field site, or the **primary variable tested**. For example if the main point of the paper is to test different type of feedstocks you could enter the names here, etc. An example of how to fill this out efficiently is also given in Figure 1 below.
Try to make experiment names as descriptive as possible. They should not be the same for all lines (but can repeat if for each experiment there are several experimental conditions). If a paper has several lines, each line HAS to have an experimental condition associated with it. If each experiment has only one condition, the next column may be left blank

Experimental condition: If there is a secondary condition tested, you can enter this here. For example if for each feedstock different amounts of precipitation are tested, you could enter this here. Often, this will also be the time step. Several conditions can also be combined.
For example: "1000mm_precip_1yr"

There is some variability on how this can be filled out – please be as short and precise as possible. The most important thing is that you make sure that within one paper only unique names are chosen. Feel free to consult with Jesper on this if you need help.

Unique data/experiment identifier: Gives each column a unique name based on the previous columns. This should fill automatically. If it does not, append the text from the columns "unique paper identifier", "name experiment", "experiment condition" and put a "_" in between.
For example: Author_Jounral_Year_basalt_1000mm_precip_1yr

Uncertainty data: Is any uncertainty data associated with this row? Enter the name of the corresponding excel sheet where the data can be found.

Multiple line note: If the paper is entered as multiple lines of data, insert the fields that caused the splitting up into several lines. Because several lines may contain the same data (e.g., different plant parts and their composition, but the same soil compositions), this can be difficult to see.
For example, if the reason you split into two rows is treatment and control data, put treatment/control. If the reason is several feedstocks, put "feedstock", different time steps put "time steps" – etc. This can also be a combination of these; if so please try to follow the logic of the experiment.
E.g., "treatment/control, feedstock, crop, timesteps"
There is no "correct" way to fill this out, but the most important fields the different lines are differing on should be mentioned. No need to go into detail; e.g. put "soil composition" rather than listing individual elements etc.

Please make sure that each line of data contains all necessary data for this row to make sense independently of others contained for this study. This means copying all relevant (meta-) data into all rows. Where data is copied (e.g., the same feedstock composition in several rows), all but the first entry should receive a note that this data was reported already – this is explained in more detail in section 7.1 where this data category is used for the first time.

a) inefficient naming

1.2 Unique data and experiment identification

unique paper identifier	# data entry	name experient	experiment condition	unique data/experiment identifier	multiple line note
Example_FC_2019	1	basalt as fertiliser in irrigated an non irrigated	Irrigated basalt rate (mg kg−1) 0	Example_FC_2019_basalt as fertiliser in irrigated an non irrigated_Irrigated basalt rat	the way soils are irrigated and feedstock is applied
Data missing	2	Each line needs a value	Irrigated basalt rate (mg kg−1) 42	Example_FC_2019_basalt as fertiliser in irrigated an non irrigated_Irrigated basalt rat	the way soils are irrigated and feedstock is applied
	3		Irrigated basalt rate (mg kg−1) 208	Example_FC_2019_basalt as fertiliser in irrigated an non irrigated_Irrigated basalt rat	the way soils are irrigated and feedstock is applied
	4		Irrigated basalt rate (mg kg−1) 542	Example_FC_2019_basalt as fertiliser in irrigated an non irrigated_Irrigated basalt rat	the way soils are irrigated and feedstock is applied
	5		Irrigated basalt rate (mg kg−1) 1542	Example_FC_2019_basalt as fertiliser in irrigated an non irrigated_Irrigated basalt rat	the way soils are irrigated and feedstock is applied
	6		Non-Irrigated basalt rate (mg kg−1) 0	Example_FC_2019_basalt as fertiliser in irrigated an non irrigated_Non-Irrigated bas	the way soils are irrigated and feedstock is applied
	7	They should not all be the same value	Non-Irrigated basalt rate (mg kg−1) 150	Example_FC_2019_basalt as fertiliser in irrigated an non irrigated_Non-Irrigated bas	the way soils are irrigated and feedstock is applied
	8		Non-Irrigated basalt rate (mg kg−1) 238	Example_FC_2019_basalt as fertiliser in irrigated an non irrigated_Non-Irrigated bas	the way soils are irrigated and feedstock is applied
	9		Non-Irrigated basalt rate (mg kg−1) 411	Example_FC_2019_basalt as fertiliser in irrigated an non irrigated_Non-Irrigated bas	the way soils are irrigated and feedstock is applied
	10		Non-Irrigated basalt rate (mg kg−1) 933	Example_FC_2019_basalt as fertiliser in irrigated an non irrigated_Non-Irrigated bas	the way soils are irrigated and feedstock is applied
Scientist_GC_2023	1	Biochar as fertiliser with olivene and microorganisms	100% fertiliser	Scientist_GC_2023_Biochar as fertiliser with olivene and microorganisms_100% fert	differences in amendment type and microbes added
Data missing	2	Each line needs a value	75% fertiliser + olivine	Scientist_GC_2023_Biochar as fertiliser with olivene and microorganisms_75% fertil	differences in amendment type and microbes added
	3		75% fertiliser + olivine + Microorganism spp. + Bacillus s	Scientist_GC_2023_Biochar as fertiliser with olivene and microorganisms_75% fertil	differences in amendment type and microbes added
	4		75% fertiliser + olivine + Microorganism spp.	Scientist_GC_2023_Biochar as fertiliser with olivene and microorganisms_75% fertil	differences in amendment type and microbes added
	5	They should not all be the same value	75% fertiliser + olivine + Bacillus spp.	Scientist_GC_2023_Biochar as fertiliser with olivene and microorganisms_75% fertil	differences in amendment type and microbes added
			Inconsistent use of fertilizer and biochar		

b) more efficient naming naming

unique paper identifier	# data entry	name experient	experiment condition	unique data/experiment identifier	multiple line note
Example_FC_2019	1	irrigated	0_mg/kg_basalt	Example_FC_2019_irrigated_0_mg/kg_basalt	irrigation type, application rate
Example_FC_2019	2	irrigated	42_mg/kg_basalt	Example_FC_2019_irrigated_42_mg/kg_basalt	irrigation type, application rate
Example_FC_2019	3	irrigated	208_mg/kg_basalt	Example_FC_2019_irrigated_208_mg/kg_basalt	irrigation type, application rate
Example_FC_2019	4	irrigated	542_mg/kg_basalt	Example_FC_2019_irrigated_542_mg/kg_basalt	irrigation type, application rate
Example_FC_2019	5	irrigated	1542_mg/kg_basalt	Example_FC_2019_irrigated_1542_mg/kg_basalt	irrigation type, application rate
Example_FC_2019	6	non-irrigated	0_mg/kg_basalt	Example_FC_2019_non-irrigated_0_mg/kg_basalt	irrigation type, application rate
Example_FC_2019	7	non-irrigated	150_mg/kg_basalt	Example_FC_2019_non-irrigated_150_mg/kg_basalt	irrigation type, application rate
Example_FC_2019	8	non-irrigated	238_mg/kg_basalt	Example_FC_2019_non-irrigated_238_mg/kg_basalt	irrigation type, application rate
Example_FC_2019	9	non-irrigated	411_mg/kg_basalt	Example_FC_2019_non-irrigated_411_mg/kg_basalt	irrigation type, application rate
Example_FC_2019	10	non-irrigated	933_mg/kg_basalt	Example_FC_2019_non-irrigated_933_mg/kg_basalt	irrigation type, application rate
Scientist_GC_2023	1	biochar		Scientist_GC_2023_biochar	soil amendment type, microbe species
Scientist_GC_2023	2	biochar_olivene		Scientist_GC_2023_biochar_olivene	soil amendment type, microbe species
Scientist_GC_2023	3	biochar_olivene_microorganism_bacillus		Scientist_GC_2023_biochar_olivene_microorganism_bacillus	soil amendment type, microbe species
Scientist_GC_2023	4	biochar_olivene_microorganism		Scientist_GC_2023_biochar_olivene_microorganism	soil amendment type, microbe species
Scientist_GC_2023	5	biochar_olivene_bacillus		Scientist_GC_2023_biochar_olivene_bacillus	soil amendment type, microbe species
all rows have data		correspond to primary variable tested	no spaces used	more efficient unique_data_identifiers	efficient naming

Figure 1: Example of erroneous and inefficient data input (a) and correct and efficient data input (b). Note that each line should have all associated meta data. Experiment names should be chosen that best describes the main variables that were tested, and should not be the same for all lines of data. Same is true for experimental condition, though this is only necessary if there are several conditions per experiment. If an experiment combines several conditions, the experimental condition can also be a list of several variables. Avoid using spaces and use "_" instead.

S3.1.3 Uncertainty meta-data (uncertainty files only)

As introduced in section 0.3, we are also attempting to capture some uncertainty data. To make this possible, the uncertainty database files have some additional input categories.

As mentioned above, non-quantitative input fields are removed from the uncertainty files, and they only contain data categories where quantitative information is the default input. For these categories, the uncertainty files still contain the unit categories, primarily to make it easier to format data and units at the same time before input is copied into database files.

The uncertainty files include these two additional categories to clarify units as well as type of reported uncertainty.

Uncertainty units: What units are reported in this database? Possible values are:

- Same as database file (absolute uncertainties): The assumption is that you are inserting absolute uncertainties as well as the absolute range of data (min and max files) in the same unit as the primary data is reported. In this. Case you may leave all unit categories where this applies blank. You can indicate categories where, for whatever reason, you use another unit by entering the units for these categories (- but generally try to avoid this!).
- Relative uncertainties: all in % and Relative uncertainties: all as a fraction (0-1): Relative uncertainties can be either entered in % or as a fraction of the data value (0-1). Please insert which of these is the case and be consistent throughout the entire file.

Uncertainty type: Studies may report different types of uncertainties. Please insert which of the possible input values is reported in this data file.

S3.2 Builds upon / cross references

This category is meant to capture dependencies within studies. Some publications heavily rely on the results of other studies. If so, the study (or even experiment) that is relied upon should be entered here. By this we do not mean to enter every reference in the paper, but only if one paper directly uses data/experiments that were published elsewhere.

The challenge in this category is that we are building this database in parallel and not all (chronologically) previous database entries are available to you. So you do not have a list of all available unique data/experiment identifiers at your disposal yet. So, at a minimum, enter the unique paper identifier following the rules lined out above (section 1.2).
If possible, also add unique data/experiment identifiers to the best of your ability, and we will subsequently make sure they are consistent with the naming conventions used by the person that filled in the cross-referenced publication.

Enter all cross references that apply (within the same row for each data entry), after all one paper may heavily rely on data of several others. This category is also relevant for follow up studies, where previous results of the same experiments are already published – in this case the previous results should definitely be cross referenced here.

An example: Amann_FC_2022a and Amann_FC_2022b make use of data from the same small-scale lab experiment under ambient vs elevated CO2. Therefore, the 'builds upon' column corresponding to Amann_FC_2022a should specify that the paper identifier Amann_FC_2022b makes use of data from the same experiment, and vice versa.
Just relying on someone else's data (e.g., papers using the feedstock composition reported in Lewis et al. (2021) is not enough to qualify for this section, it should be shared experiments or follow up studies, etc.

S3.3 Actors involved

This is a mandatory section. We would like to be able to capture what kind of (commercial) interests are present in a study, and how this has evolved over time.

Decide based on conflict of interest and acknowledgements sections as well as affiliation list. Note that sometimes affiliation with non-profit actors is not listed in the conflict-of-interest section, particularly if affiliation is in the form of scientific advisory, so check the affiliation list carefully. Note that the category "non-profit" includes for-profit B-corps that are owned by a non-profit entity (e.g., Mati Carbon which is owned by the non-profit Swaniti Initiative). Fill this in to the best of your knowledge.

Options:

1. *Academics only;* no co-author has any role in any kind of non- or for-profit supplier, platform, standards body, etc.

Funding:

2. Study was funded by a for-profit party. This option is irrespective of authors employment; list even if no author is affiliated with the for-profit stakeholder.

Scientific advisership or similar:

3. at least one person involved is scientific adviser or similar to a for-profit ERW supplier.
4. at least one person involved is scientific adviser or similar to a non-profit ERW supplier (or that is owned by a non-profit).
5. at least one person involved is scientific adviser or similar to a for-profit ERW platform or standards body.
6. at least one person involved is scientific adviser or similar to a non-profit ERW platform or standards company (or that is owned by a non-profit).

Employment:

7. at least one person involved is directly employed by a for-profit ERW supplier.
8. at least one person involved is directly employed by a non-profit ERW supplier (or company owned by a non-profit).
9. at least one person involved is directly employed by a for-profit ERW platform or standards company.
10. at least one person involved is directly employed by a non-profit ERW platform or standards body (or that is owned by a non-profit).

Equity:

11. at least one person involved has equity in a for-profit ERW supply company or for-profit platform or standards body.

Note that if any of 2, 3, 5, 7, 9, 11 values are used, the name of the for-profit actor needs to be listed in the second column of this category**.** List all that apply, e.g., "2, 8, 11".

S3.4 Type of study

What is the primary focus and the secondary focus for the study? Select most applicable of the possible pre-defined answers. You can leave secondary focus empty should there be none.
The type categories refer to the methods a study is employing, while the focus categories refer to the main topic of a study. The non-self-explanatory categories are defined below.

Type categories:

Experimental categories: Any studies for ERW (or rock flour as fertilizer) that manipulate experimental variables to test cause and effect relationships. Such experiments do not need to be included in "data analysis" if they only do basic calculations.
Laboratory studies include highly controlled settings such as all experiments on soil cores, batches, etc. within labs but also mesocosm experiments (i.e., both in- and outdoor mesocosm experiments are counted as laboratory studies).
Field studies include all studies that happen outside of highly controlled settings such as on forests and in fields.
ERW studies are looking at rock flour addition in the context of CDR, while "rock flour as fertilizer" studies only look at agronomic effects, without mentioning CDR.

Natural Analogue Study: A natural analogue study examines real-world, naturally occurring processes that mimic enhanced rock weathering (ERW). These studies provide empirical evidence from natural settings where similar mineral weathering processes occur, helping to understand potential outcomes of ERW implementation.

Integrated Assessment Model (IAM): IAMs are used to explore and quantify the interactions between different systems, such as climate, economy, and energy, in response to policies and technology approaches like enhanced rock weathering. These models integrate scientific, economic, and environmental data to predict the large-scale outcomes of ERW on global carbon cycles. More info on what IAMs are can also be found at this UN definition.
These models are mentioned separately here because they may prescribe emissions / CDR pathways, and hence derived CDR estimates may not necessarily reflect geochemical bottom up efficacy estimates, but top down ideal pathways within a given policy.

Life Cycle Assessment (LCA) Model: An LCA assesses the environmental impacts of enhanced rock weathering throughout its entire life cycle, from mining and transportation of minerals to application in fields and eventual carbon sequestration. The aim is to evaluate the sustainability and carbon efficiency of the entire process.

Modeling paper (Other): This category includes other modeling studies not covered by IAMs or LCA. These may involve the simulation of reactive transport models, mineral weathering kinetics, field-scale deployment models, or biogeochemical interactions within soils after ERW application.

Opinion/editorial/perspective: These are articles that provide subjective analysis or expert insights into the potential, challenges, and future direction of enhanced rock weathering. They may not involve original data but rather offer critical discussion or propose conceptual frameworks for

research and policy. This also includes (critical) comments of published papers as a response in the same journal.

Review: A review synthesizes existing research on enhanced rock weathering, summarizing key findings, methodologies, and gaps in knowledge. It offers a broad overview of the current state of the field.

Systematic review: This is a rigorous evidence synthesis method that uses reproducible methodologies to collect and analyze secondary data on enhanced rock weathering. It may involve both quantitative and qualitative analyses, often including meta-analyses of multiple datasets. If something is a meta-analysis without being a systematic review (i.e., without a clearly defined search criteria for all available literature) label as data analysis.

Data analysis: Data analysis involves the use of statistical methods, machine learning, or other computational techniques to analyze data specific to enhanced rock weathering. This may include remote sensing data, experimental data, or meta-analyses (without being a systematic review).

Survey: A survey gathers data from individuals or groups, such as experts, farmers, or stakeholders, to gauge their perceptions, knowledge, or experiences related to enhanced rock weathering. This may include structured questionnaires or interviews.

Qualitative research: This label applies to research that uses non-quantitative methods to explore aspects of enhanced rock weathering, such as social, cultural, or political implications. It may involve case studies, frameworks, or focus groups.

Other (Specify): This category captures studies that don't fit neatly into the other labels but still contribute to the enhanced rock weathering literature. The specific nature of the study should be described.

Focus categories:

Public perception & acceptance: This topic explores the attitudes, knowledge, and concerns of the public regarding the use of enhanced rock weathering as a carbon dioxide removal (CDR) method. It assesses the societal readiness and willingness to adopt ERW technologies, including potential resistance or enthusiasm.

Policy, politics & governance: This focus area examines the political landscape, regulatory frameworks, and governance issues surrounding the deployment of enhanced rock weathering. It looks at how ERW could be incentivized, regulated, and integrated into national and international climate policies.

Equity & ethics: This topic addresses the ethical implications of enhanced rock weathering, particularly in terms of its fairness and impact on different populations. It considers how ERW might affect marginalized communities, global equity, and the broader ethical questions of geoengineering interventions.

Environmental side effects: This focus explores the potential unintended environmental consequences of ERW, including effects on ecosystems, biodiversity, water quality, and soil health. It assesses both positive and negative impacts that ERW could have beyond its primary goal of carbon sequestration.

Non-environmental side effects: This category considers the unintended consequences of ERW that are not directly environmental but might affect social, economic, or other human systems. It includes concerns such as disruptions to local economies, food systems, climate change resilience, etc.

CDR through ERW: This topic focuses on assessing the potential of enhanced rock weathering as a carbon dioxide removal (CDR) technique. It includes the study of weathering rates, the amount of CO_2 sequestered per ton of rock applied, and other factors affecting the efficiency of ERW, as well as studies looking at the potential of ERW for CDR.

Carbon losses & signal lag after initial weathering: This area examines the carbon dynamics after the initial application of ERW, including potential losses of removed carbon and the lag time before carbon capture is detectable. It explores how CO_2 might be re-released or delayed in its sequestration.

Silicate rock flour as fertilizer, agronomic effects of ERW: This topic looks at the potential agronomic benefits of using silicate rock flour as a soil amendment and fertilizer, such as impacts on crop yields, improve soil health, and provide essential nutrients to plants. If the study looks at CDR at all, use CDR through ERW as primary label. If additional agronomic benefits are mentioned, you may use this as the secondary label.

Scoping of available rock/land resources: This focus assesses the availability and accessibility of suitable silicate rocks and land areas for ERW deployment. It looks at geographic, logistical, and economic factors influencing where and how much ERW can be implemented.

Technological & logistical implications, technology readiness: This area explores the technological requirements and logistical challenges of implementing enhanced rock weathering at scale. It assesses the readiness of current technology, including rock grinding, transportation, and spreading, and the potential for innovation.

Urban soils as CO_2 sinks: This category explores how engineered urban soils may be used to remove CO_2 from the atmosphere, or how existing urban soils are already removing additional CO_2 from the atmosphere, e.g., due to concrete residues in soils.

Other (specify): This category captures studies that focus on topics not specifically covered by the other labels but are still relevant to enhanced rock weathering. The specific focus of the study should be described in detail, with as few words as possible.

S3.5 Methods of modeling studies and reviews/social science studies

This section aims to capture method details for modelling studies as well as literature reviews and social science studies, e.g., quantitative/survey-based assessments of public perception, etc.
Although these are clearly very different types of studies, all "non-experimental" studies are grouped only to keep the number of total sections to a minimum.

S3.5.1 Model & survey methods

This section has a few categories aimed to capture the employed methods of these types paper. If one of these categories applies, please enter a short descriptive text on the model details, scale, etc. If a study combines several types of models, please insert short information on each of these (e.g., some papers combine reactive transport and GCM models, etc.).

For example if it is a shrinking core model, add whether it considers a particle size distribution or single particle size, whether it assumes spherical particles or not etc.

Note that GCM stands for global circulation model, and CMIP for coupled model intercomparison project. Ocean biogeochemical models includes published models such as GEnIE (Grid Enabled Integrated Earth System Model). List the name of the specific model in the correct category. IAM and LCA models are also further defined in section 4.

S3.5.2 Non-experimental studies: underlaying data & assumptions

These categories allow to add some of the data sources the models are based on, or associated assumptions.
If possible, also add the DOI or a working hyperlink to the original databases (preferrable DOI), for example:

USGS: Area- and Depth-Weighted Average of Soil pH from STATSGO2 for the Conterminous United States and District of Columbia, https://doi.org/10.5066/P95YOSFQ
USGS: Geochemical and Mineralogical Data for Soils of the Conterminous United States, https://pubs.usgs.gov/ds/801/

Transport networks: Enter details on how the study models transport distances. Is it just using distance from rock outcrops or whether road networks are considered (and if so, what data/modelling process is used?

S3.5.3 Review and social science paper methods

If this study is a review paper or a social science paper (assessment of public perception, expert interviews, etc.) add a few words on the methods into the category that fits best.

S3.6 Experimental conditions (lab & field experiments, modelling studies)

This section is primarily meant to cover all experimental conditions of experimental lab and field studies. However, relevant data from modelling studies should also be inserted where relevant.
For example for reactive transport models, how deep is the modelled soil column and what is the water throughput etc. Enter all information that makes sense for the specific study.

S3.6.1 study scale and site

This section gathers basic information on the experiments of that study.

Note that many of these categories are what sets different experimental conditions of the same study apart. If one study for example has one experiment with 400 mm/yr irrigation, and another one with 1000/yr, put these in different lines and fill remaining categories either with the same data that is general to the paper, or different data specific to this datapoint (e.g., post-amendment composition or CDR).
Refer to sections 0 and 1.2 for more information.

Type of experiment: Enter the category that best fits the type of experiment

Volume: enter the volume of the experiment for any mesocosm (incl. pot), soil core, and lab/batch experiment. This does not need to be entered for agricultural/forest field trials. Enter the unit in the corresponding

Surface area: Enter the surface area of the experiment or model. This is particularly important for calculating some of the application and CDR rates – if not directly given in the main text, please check SI material; you may also be able to calculate this from the volume and depth.

Depth/length: Enter the depth of the experiment, you do not need to enter this for field experiments, but it is particularly important for soil core studies or lysimeters that look at effluents below a soil column as well as reactive transport models.

Duration: Experimental or model duration is an especially important variable, as this is fundamental to convert many of the measurements. Here, make sure to enter the duration of the entire study.
For example if the total length of experiment is 4 years, but the data that you report in this line is for after 2 years, enter 4. The duration for this specific data point is entered in section 8.

LON/LAT: enter coordinates in decimal degrees formatting, omit the °. Enter N and E coordinates as positive values, S and W as negative values.
E.g. 10.212312 °W = -10.212312
Convert from other formats (e.g. including minutes and seconds) using this website: https://www.pgc.umn.edu/apps/convert/.

Location (4 categories): The world region category may should be interpreted in the way the study treats relevant categories; if ERW is discussed by continent, then continents should be used. In some cases it may be more appropriate to use climate zones or for example "hinterlands" if a study looks at ERW on all hinterlands, or "farm land" if a study models ERW on all global farm land.
In the "Municipality/site" category, be as precise as possible.

Climate; describe climate type if given in paper (use the Köppen-Geiger climate classification if given).

Soil type: provide the full name of a soil or soil type if it is given in the paper. Please also include if national or international classification such as WRB or USDA Soil Taxonomy was used. For example: WRB: Albic Endostagnic Luvisol (Loamic)

Vegetation/crop present: Does the experiment or model condition include plants (or funghi, etc.) as an experimental condition? Specify which plants. There is going to be a more detailed category later which allows to also specify plant parts etc.

Crop rotation: Enter what kind of crop rotation was performed on this field. In each line of data, specify in the vegetation/crop present field what was planted in the year of sampling**.**

Biota present: Does the experiment or model include animals as an experimental condition? For example Earthworms.

Soil present: Does the experiment or model include soil as an experimental medium, or does it use pure feedstock minerals in e.g. a batch experiment? If soil is present enter "yes" or add more detail, if no soil is present leave blank.

Water parameters: Note that there are both values per year and for the experimental period. Only enter given values and do not convert for now.
It is very important that the study parameters like (L/pot) etc. are converted to water column or water column/time values (e.g., mm or mm/yr) using the geometry of the experiments whenever possible.

Control/treatment/baseline fields/definitions: A few definitions; a treatment is an experimental condition where feedstock is added. A control is an experimental condition where no feedstock is added. The control is often also sampled at the same time steps as the treatment to capture natural variation and trends and soils. Baseline samples are taken before the start of experiments and may be taken both on control and on treatment fields, where the control baseline is important to monitor temporal variability in the field after deployments started on treatment fields.

Baseline type: Often, treatment and control conditions use a baseline to compare the effect of the experiments to. In almost all cases, this is going to be the same field (control or treatment) before deployments initially started, i.e., at t=0. In some special circumstances, treatments and controls may choose to use the previous years data as baseline (even if they were deployments before), or

treatments may choose the control (at either t=0 or t=n) as baseline. Although these are uncommon, this category is designed to capture such choices.
In this context, "treatment plot before last application (t=n-1) refers to the case when the treatment field data before the latest deployment (but after previous deployments) is used (same for control fields).

Pooled samples: How many samples/sediment cores were pooled to give one sample that was analyzed? For example, if 5 cores were pooled into 1 sample, insert "5" in the pooled samples category. If 4 of such samples were taken for a specific experiment, put "4" into the category " # samples for this field/condition" – do not calculate the product.
The samples per area can be calculated for field deployments.

Soil sample preparation: in soil analysis, the information about soil preparation is important factor. Please describe here the conditions for soil sample storage and pre-treatment, such as drying method (air-drying or oven-drying), sieving method, sieve sizes and grinding method.

S3.6.2 irrigation water composition

Irrigation water composition data fields. If irrigation water was used (or modelled), and the water composition is reported, enter all fields that apply.
Please report in any of the pre-defined units if possible, else provide the unit and details necessary for conversion. Compounds like e.g. NaNO3 should be split into individual elements (by their molar mass) if mass concentrations are used – in such cases it may be easier to report molar concentrations if given.

The section is subdivided into:

S3.6.2.1 rain/irrigation water elemental composition

S3.6.2.2 irrigation water carbon species, carbonate system, and major ions

S3.6.2.3 irrigation water isotope composition

S3.6.3 experimental data/medium

Sample depth parameters: If a single sample depth is provided for a specific 1-cm depth increment, put this value, else use sample depth minimum and maximum fields to set range (e.g., 0-10 cm).

Soil/other phase assessed: While some studies look at bulk soil, others look at only part of the soil, for example at exchangeable phases (or any other leach-based approach), the composition of secondary carbonates, etc. This is what is meant in this category – what is the phase this study assesses?

For studies assessing several phases, these fields should match the data reported in this line. Hence, put only 1 value per line, if several are assessed, add a new line. The concentration data for these measurements are entered in sections 7.1 (pre-amendment) and 9.1 (post-amendment).

Exchangeable leach done for cleaning: Some studies employ leaching methods. This can be done to either clean the sediments before other phases were assessed (i.e., the leachate phase is not measured). In this case enter the reagent used for cleaning in this category.

Leachate used for analysis: Other studies are using leaching methods to actually assess selective sedimentary phases. If this is the case for this line of data, enter the reagent in this column. This includes the analysis of exchangeable phases; enter the method here.

pore H2O extraction method: If a line of data contains pore water data, please specify the extraction method.

S3.7 pre-amendment data

These categories are aimed to capture all relevant experimental parameters before the start of the experiment.

Control/treatment: This is a very important category that captures what type of data is collected in this row: control or treatment data.
Each row of data is a pre- and post- treatment sample pair for either a control condition or a treatment condition.

S3.7.1 pre-amendment soil elemental composition

The data inserted here should correspond to the phase indicated in section 6.3. E.g., if you have indicated in 6.3 that this line of data is for bulk soil concentrations, insert the bulk soil composition in this section.

Data reported already?: Some of the data entered may not be unique, i.e., occur in several rows. For example the data collected at different timesteps will always be paired with the same baseline data (for both treatment and controls), or when several parameters are reported while other are constant (e.g., several different plant parts that all correspond to the same soil concentration).
To not double count data in statistical analysis, insert "yes" If this data has already been reported in another row, but is still entered because it is e.g. the same baseline data, or because other data parameters in this row are different to the previous row.
Note that this only refers to data that has already been mentioned in a different row within the same category (i.e., 7.1 in this case), not for post-amendment categories that have been mentioned in the pre-amendment sections.

Enter further soil data to the best of your ability as it is given in the paper.

S3.7.2 pre-amendment soil elemental concentration by oxide composition

In some cases, soil composition may be given in elemental oxide composition. If so, insert the concentration in this section. If concentration is given in both elemental concentrations (section 7.1) and oxide composition, it is sufficient to fill in 7.1 and leave 7.2 blank – only one of them is necessary with elemental concentrations preferred.

The data inserted here should correspond to the phase indicated in section 6.3. E.g., if you have indicated in 6.3 that this line of data is for bulk soil concentrations, insert the bulk soil composition in this section.

S3.7.3 pre-amendment soil isotope composition

The data inserted here should correspond to the phase indicated in section 6.3. E.g., if you have indicated in 6.3 that this line of data is for bulk soil concentrations, insert the bulk soil composition in this section.

S3.7.4 pre-amendment soil mineralogy

Any missing phases should be entered in the column "other", list several by separating with commas if necessary.
In this case the value column would be "1.23, 5.23", the unit column "wt%, wt%", and the minerals can be entered in the comment column "mineral X, mineral Y".

S3.7.5 pre-amendment soil physical and hydrological parameters

Porosity: The proportion of pore space in a soil sample, indicating its ability to hold air and water.

Void ratio: The ratio of the volume of voids to the volume of solids in a soil sample, representing the degree of soil compaction.

Field capacity: The maximum amount of water held by soil against gravity after excess water has drained away.

Permanent wilting point: The moisture level at which plants wilt and fail to recover, indicating the minimum water content required for plant survival.

Bulk density: The mass of soil per unit volume, including both solids and porosity.

Specific gravity: The ratio of the density of a substance to the density of water, providing a measure of the density of soil solids relative to water.

Moisture content: The amount of water present in a soil sample.

Texture: The relative proportions of sand, silt, and clay particles in a soil sample, influencing its water retention, drainage, and nutrient-holding capacity.

S3.7.6 pre-amendment chemical parameters

pH method: Often pH measurements (that are not buffer pH!) are measured in pure H_2O. However, other solvents may be used to help stabilize pH. Please insert the solvent used here – if not H_2O, also specify the concentration or dilution. (E.g. water 9 : 1 KCl).

buffer pH: The pH of soil after a buffer solution has been added. Theh test is used to measure the buffering capacity of soil, specifically its ability to resist changes in pH when lime is added. This test is critical for determining the amount of lime required to raise soil pH to a desired level, particularly in acidic soils. Specify also the buffer solution (and soil:buffer ratio) used.

SOC/SOM method: Measurements total organic carbon / organic matter concentration of the bulk soil may depend on the method used, hence it is important to capture some method information. Please include here applied method (dry or wet combustion, temperature etc).
SOC: Soil organic carbon (the mass is a subset of SOM)
SOM: Soil organic matter

SOM fractionation method: soil organic matter can be divided into two main fractions that are mineral-associated organic matter (MAOM) and particulate organic matter (POM), which provide information about residence time of organic matter in the soil. Please describe here OM fractionation method by providing information about the solution used for the dispersion (together with concentration), applied physical dispersion, and names and sizes of fractions that were separated. In case of density fractionation was used, include also the density of the solution.

POM: POM primarily consists of lightweight and partially decomposed organic compounds, such as leaf litter fragments, and its residence time in soil is short. After density fractionation, this parameter will be characterized by measuring C concentration and expressed as POM-C, usually in mg/g soil.

MAOM: this fraction describes the organic matter fraction associated with silt- and clay-sized soil minerals, and have long residence time due to the adsorbance on the mineral phase. After density fractionation, this parameter will be characterized by measuring C concentration and expressed as MAOM-C, usually in mg/g soil.

S3.7.7 pre-amendment pore water composition

S3.7.7.1 pre-amendment pore water elemental composition

S3.7.7.2 pre-amendment pore water carbon species, carbonate system, and major ions

S3.7.7.3 pre-amendment pore water isotope composition

S3.7.8 pre-amendment exchangeable pool composition

The composition of the cation exchangeable complex (if such a leach has been done and reported).

Concentration info: Exchangeable ion concentrations are sometimes expressed as relative to the leachate or relative to the initial soil – this information is important to capture to close mass

balances. Hence, indicate whether reported concentrations are with respect to leachate or soil volume/mass.

S3.7.9 pre-amendment effluent water composition

Note that effluent water is meant to capture any water flowing out of the soil column; depending on the study this may be leachate dripping out of the soil column, or stream/river water.

S3.7.9.1 pre-amendment effluent water elemental composition

S3.7.9.2 pre-amendment effluent water carbon species, carbonate system, and major ions

S3.7.9.3 pre-amendment effluent water isotope composition

S3.7.10 pre-amendment biomass

Note that the concentrations here refer to concentrations in plant biomass. The section is split into two units, the first being agronomic yield information, and the second being data on individual plant samples.

S3.7.10.1 pre-amendment harvest agronomic data

Harvest Index (HI): The ratio of economic yield (e.g., grain) to total above-ground biomass. This helps assess the efficiency of the crop in converting total biomass into usable products.

Nutrient Use Efficiency (NUE): The ratio of nutrient export via harvest to nutrient inputs (e.g., kg N in harvested biomass per kg of N fertilizer applied). This indicates how efficiently the crop utilized available nutrients for growth.

Root:shoot ratio: Ratio of the dry weight of root biomass to the dry weight of the shoot biomass.

Further information on biomass composition (incl. major and trace elements and nutrients) can be inserted in the next section.

S3.7.10.2 pre-amendment biomass data

This section is meant to provide some more details on the composition of relevant biomass. This can either be the composition of the harvested primary crop, or of other plants or animals.

Species: If this section contains data on the primary crop harvested, add “harvested main crop”, else specify for what species data is provided for – this can be plant or animal. Define what plant/animal part in the next category.

Material: Define what material concentrations are referring to, i.e., if they are referring to wet or dry biomass, etc.

S3.7.11 pre-amendment soil structure

Aggregate fractionation: please describe here applied method for aggregate fractionation by including sieving method, sieve sizes, intensities of sieving and if ultrasonic dispersion was used (please specify applied energy). Please specify fractionated aggregate size classes and their size ranges.

Large macroaggregate: please provide the mass distribution of each separated aggregate size classes.

SOC large macroaggregate: please provide their SOC concentration if it was measured.

KLD index: the Kullback-Leibler divergence is a recently developed index to describe soil structural quality, which is quantified from the soil water characteristic curves using the concept of relative entropy. Please insert the value of KLD index, if it was calculated.

S3.7.12 pre-amendment soil GHG emissions

Please insert soil GHG emissions in this section. There are a lot of combinations of units possible, please use one of the pre-defined ones if possible, else clearly specify the unit used.

Scale note: For large scale studies or inventories, emissions may be reported in mass/molar amount per time – in this case indicate here the spatial scale of this estimate.

S3.8 Soil amendments → for modelling & experimental studies

S3.8.1 Feedstock type and application

Feedstock type, possible values:

- 1 – use if virgin material, mined for the purpose of ERW
- 2 – mine waste, extra grinding required
- 3 – mine waste, used directly (no extra grinding)
- 4 – industrial waste / cement, extra grinding required – enter if steel slag, mine tailings, concreate/construction wastes etc.; it has already gone through an industrial process other than just grinding, and it still requires grinding
- 5 – industrial waste / cement, extra grinding required – no further grinding required
- Other (specify)

feedstock-soil mixing depths: If not specifically reported, but plowing depth is, assume this is the ploughing depth.

Elements used for reported CDR potential: list all that apply.

CDR potential discounts for downstream loss: Some studies discount expected downstream CO_2 evasion (e.g. in rivers and oceans) from the CDR potential of rocks. Please read these sections carefully. This is often the case if Renforth 2012 or other Renforth based formulations of the Steinour formulation are referenced. By default, Renforth 2012 uses an ω value of 1.7, which essentially assumes 15% loss in rivers and oceans. If no such losses are considered, leave blank. Please look at the details, when in doubt contact Jesper and/or Tyler.

Single application amount: Some experiments give feedstock application in g/experiment or similar. This is not useful to compare with other data. Please make sure to use on of the pre-defined units or a add comment about conversions to them if not.

Single application mass ratio: Some experiments, particularly laboratory ones, do not report application rates but mix soil and feedstock by mass. In this case, add the feedstock mass ratio here. You do not need to calculate this.

Repeated application rate: If given, please report in tons per hectare.

Duration: This is extremely important for a lot of the meta analysis. Make sure to enter the duration that is relevant to this line of data.
For example if the total length of experiment is 4 years, but the data that you report in this line is for after 2 years, enter 2, not 4. The duration of the entire study is entered in section 6.

S3.8.2 feedstock particle size

Unfortunately, the way that literature reports feedstock particle size is very inconsistent. Hence, here we provide a range of different ways reflecting common ways how this is reported. Please do not fill min and max values into files for uncertainty ranges, but fill them here as they do not reflect uncertainty but the actual size range.

It is important to capture whatever particle size information a study provides – if it cannot be captured with the existing categories, please add the information in the "other measure" category.

Continues particle size distribution given?: Insert "yes" if the paper reports the whole particle size distribution (i.e., in a plot or as meta-data).

p80, p90, and p97: These parameters refer to the particle size that 80, 90, or 97% of particles (by mass) are smaller than. E.g., a p90 of 100µm means that 90% of the particles have a diameter smaller than 100 µm.

S3.8.3 feedstock elemental composition

Phase assessed: Fill in case the elemental composition is not for bulk feedstock data but for individual mineral phases, leached phases or similar.

S3.8.4 feedstock elemental concentration, oxide composition

S3.8.5 feedstock mineralogy

S3.8.6 fertilizer amendments

In the database file, this section is separated into two, one for basic data and the second for more detail on the fertilizer composition

S3.8.7 biochar amendments

Preaging/treatment: Some biochar is treated or pre-aged to e.g. increase CEC. If anything like that has been done, please specify.

S3.9 Post-amendment soil characteristics

The categories in this section are identical to the pre-amendment categories in section 7.

Some studies report post amendment data **only** as the difference to the initial conditions. If a study gives both the pre- and post-amendment data as well as the difference between the two, fill in the pre- and post-amendment data. In this case you can ignore the reported differences as they can easily be computed. However, some studies only report the difference between pre- and post-amendment. In this case you may fill these data in the post-amendment section by using either the units "delta %", or any of the predefined units with a "delta" in front, such as "delta ppm" – if this is the case leave the relevant categories in section 7 blank. Having complete pre- and post-deployment data pairs is preferrable, so do this only when the relevant data is not reported.

Please only add post-amendment data that is specifically mentioned**.** Do not enter any pre-amendment parameters that you assume have stayed the same if they are not explicitly reported. Similarly, regarding the "data reported already?" categories – this only applies to data in several lines within the same category. If for example there are several plant parts that were measured in the post-amendment section, but each refer to the same post-amendment soil composition, you would put "yes" – this data was reported already – for the second, third, etc. time this data is listed in the same category.

Control/treatment: Controls are often sampled at the same time as the treatment plots. Make sure to keep each row either as control/treatment, but do not mix between pre- and post-amendment (i.e., do not put pre-amendment control field data in the same line as post-amendment treatment field data). For more explanations on these definitions also see section 7.

Definitions of the categories in the sub-sections below can be found in the corresponding sections of section 7 in this code book.

S3.9.1 post-amendment soil elemental composition

S3.9.2 post-amendment soil elemental concentration, oxide composition

S3.9.3 post-amendment soil isotope composition

S3.9.4 post-amendment soil mineralogy

S3.9.5 post-amendment physical soil and hydrological parameters

S3.9.6 post-amendment chemical parameters

S3.9.7 post-amendment pore water composition

S3.9.7.1 post-amendment pore water elemental composition
S3.9.7.2 post-amendment pore water carbon species, carbonate system, and major ions
S3.9.7.3 post-amendment pore water isotope composition

S3.9.8 post-amendment exchangeable pool composition

S3.9.9 post-amendment effluent water composition

S3.9.9.1 post-amendment effluent water elemental composition
S3.9.9.2 post-amendment effluent water carbon species, carbonate system, and major ions
S3.9.9.3 post-amendment effluent water isotope composition

S3.9.10 post-amendment biological plant parameters

S3.9.10.1 post-amendment harvest agronomic data
S3.9.10.2 post-amendment biomass data

S3.9.11 post-amendment soil structure

S3.9.12 post-amendment soil GHG emissions

S3.10 CDR

This section aims to capture different types of CDR . It is designed to capture both experimental/model data of small experiments and deployments, as well as national or even global estimates of CDR, the CDR potential of available feedstocks, as well as the (cumulative) potential for CDR at different scales in the future. Hence, there are a couple of important categories necessary to define which of these estimates a certain line of data is reflecting, and what the underlaying spatial and temporal assumptions are.

As this section captures CO_2 removal, positive values indicate removal. If processes are a CO_2 source, please insert a negative value.

Do not convert between mass/area/time, mass/year, and mass based on the data given. Only report data as it is given in the study. However, make sure that the duration and size data (section 6.1, 8.1) of experiments is correct such that we could still do such conversions at a later point.

Estimates of cumulative CDR potential of ERW by a certain date (and over a defined spatial scale) need to be inserted in section 10.5.

In general, a CDR estimate should correspond to the method and other data reported in the same line (e.g., for example what type of water or sedimentary phase an estimate is based on). Nevertheless please make sure to clarify any uncertainties in the CDR plain language and comment categories at the end of each section!

S3.10.1 general CDR categories

Feedstock dissolution detected: If significant dissolution is detected in some but not all data (e.g., it shows up in Mg data but not Ca), enter “yes” or add additional information (e.g. for which variables dissolution was detected - any value will count as yes).

Fraction feedstock dissolved: Insert as fraction of the cumulative feedstock applied that has dissolved by this data point.

Fraction Ca/Mg/Na/K lost: Sometimes studies report the fraction of major cations introduced to the experiment through feedstock addition that has been lost due to chemical weathering. These are often studies using solid-based mass balance approaches to quantify CDR, assuming that the mobile cations released from feedstock have left the soil. If the study reports the fractions of major cations lost from feedstock due to dissolution please report them here.

Loss of SOC/SOM detected: If the study finds that a significant amount of SOM is lost in the experiments/model runs, enter “yes” or add additional information (any value other than no will count as yes). If loss was detected in some but not all experimental conditions, enter details. If it was assessed but no loss was detected, enter “no”.

Concerns about data quality for CDR calculations: This category is extremely important. Insert any subjective concerns you have regarding the suitability of the data for a general CDR assessment or including the data in a meta-analysis together with other experiments. These concerns can be both in terms of experimental set up (e.g., particle size, too dry, etc.) or of analytical/methodological nature.
Specify what your concern is – for example "Feedstock too large (>1mm)" or "Too dry for weathering to occur", etc. Please enter a note for feedstock if more than 50% of the feedstock is larger than 1mm.
This category is not meant to directly disqualify data from meta-analysis, but as a highlight for users of the database to actively consider whether they want to use this data.
If you have no concerns, leave blank.

S3.10.2 CDR quantification and carbon budgets: mass CO2 per area per time

The purpose of this section is to capture CDR estimates made in mass per area per time. The first categories are designed to capture some basic data, including on the temporal and spatial scales of the estimate.

Based on published data: Enter yes if this study comes up with CDR estimates based on a (meta-) analysis of already published data. If the data reported here is based on new experimental or model findings of this study, leave blank.

Estimate type: Please insert what type of experiment/model this CDR estimate is based on. This doubles to some extent with data from section 5 and 6, but it is crucial that this meta-data is correct, hence we have added a similar category here, again.
If this datapoint reflects the theoretical CDR potential of available feedstock based on an assessment of feedstock availability, choose "feedstock availability/CDR potential".

CDR potential estimate year: Enter for which year this CDR estimate is for. If it is for the specific year of the study simply insert that, but some studies also estimate CDR rates at future points in time. In this case enter this date here, e.g., "2050" or similar.

CDR rate/potential type: For any type of CDR estimate, irrespective of scale and if for present/future, fill in which type of CDR potential is reported. Choose of either of the following definitions, and, if necessary clarify in the following comment category:

- CDR Potential of Rock/Waste Feedstocks: The total amount of carbon dioxide removal achievable by fully carbonating all available rock or waste feedstocks (defined for a certain spatial extent) suitable for enhanced weathering. This estimate does not account for practical limitations such as deployment feasibility on agricultural land, transport logistics, geochemical weathering limitations, or geographic constraints.
- Maximum Potential (Technical Potential): The amount of carbon dioxide removal achievable through the full implementation of enhanced rock weathering, considering natural and physical limitations like weathering rates, climatic conditions, and the capacity to export alkalinity to the oceans (i.e., considering losses). However, this potential ignores economic costs, logistical challenges, policy constraints, and societal barriers. For

example, this potential assumes that ground silicate rocks are spread across all viable land surfaces to the fullest extent that chemical and environmental conditions allow.

- Economic Potential: The projected carbon dioxide removal achievable when all relevant social costs and benefits of enhanced rock weathering are considered, under the assumption of efficient market dynamics and full transparency of information. This potential includes the pricing of negative externalities, such as environmental impacts, and co-benefits like improved soil health. It assumes that economic systems adjust optimally over time and across regions, with social discount rates balancing the needs of current and future generations. This is the type of potential typically estimated by integrated assessment models (IAMs), factoring in data on constraints like carbon capture capacity and political or governance barriers for further discussion.
- Sustainable Potential: The economic potential of enhanced rock weathering, but adjusted to account for specific social and environmental sustainability indicators. This potential is constrained by factors like environmental side effects, biodiversity concerns, and social equity, ensuring that the deployment of ERW remains environmentally and socially responsible. Sustainability constraints are based on a thorough assessment of the impacts and trade-offs associated with enhanced weathering.

Spatial scale type: For which spatial scale is this estimate valid. Is it literally for one field, or an average for (the agricultural area) in a whole region or nation, etc.? Choose out of the possible choices indicating the scale of this estimate.
Sometimes data from experiments is based up to larger spatial scales. If it is the same number that is used, report it as the spatial scale for the experiment. If some modelling is done to adjust for other climate/soil type, etc. report it as the scale that was extrapolated to.

Spatial scale area: Enter the area this estimate is for.

Timescale: What is the timescale of the estimate?
Is it the weathering rate at this specific point in time? – in this case "instantaneous".
If it is the average over the time of the experiment, enter the length of the experiment. Some studies also average over 10 years or until a certain date, in this case enter the duration over which this is averaged. Some studies average over a time frame until an end date, in this case calculate the difference between the start of the study and this date.
If the timescale reflects an estimated, projected value for CDR in the future, enter "estimated annual future rate"
Note that the duration of the entire experiment should also be entered in section 6, and the duration for this specific datapoint in section 8.

Experiment < 1yr: Some studies are less than a year long, but linearly extrapolate the weathering rates they found to a year. For example, the experiments were 2 months long, and then they assume the weathering rate in mass/area/year is 6 times that. This can be problematic because it may bias estimates towards too high values because rates are usually highest in the beginning.
Yes, to some extent this is captured in previous categories, but since this is an important category, here you can supply some more data. If the study you are looking at is scaling short experiments to rate estimates per year, please specify the duration from which it is scaled up. Also specify if

this scaling up is somehow corrected for using a modeling approach. If the study is not doing such scaling, leave blank.

initial CDR before losses: If the study uses a soil-based approach like TiCAT that is derives CDR estimates by quantifying loss of feedstock, enter the CDR estimate here.
Do not enter studies that quantify CDR from the accumulation of SIC (unless they account for the loss of CO_2 due to the precipitation of SIC) – but do not enter SIC based estimates that purely consider the amount of C stored in SIC because the precipitation itself has caused losses. This can be complicated – please reach out to Jesper when in doubt how to enter an SIC based estimate.
Some studies also use a water-based approach (i.e., measures weathering products) to quantify CDR. "initial CDR" in this context means the CDR without accounting for further downstream losses (e.g. CO_2 evasion in rivers), though, depending on the methodology, some loss processes like strong acid weathering may already be included in this estimate.
This can also be a model-based estimate if the model afterwards explicitly accounts for loss pathways.

initial CDR method: What kind of MRV is used to arrive at the initial CDR estimate in the previous method?

Assumed fraction all CO_2 loss (field & downstream processes): Some studies simply assume that a defined fraction of gross CDR is lost as a result to downstream processes. Enter the fraction here (as a value from 0-1, not percent), if one single fraction is assumed for losses in fields (e.g. strong acid weathering), and downstream processes (e.g., CO_2 evasion in rivers).
Put the fraction with a negative number, i.e. 20% loss should be -0.20.

Assumed fraction downstream loss/assumed fraction field loss: Like the category above, but if there are separate values for field loss fraction and downstream loss fraction.
Put the fraction with a negative number, i.e. 20% loss should be -0.20.

Loss strong acid weathering: Add CO_2 loss relative to the maximum CDR potential that is caused by weathering of strong acids.
Enter as a negative number.

Loss SIC formation: Add CO_2 loss relative to the maximum CDR potential that is caused by precipitation of secondary carbonates **in soils**.
Note that some studies use the accumulation of SIC to quantify CDR. CO_2 is then considered in a relatively stable reservoir, and from this CDR value often no losses are subtracted – so for such studies you can enter this amount of CDR as the net CDR**.** Enter the same amount as net CDR also for Loss SIC formation because as bicarbonates precipitates into carbonates, the same amount of carbon is emitted as is stored in carbonates (again – when in doubt, reach out to Jesper).
Enter as a negative number.

Loss clay formation: Add CO_2 loss relative to the maximum CDR potential that is caused by precipitation of clays in soils – so this does not include, e.g., reverse weathering in marine sediments.
Enter as a negative number.

CO_2 loss in rivers: Enter CO_2 losses as a result of CO_2 evasion in rivers.
Enter as a negative number.

CO_2 loss mining: Enter the CO_2 that is lost relative to maximum potential CDR as a result of CO_2 emissions during mining.
Enter as a negative number.

CO_2 loss grinding: Enter the CO_2 that is lost relative to maximum potential CDR as a result of CO_2 emissions during grinding of feedstock.
Enter as a negative number.

CO_2 loss mining and grinding: Enter the CO_2 that is lost relative to maximum potential CDR as a result of CO_2 emissions during mining and grinding (if only the sum of both is reported, no need to add up if they are reported separately. If they are reported both individually and as sum still enter it).
Enter as a negative number.

CO_2 loss transport: Enter the CO_2 that is lost relative to maximum potential CDR as a result of feedstock transport.
Enter as a negative number.

Change in SOC/SOM: Enter the CDR effect of changes in soil biomass – if SOC/SOM decreases, this will be a negative value, if SOC/SOM increases this value will be positive.

change non-CO2 GHG (eCO2): Some studies also quantify the impact of EW on non-CO_2 green house gases, e.g. CH_4 or N_2O – often this is quantified as eCO_2. Enter this effect here – if there is a reduction in eCO_2, enter a positive value, else a negative value.

Net-CDR best estimate: The best estimate of net CDR of EW after all loss processes have been included as given in the paper.
If EW is a net source of eCO_2, put a negative value.

Net-CDR best estimate type: If the study is estimating CDR rates based on multiple datapoints, what does the value inserted as best estimate represent? Is it the mean of all data, the median, or maybe something based on the authors judgement (specify)? If the estimate is based on a single data point insert "only value assessed".

Significant: is the detection of net CDR statistically significant? If the 1SD uncertainties include 0, enter no. If no losses are accounted for, this question refers to the gross/initial CDR number. Please make sure to enter yes if it is because else this data may not be counted.

Net-CDR min/max range: Some studies also report a range of possible CDR rates – since this is not necessarily the same as uncertainty (which should be captured in associated uncertainty files to the database) but important information on the potential efficacy of ERW, these data should be

inserted here if given. If a study reports a best estimate with an uncertainty range, you can extract the min and max value of the given uncertainty here.
These values then do not need to be inserted into additional uncertainty files to the database anymore.

Biomass accounted: Does the net-CDR number account for changes in biological reservoirs/bio production or degradation.

Non-CO_2 accounted: Does the net-CDR number include effects of non-CO_2 green house gases

CDR plain language summary: In as simple terms as you can, describe what the paper has done to calculate (net) CDR. What is the primary estimate of initial CDR and what kind of losses/additional sinks are taken into account?

S3.10.3 CDR quantification and carbon budgets: mass CO_2 per area

These are essentially the same categories as for 10.2 but all in mass CO2/area. Please see section 10.2 for definitions of the categories.
Make sure to pay attention in particular to the timescale definitions because mass CO2/area estimates usually assume an underlaying timescale.

If the data is the average of some experiment or model for a certain period of time, enter this duration in the "timescale" category. If this timescale reflects the period until a certain date/year (e.g. estimates from the time of the study to 2050), make sure to also insert this date in the column "CDR potential estimate year" in addition to the duration of the estimate.

S3.10.4 CDR quantification and carbon budgets: mass CO_2 per time

This are essentially the same categories as for 10.2 but all in mass CO_2/time. Please see section 10.2 for definitions of the categories. Make sure to pay close attention to the spatial scale categories as per time estimates usually require assumptions on the scale of EW.

S3.10.5 CDR quantification and carbon budgets: mass CO_2 (cumulative global/field/regional CDR potential)

For this category usually both assumptions for spatial scale and timeframes are important, make sure to fill in these categories (see definitions in section 10.2). Often, CDR estimates in tons of CO_2 will estimate the cumulative potential over a certain period or until a certain date.

Hence, make sure to pay close attention to the spatial and temporal scale input categories.

S3.10.6 cost estimates

Cost: The cost as reported in the paper. In the "best estimate" category enter what is presented as the best guess in the paper, which may be the mean of several data points. If a range of cost estimates is presented, also enter the minimum and maximum values in the corresponding categories.

Cost in year X: What year is this cost estimate for? If it is not specified and a current estimate, simply enter the study year. If it is a price projection for some time in the future enter that year.

Cost base year: This is not meant to be for price estimates in a certain year, e.g. estimated price in 2050, but to adjust prices of past publications by inflation. Some studies for example adjust prices in the past or future to, e.g. the dollar value in the year 2000. If something like this is given, enter the base year for prices. If not specifically given leave.

Some studies may provide further information on the breakdown of the final CDR price; if so, the following categories may be relevant.

CAPEX (Capital Expenditure) cost: CAPEX refers to the upfront costs associated with the initial investment in infrastructure and equipment for enhanced rock weathering. This includes expenditures such as purchasing and installing machinery for grinding and spreading rock, transportation or rock powder spreading vehicles, storage facilities, and any other necessary infrastructure to launch ERW operations.

Fixed OPEX (Fixed Operating Expenditure) cost: Fixed OPEX refers to ongoing operational costs that do not change with the scale of ERW deployment. These may include expenses such as equipment maintenance, rent, salaries of permanent staff, and insurance. These costs remain constant regardless of the amount of rock being processed or spread and the CDR realized.

Variable OPEX (Variable Operating Expenditure) cost: Variable OPEX represents the operating costs that fluctuate depending on the scale of ERW activities. This could include fuel for transportation, energy costs for grinding and spreading rock, and materials such as replacement parts for machinery. These expenses increase or decrease based on the volume of rock processed and applied to the land, and to the CDR realized.

MRV Cost (Monitoring, Reporting, and Verification Cost): MRV cost refers to the expenses related to tracking, measuring, and verifying the effectiveness of enhanced rock weathering in capturing carbon, which may be considered an ERW specific part of OPEX. This includes costs for scientific monitoring equipment, data collection and analysis, reporting requirements to regulators, and verification processes to ensure that carbon sequestration claims are accurate and meet regulatory or certification standards.

Total OPEX (Total Operating Expenditure) cost: Total OPEX is the sum of both fixed and variable operational expenditures. It represents the overall ongoing cost of running ERW operations after the initial setup, encompassing both constant expenses (fixed OPEX) and scalable expenses (variable OPEX).

S3.10.7 additional LCA categories

This section captures some more details relevant to life cycle assessments of ERW. In this context, make sure you have also entered what type of feedstock was used (virgin material vs. some form of waste stream) in section 8.1.

Mining + grinding process: Enter any detail on the mining process if given in the pape as well as any notes on how rocks were grinded/milled/etc. into smaller particles.

Grinding required?: Enter if the feedstock required grinding before operation or if it was already ground small enough which may be the case for some waste feedstocks.

If specified, enter the GHG intensity of the energy used for mining and grinding.

Location of mine: please be as precise as possible.

Mode of transport: How was feedstock transported?

Transport distance: How far was feedstock transported?

Mode of application: How was the rock powder spread?

waste material treatment: If the study used waste streams, e.g., steel slag, or leftover quarry fines, how were these processes treated from an LCA perspective? Did the embodied emissions of the previous process count into the ERW LCA, or was 0 emissions assumed for waste streams?

Counterfactual considerations: Are any counterfactuals discussed, and if so, how are they treated?
For example, if ERW is done with mine wastes, is the weathering that would have occurred in the mine tailings anyways subtracted from the ultimate CDR estimate because it is not additional? This may also refer to discussions of ERW vs. liming.

life cycle missions: If given, report all life cycle emissions in t eCO_2 per tons of CO_2 removed. In a way, this is the most important LCA category – make sure to report this if it is contained in the study.

Life Cycle Impact Assessment (LCIA) Method(s): Some LCA studies use a standardized set of impact categories to quantify LCA impacts of processes. LCIA methods assign characterization factors to process emissions (air, soil, water) to quantitatively determine environmental impacts according to a set of impact categories. Impact categories describe different effects on the

environment, and may include things like climate change, acidification, or human health, each with a distinctive unit. Different LCIA methods include different sets of impact categories, and researchers may choose their LCIA method(s) based on which impact categories they deem most critical to their study's purpose or what region they're analyzing (as some methods have more robust characterization for certain geographic locations).

If literature contains standardized LCA, LCIA method(s) will likely be stated in the methodology section. If such impact categories are used, please choose the specific method used in this category, you may also add additional information on the assessed categories in the comment (and add it to the relevant discussions categories)

S3.11 CDR calculation details

This section is designed to capture some more technical details of the MRV approach, both in terms of what processes were considered and analytical approaches.

S3.11.1 MRV approach

water-based MRV approach method: What are the measured and primary quantities used for the carbon dioxide quantification? List all that apply.
Note that only the actual, measured proxies should be listed. For example if one study measures the release of Mg from feedstocks or Mg fluxes, but then uses the feedstock stoichiometry to calculate CDR based on other cations, too, only Mg should be listed here. This is true if other cations were technically measured, but not used as a primary means to calculate CDR. In this case please explain this as a note in the next category.

Water-based type: What kind of water sample was used.
Pore water: usually the case for in field samples, extracted by e.g., Rhizon soil samplers:
Soil core leachate: Mostly pot/core experiments where the water dripping out at the bottom is collected.
Ground water: select if a well is used to get water samples, typically the depth exceeds that of the soil layer.

Comment water based approach: Insert any information crucial to understand water based approaches, be brief.

Exchangeable-based MRV approach: List all species that were used as a directly measured proxy – see "Water based MRV approach above".

Comment exchangeable based approach: Insert any information crucial to understand exchangeable based approaches, be brief.

Soil-based MRV approach: Which soil (solid) based MRV approach was used?

immobile element used as proxy for feedstock addition: Many studies using soil-based MRV approaches use immobile elements to quantify feedstock addition in a given sample. List all that apply (i.e. that were directly measured/or assessed in a modeling study).

soil-based comment: Insert any information crucial to soil based MRV approach, be brief.

Gas-phase MRV: What direct gas measurements were used to constrain CDR?

soil-based comment: Insert any information crucial to gas based MRV approach, be brief.

MRV system boundary: What is chosen as the system boundary for the MRV calculations? If only the surface sediments or gas phase measurements were used, this is likely surface soil.

To a large extent, this will correlate to the phases that were measured (and depth of soil samples).

Possible values:

- (no value/leave blank)
- surface soil (specify depth of cut-off)
- deep soil (define depth of cut-off)
- river
- ocean
- deep sea sediments
- other (specify)

S3.11.2 Losses taken into account

This section aims to capture some more detail about the CO_2 losses taken into account.

For each section, list all that apply. Provide more detail in the respective comment sections.

S3.11.3 Analytical details

If a paper has several lines, the same data should be included in all lines where it applies.

This section aims to capture some more details of the analytical methods. Please choose all that apply for each section and add brief detail to relevant comment sections.

Sensors: Electrical sensors used or discussed in this study.

Analytical methods: Analytical methods used or discussed, enter all applicable possible values. For your reference here the full names of abbreviated methodologies. List all that apply to a certain line of data by inserting as a list into this cell, separating each by comma. As always, please make sure to copy and paste answer values to avoid typos.

- **ICP-MS (Inductively Coupled Plasma Mass Spectrometry)**: An analytical technique used to detect and quantify trace elements in soil samples. It involves ionizing the elements in a plasma state and then analyzing them based on their mass-to-charge ratios.
- **ID-ICP-MS (Isotope Dilution Inductively Coupled Plasma Mass Spectrometry)**: A specialized method within ICP-MS that utilizes isotopically enriched standards to accurately quantify the concentration of specific elements in soil samples.
- **XRD (X-ray Diffraction)**: A technique used to identify the crystalline structure and mineral composition of soil minerals by analyzing the diffraction patterns produced when X-rays interact with the soil sample.

- **XRF (X-ray Fluorescence)**: A method for determining the elemental composition of soil samples by measuring the characteristic X-ray emissions that occur when the sample is irradiated with high-energy X-rays.
- **Raman Spectroscopy**: A spectroscopic technique used to study the vibrational, rotational, and other low-frequency modes in soil samples. It provides information about molecular structure and composition.
- **BET (Brunauer–Emmett–Teller)**: A method used to measure the specific surface area of soil particles by adsorbing gas molecules onto the surface and analyzing the adsorption isotherms.
- **Aqueous sensors (e.g., conductivity, pH)**: Sensors used to measure various properties of soil solutions, such as electrical conductivity and pH, which provide insights into CDR and important soil characteristics.
- **TGA (Thermogravimetric Analysis)**: A technique that measures changes in the weight of a soil sample as it is heated, providing information about composition, moisture content, and thermal stability.
- **Microscopy**: The use of microscopes to visually examine soil samples at a microscopic level, allowing for the identification of soil components, structures, and microbial life.
- **Ion chromatography**: A method used to separate and quantify ions in soil extracts, providing information on nutrient content, ion composition, and pollutant levels.
- **SEM (Scanning Electron Microscopy)**: A microscopy technique that uses electrons to image the surface of soil samples at high resolution, revealing details of particle morphology and composition.
- **Particle size distribution**: The distribution of particle sizes in a soil sample, which is critical for understanding soil structure, porosity, and water retention.
- **Volumetric calcimetry**: A method for quantifying carbonate content in soil by volumetric titration.
- **AAS (Atomic Absorption Spectroscopy)**: An analytical technique used to measure the concentration of specific elements in soil samples by measuring the absorption of light at characteristic wavelengths.
- **ICP OES (Inductively Coupled Plasma Optical Emission Spectroscopy)**: Similar to ICP-MS, this method analyzes the elemental composition of soil samples based on the emission of light at specific wavelengths from excited ions in an inductively coupled plasma.
- **ICP AAS (Inductively Coupled Plasma Atomic Absorption Spectroscopy)**: Another variant of ICP-based spectroscopy used to quantify trace elements in soil samples based on their absorption of light at specific wavelengths.
- **FTIR (Fourier Transform Infrared Spectroscopy)**: A spectroscopic technique used to identify functional groups and molecular structures in soil organic matter and minerals based on their infrared absorption patterns.
- **LIBS (Laser-Induced Breakdown Spectroscopy)**: A technique that uses laser pulses to generate plasma on a soil surface, analyzing the emitted light to determine the elemental composition of the soil.
- **Sequential extraction**: The use of different consecutive chemicals to extract specific geochemical phases from the sediments.

- **Impact of sampling/pooling:** How does sample density and pooling of different subsamples into one sample affect accuracy and precision?
- **Thermo-gravimetric analysis**: A method used to analyze changes in mass as a function of temperature. It involves heating a sample incrementally while measuring the weight loss due to processes like decomposition, desorption, or oxidation.
- **Ramped combustion**: This refers to a controlled process of burning a soil sample at gradually increasing temperatures. It's often used to determine the organic carbon content of the soil by measuring the released gases or combustion residues.
- **Laser diffraction**: A technique that uses laser light scattering to measure particle size distribution in soils. By analyzing how light scatters as it passes through a dispersed soil sample, the particle size distribution can be determined.
- **X-ray scattering**: An analytical method that uses X-rays to probe the atomic and molecular structure of soils. This can provide information on mineral composition, crystallinity, and other structural characteristics of soil components.
- **Water flow test**: A test used to measure how water moves through soil under specific conditions, typically involving the application of water to a soil sample and monitoring its movement over time.
- **Dry combustion**: A method used to determine the total organic carbon content in soil by combusting the sample at high temperatures in an oxygen-rich environment and measuring the resulting carbon dioxide (CO2) emissions.
- **Walkley-Black method**: A widely used method for determining soil organic carbon content. It involves oxidizing the organic matter in a soil sample with a mixture of potassium dichromate and sulfuric acid, followed by titration to quantify the amount of oxidized carbon.
- **Gas flux chamber**: A device used to measure the exchange of gases (e.g., carbon dioxide, methane) between the soil surface and the atmosphere. By enclosing a portion of soil, changes in gas concentrations can be monitored over time.
- **Eddy covariance tower**: An instrument used to measure fluxes of trace gases (e.g., CO2, water vapor) between the earth's surface and the atmosphere. It operates by analyzing the turbulent exchange of gases through the measurement of wind velocity and gas concentrations at a height above the surface.
- Other (specify)

pH scale used: The concept of different pH scales arises from the need to tailor pH measurements to specific environmental conditions and chemical contexts, with each scale offering unique insights into acidity, alkalinity, ion activity, or solution composition based on varying measurement principles and considerations. Only if specified in the study, which of these pH scales was? If you put in a value, please also mention what calibration points were used (e.g., pH 3, 7, and 10).

- Total pH: Total pH refers to the overall pH of a solution, considering various forms of hydrogen ions present (free and bound), including those associated with sulfate ion [HSO4-]. Total pH provides a comprehensive measurement of hydrogen ions, accounting for complex interactions in solution, which may be important in aquatic chemistry and environmental studies.

- Free pH: Free pH measures the pH of a solution based solely on the concentration of free (unbound) hydrogen ions, excluding those bound to dissolved species or complexes (such as [HSO4-]). Free pH specifically focuses on the pH attributed to unbound hydrogen ions, providing insights into chemical equilibria and ion activities in solution.
- NBS pH (National Bureau of Standards pH): NBS pH refers to a historical pH scale developed by the National Bureau of Standards (now NIST) with specific reference materials and protocols for pH measurements. NBS pH is used for very low ionic strength solution medium. It is gradually replaced by more standardized and internationally recognized pH scales, such as the Total pH scale.
- Other (specify)

Data was extracted from graphs: A few of the studies require extracting data from graphs. If this is the case in this study / row of data, insert which categories/parameters where extracted from Figures in the study (in contrast to data being provided in table form).

Sequential chemical extraction purpose: Some studies may use specific chemicals to extract targeted sediment phases. If several of these are used in one study enter all that apply in the order they were performed by listing their purpose/target phase. Also specify if leaches were analyzed. For example, if one study first extracts exchangeable phases (not measured), then carbonate bound phases (measured), and then Iron and Mn phases (measured) enter:
Exchangeable, carbonate (measured), FeMn (measured).
Make sure to enter all relevant data, and specify for each line which sediment phase the data is referring to in section 6.3: soil phase assessed.

Sequential chemical extraction leachate: This essentially mirrors the preceding category, but instead of listing target phase/purpose, insert the used reagents. Following the same example, these would be:
1 M MgCl,, pH 7.0,1 M NaOAc adjusted to pH 5.0 with acetic acid (HOAc) (measured), 0.04 M NH20H-HCl in 25% (v/v) HOAc (measured)
Make sure to enter the data of each measured phase, and specify what leachate the line refers to in section 6.3: leachate used for analysis (reagent)

Used isotopes: list all isotopes that were either used in an experiment/model, or that were extensively discussed in a review.

S3.12 Discussion and main messages

This section is designed to capture the content of the discussion. The first catalogue discussed impacts on soils and heavy metals, while the last is designed to capture how much specific topics are a focus of the paper/discussion.

If a paper has several lines, the same data should be included in all lines.

S3.12.1 Discussion and side effects – topics discussed

The first two categories are "**impacts on soils discussed**" and "**environmental risks discussed**" – mention all categories that are discussed in the paper.

The last 2 categories relate to the discussion of side effects of ERW. List all side effects that are discussed in the paper in the relevant category (i.e., if they are mentioned in a negative or positive way). Please insert only those discussed effects that are a major part of the paper (i.e., occupy at least a paragraph in the discussion), not those that are just briefly mentioned. For example, many papers will claim positive impacts on soil quality and yield in the introduction without provided proof or discussing it – in this case do not mention this side effect. The side effect needs to be a major part of the study outcome itself for it to be should listed.

The different side effects are defined in more detail in section 13 of the code book (where there is also space to provide more detail in the database). Decision if side effects are mentioned in a negative or positive way should follow the following rules.

Negative:

- The study highlights concerns or emphasizes risks associated with the side effect.
- The side effect is discussed in terms of harmful impacts, such as soil degradation, negative effects on human health, increased emissions, or reduced biodiversity.
- Findings suggest the side effect may pose significant challenges or cause unintended consequences, such as heavy metal accumulation or water quality deterioration.
- The study's conclusions focus on preventative measures, mitigation strategies, or potential regulatory action to minimize the side effect.
- Recommendations or policies based on the side effect are cautious or discouraging, suggesting that the risks are too high or not worth the trade-off.

Positive:

- The study explicitly supports or emphasizes benefits related to the side effect.
- The overall findings suggest that the side effect leads to improvements in relevant areas (e.g., environmental health, productivity, sustainability).
- The side effect is highlighted as contributing to positive outcomes such as increased soil fertility, improved food quality, better water quality, enhanced biodiversity, or reduced emissions.
- Potential risks are mentioned but are minimal or well-mitigated, with the benefits clearly outweighing the negatives.

- Recommendations or policies based on the side effect are favorable, suggesting encouragement or promotion of the practice.

If a study treats a side effect in a neutral way that discusses both positive and negative effects in a balanced way you should enter the side effect both in the positive and negative category.
Such studies present mixed findings, with no clear bias toward positive or negative outcomes.

- The side effect is discussed as a trade-off, meaning both positive and negative aspects are mentioned without one being dominant.
- The research does not indicate significant harm or benefit, suggesting that the side effect might be context-dependent or inconclusive.
- No major risks or benefits are identified, or the side effect is treated as insignificant to the overall outcomes.

S3.12.2 heavy metals

List all values that apply.

Finds accumulation: Remember to put in negative results. If the study does not look at it, leave blank. It does but finds no accumulation, put no.

Mitigation discussed: Does the study discuss or test how potential heavy metal contamination can be prevented or reversed?
Phytoremediation/Phytoprevention refers to using hyper-accumulating plants that remove heavy metals from the field after (phytoremediation) or to prevent contamination (phytoprevention)

Limit on ERW: Does the study discuss the extent to which heavy metal contamination may limit CDR deployment? Potential? If so, add a few words, and try to also specify what rock, what metal, and what location it is referred to, e.g.:
<150 t/ha CDR from basalt before Cu thresholds are broken in England.

S3.13 discussion topics

This section aims to capture different topics of the discussion, which are defined below. Some of this data is captured in quantitative form in the previous sections, but they are included here to provide more qualitative detail as well as an indicator of how much a certain topic is the focus in literature. **If a paper has several lines, the same data should be included in all lines.**

For each topic, there are two fields.

The first one is an indicator of how much a certain topic is the focus of a study (called indicator "category name"). These categories have several purposes. First, they make it easy for database users to see which papers really focus on a certain topic. Furthermore, they make possible to assess how similar different people have filled in data for the same paper: text inputs can't be easily compared, but numbers can easily be used to make statistic statements on reliability of the database. Furthermore, this field defines whether more info is required in the second category for each topic. The possible values for each topic are:
(no value/leave blank): This topic is not important for the discussion of the topic or interpretation of results. It may still be mentioned in passing in the introduction, this is irrelevant.
1: This topic is a major focus of the discussion, or the topic is extremely important to interpret the data/findings. Usually, this topic will have a dedicated section in the discussion – more than a sentence or two as part of a paragraph.
2: This is literally the main topic of the paper. Usually, this means the study contributes new findings to this topic. As a result, in most cases, each paper should not have more than 1 (maybe 2-3 in slightly overlapping categories) values of "2" in these discussion sections. For example, papers whose whole purpose is to assess the impact of particle size on ERW CDR would get a 2 only in this category.
Some categories are also sub-categories of others, for example there is a general life cycle assessment (intensity of energy used) category, and then several LCA relevant sub-categories like for example raw material extraction and material transport. In such a case you would give the main category a "2" and the sub-categories a "1" if several of them are discussed.

The second category for each topic is meant to capture some details on the findings of the study (called content "category name"). This category needs to be filled out of the previous category has a value (1 or 2, rather than no value) Try to capture as much of the content as possible with as few words as possible, i.e., not more than one sentence.

The discussion topics are sub-divided into several categories (13.1-13.9).

S3.13.1 Side effects – soil & agronomy

- **Soil CO_2 emissions**: Emissions of carbon dioxide from soil because of biological and chemical processes that occur during the application of ERW. This includes the direct or indirect release of CO_2 from soil respiration, organic matter decomposition, or changes in soil chemistry triggered by the introduction of rock materials.

- **Soil non-CO_2 GHG emissions**: Emissions of greenhouse gases other than CO_2, such as methane (CH_4) and particularly nitrous oxide (N_2O), caused or avoided by the use of ERW as a carbon dioxide removal (CDR) measure. These emissions may arise from soil microbial activity, changes in land use, or interactions with agricultural practices influenced by ERW.
- **Impact on soil organic carbon stocks**: The effect of ERW on the amount of carbon stored in soil organic matter, including any increases or decreases in soil carbon due to changes in microbial activity, decomposition rates, or other factors influenced by the application of rock materials. May also include changes from organic matter to DOC or impacts on mineral associated organic matter (MAOM) or phytolith carbon (PhytoOC).
- **Impact on soil inorganic carbon stocks**: The effect of ERW on the amount of carbon stored in soil inorganic forms, such as carbonate minerals, and the potential for changes in carbon sequestration or release due to chemical reactions between rock minerals and the soil environment.
- **Heavy metal (HM) accumulation and bioavailability in soils**: Monitoring and addressing the accumulation of potentially harmful heavy metals in soils resulting from ERW processes as well as their bioavailability. This also includes discussion of possible prevention and mitigation strategies.
- **Impact on soils:** Understanding how ERW activities may alter soil properties, structure, and fertility over time. This primarily relates to any properties not captures above (e.g. SIC and SOC stocks) such as any physical, chemical, or biological properties such as pH, soil structure, soil temperature, soil water holding capacity, water infiltration and hydrological properties, CEC, etc.
- **Agronomic impacts of ERW:** Assessing the effects of ERW applications on soil fertility, nutrient cycling, crop yields, and agricultural productivity. This also includes discussion of rock flour as fertilizer (not directly relating rock flour use to CDR) which is particularly relevant for the large body of early literature where this, rather than ERW, is the primary focus.
- **Impact on food quality**: Assessing the influence of ERW-derived minerals or elements on the nutritional content, safety, and quality of food crops grown in treated soils.

S3.13.2 Side effects – air & water

- **Air quality**: Any impacts on air quality resulting from ERW activities. This includes direct pollution with fine particulate matter/dust because of ERW application but also changes in ground-level ozone levels, sulfur dioxide (SO_2), nitrogen oxides (NO_x), or other air pollutants that could arise from the mining, transportation, or application of ERW materials.
- **Noise pollution**: The generation of audible noise associated with the implementation of ERW measures, including noise from mining, transportation, or field application of rock materials. This category assesses the potential disruption to communities, wildlife, and natural habitats.

- **Water quality**: Changes in water quality resulting from ERW activities, such as reductions in nutrient runoff, increased pH stabilization. Impacts may extend to discussion of ecosystem health in water bodies.
- **Heavy metal (HM) accumulation and bioavailability in waters**: Monitoring and addressing the accumulation of potentially harmful heavy metals in effluent water resulting from ERW processes as well as their bioavailability. This also includes discussion of possible prevention and mitigation strategies.
- **Water demand**: The amount of water required for ERW processes, including the mining, processing, and application of rock materials. This category considers the potential strain on local water resources and competition with other water uses, particularly in water-scarce regions.
- **Underwater light attenuation in rivers or oceans**: The reduction in light penetration through water columns as a result of ERW measures, particularly due to the addition of suspended particles or changes in water chemistry. This can impact aquatic photosynthesis, nutrient cycling, and the health of marine ecosystems.

S3.13.3 Side effects – plants and animals

- **Biodiversity**: The positive effect of ERW on species richness, ecosystem diversity, and genetic variation within affected environments. This includes potential positive impacts on pollinators, ecosystems, and habitat health due to changes in soil properties, water quality, and land use.
- **Heavy metal (HM) accumulation in plants or animals**: Monitoring and addressing the accumulation of potentially harmful heavy metals in animals and plants. This also includes discussion of possible prevention and mitigation strategies.
- **Impact on terrestrial plants** : Evaluating the effects of ERW applications on plant growth, health, and pest resistance, as well as potential indirect impacts on animal life and habitats.
- **Impact on soil biota – positive/negative**: Evaluating the effects of ERW applications on soil organisms such as earthworms, microbes, microbiomes in general, fungi, mycorrhiza etc.
- **Impact on other terrestrial animals**: Evaluating the effects of ERW applications on animal life and habitats, e.g., through changes in agricultural practices or the direct impact on rock flour/dust on organisms.
- **Impact on aquatic organisms:** Examining the potential effects of ERW activities on marine (and riverine) ecosystems, and the health of aquatic organisms, considering factors such as changes in water chemistry (e.g., carbonate saturation state) and habitat alteration (e.g., due to increased sediment load, carbonate precipitation, and/or sedimentation).

S3.13.4 Side effects – human impacts & factors

- **Public acceptance/perception**: How ERW is perceived by the general public and stakeholders, and the level of societal backing or opposition towards its adoption and implementation. This also includes discussions on how ERW is discussed in media.
- **Policy acceptance**: The degree to which ERW is accepted or supported within political discourse. This includes the perspectives of policymakers, political parties, and other influential actors on whether ERW should be integrated into climate action plans and regulatory frameworks.
- **Policy response**: The actions taken by governments and policymakers to incorporate ERW into national climate strategies. This may include funding for research and development, legislative measures, regulatory support, and the incorporation of CDR in emissions reduction targets and climate mitigation policies.
- **Employment**: The effects of ERW on job creation and the labor market, including direct employment in sectors such as mining, transportation, and application of rock materials, as well as indirect employment through the development of technologies, infrastructure, and research in the CDR field. This category assesses both potential job growth and shifts in employment patterns due to the adoption of ERW.
- **Impact on human health:** Assessing the potential direct and indirect effects of ERW activities on human health, including exposure to minerals or elements released during weathering processes, changes in food quality and safety, as well as broader implications for community well-being and public health.
- **Legal dimensions**: The legal frameworks (incl. regulation of waste products/feedstocks), regulations, and considerations that impact the implementation, operation, and accountability of ERW initiatives. Also includes discussion of possible legal conflicts.
- **SDGs (Sustainable Development Goals)**: A set of global goals adopted by the United Nations to address various social, economic, and environmental challenges, with ERW potentially contributing to several of these goals.
- **Climate justice & ethics**: Considerations of fairness, equity, and moral principles in the distribution of benefits and burdens associated with ERW, particularly in relation to vulnerable communities and future generations.

S3.13.5 Market questions, LCA, costs, etc.

- **LCA (Life Cycle Assessment):** A methodology for evaluating the environmental impacts of ERW throughout its entire life cycle, from extraction of raw materials weathering and CDR. This is the main umbrella category to be used for LCA, with several of the following categories going into more details.
- **Raw Material Extraction:** The impacts associated with the extraction and processing of rocks or minerals used in ERW, including energy consumption, land use, water use, and emissions from mining activities. If the study discusses technological readiness level for this part of the supply chain, mention the TRL here.
- **Feedstock Transportation:** The impacts from the transportation of rock materials from mining or processing sites to areas of application, considering emissions, fuel

consumption, and infrastructure usage. If the study discusses technological readiness level for this part of the supply chain, mention the TRL here.

- **Feedstock Processing:** The impacts related to crushing, grinding, and preparing rock materials for ERW, including energy use, emissions, and byproducts from mechanical and chemical processes. If the study discusses technological readiness level for this part of the supply chain, mention the TRL here.
- **Feedstock Application:** The impacts associated with the distribution and application of crushed rock materials to agricultural land or other landscapes, considering machinery use, fuel consumption, emissions, and labor. If the study discusses technological readiness level for this part of the supply chain, mention the TRL here.
- **Energy Use:** The total energy requirements across the entire ERW process, from mining and processing to transportation and field application, and the corresponding environmental impacts such as greenhouse gas emissions.
- **GHG emissions per ton of CDR:** Discussion of the total GHG emissions produced per ton of carbon dioxide removed by ERW, accounting e.g., for emissions from mining, processing, transportation, and application of the rock material as well as operational emissions.
- **Costs of ERW**: Analysis of the financial expenses associated with implementing and scaling up ERW initiatives, including capital costs, operational expenses, and maintenance. If given, provide detail on the shares of CAPEX (see 11.6 for definitions), OPEX, MRV, etc.
- **Further Economic Impacts:** The additional economic effects of ERW, both positive and negative. This includes the generation of economic value through increased wages, tax revenues, land value appreciation, and labor attraction, as well as any adverse economic impacts such as potential costs to local economies, shifts in land value, or job displacement.
- **Technological Readiness Level of ERW and Supply Chain Components:** The current state of technological development for ERW and its supply chain, including the maturity of methods and technologies for sourcing, transporting, processing, and applying rock materials. This category also assesses the technological advancements needed to scale ERW effectively and the associated carbon footprint of the supply chain.
- **Supply Chain and Logistics:** The logistical challenges and infrastructure requirements for sourcing, transporting, processing, and applying rock materials for ERW. This includes the associated carbon emissions, energy use, and any limitations or efficiencies within the supply chain.
- **Economic Incentives and Funding Mechanisms:** The financial incentives, funding opportunities, and market-based mechanisms that support the adoption and scaling of ERW. This includes the role of carbon markets, subsidies, and alternative funding models, as well as discussions on the cost distribution of ERW and potential policy frameworks. This also includes exploring alternatives to the voluntary carbon market and how they could play out with enhanced rock weathering
- **Voluntary carbon market design**: The structures and mechanisms governing the buying, selling, and trading of ERW-related products or services in the voluntary carbon market.

- **Stakeholder Engagement and Collaboration:** The processes and strategies for involving various stakeholders—such as governments, NGOs, industries, and local communities—in the planning, implementation, and scaling of ERW. This category emphasizes the importance of public-private partnerships, multi-sectoral collaboration, and transparent communication.
- **Available Land/Land Use Change:** The land requirements for implementing ERW, including areas used for mining, processing, and applying rock materials. This category also considers land use competition with agriculture, conservation efforts, and other land-based activities, as well as the spatial footprint of mining operations for ERW feedstock.
- **Standardizatio**n: Establishing uniform criteria, protocols, or benchmarks to ensure consistency and comparability within ERW practices or technologies.

S3.13.6 Efficacy of ERW

- **CDR potential (global/regional) - rate**, GtCO2/yr: Estimations of the global or regional carbon dioxide removal potential achievable through ERW activities, measured in gigatons of CO2 per year.
- **CDR potential (global/regional) - cumulative**, GtCO2: Estimations of the global or regional carbon dioxide removal potential achievable through ERW activities, measured in total gigatons of CO2 by a certain date.
- **Differences in Technical, Economic, Sustainable, and Rock Feedstock CDR Potential:** The distinctions between the various carbon dioxide removal (CDR) potentials of ERW, including technical (theoretical maximum achievable CDR), economic (practical CDR based on cost-effectiveness), sustainable (CDR aligned with environmental and social sustainability criteria), and rock feedstock (CDR potential based on the type and availability of rock materials used). This category assesses how these factors influence the overall scalability and feasibility of ERW.
- **Role of feedstock type**: The influence of different types of rocks or minerals used as feedstock in ERW processes on effectiveness, efficiency, and environmental impacts.
- **Impact of feedstock particle size**: How the size of rock or mineral particles utilized in ERW affects reaction rates, application methods, and overall effectiveness.
- **Impact of feedstock application rate:** The effects of varying application rates of rock or mineral feedstock on ERW efficiency, costs, and environmental outcomes.
- **Impact of climate on ERW**: How climatic factors such as temperature, precipitation, and humidity influence the effectiveness and kinetics of ERW reactions. Also includes how future climate change may impact the efficacy of ERW.
- **Impact of soil type on ERW**: How different soil compositions and characteristics affect the efficacy and outcomes of ERW applications.
- **Impact and processes in deep soil horizons:** Investigating the effects of ERW on soil characteristics, carbon sequestration, and nutrient cycling in deeper layers of the soil profile, considering the potential implications for long-term carbon storage and the implications of carbon release in deep soil horizons (e.g., if secondary phase formation releases CO_2, does this enter the atmosphere?).

- **Weathering rates & limitations**: The rates at which rocks or minerals undergo weathering reactions in ERW processes, as well as factors that may constrain or enhance these rates. While this is a broader umbrella category for all studies assessing feedstock dissolution, it also explicitly encompasses studies focusing on limitations to weathering rates such as surface passivation, saturation of mineral phases in pore waters, etc.
- **Agreement between models and measurements/field experiments**: The study assesses for example in how far models are able to recreate measurements/experiments, and possible sources for disagreement.
- **Additionality of ERW**: Determining whether ERW activities result in net carbon removal beyond what would have occurred naturally or through other means.
- **Impact of strong acid weathering on ERW**: How do strong acids (e.g., from fertilizer) interact with the feedstock, what is the impact on CDR, and how is this treated from a counterfactuals perspective?
- **Lag time of weathering signals**: The delay between the initiation of ERW activities and observable changes in effluent composition, soil changes, and, primarily, the realization of actual CDR due to, for example, cation exchange processes, secondary phase formation, etc.. This also includes all discussions of time considerations, i.e., discussions of ex-post and ex-ante credits, and in general how long it takes for certain fractions of CDR potential to be realized.
- **Permanence and Risk of CDR reversal:** The durability of carbon storage achieved through Enhanced Rock Weathering (ERW), including the timescales over which sequestered carbon remains locked away. This category generally considers the risk of carbon reversal with the next topics going into more details.
- **Carbon leakage in soils**: Processes that may release CO_2 in soils as a response to ERW, for example strong acid weathering, formation of secondary phases, etc.
- **Carbon leakage in rivers (+ transport bottle necks)**: Loss of CO_2 in rivers as a response to ERW, for example by re-equilibrium of the carbonate system, degassing, or precipitation of secondary phases. Includes discussions on the ability of rivers to actually transport ERW products.
- **Carbon leakage in oceans**: the potential for carbon dioxide release or loss from marine environments and considerations regarding the sequestration of ERW-derived carbonates in ocean sediments.
- **Non-CO_2/GHG climate change impacts**: The indirect effects of ERW on climate change through mechanisms other than CO_2 removal, such as changes in land cover and land use, surface albedo (reflectivity, both due to changes in land surface albedo but potentially also reflective aerosols both directly and indirectly), and other factors influencing the Earth's energy balance. These impacts could either mitigate or exacerbate climate change beyond carbon sequestration.

S3.13.7 MRV (Measurement, reporting, and verification)

- **Development of new MRV approaches**: Development of a new type of MRV for ERW; include the new approach as well as its benefits.

- **Discussion of (different) soil based MRV approaches:** Exploring (potentially multiple) methods and techniques used for the measurement, reporting, and verification of ERW efficacy, including any approaches based on solid soil samples – e.g. TiCAT, based on SIC, etc.. Also include approaches based on soil CEC here.
- **Discussion of (different) water based MRV approaches:** Exploring (potentially multiple) water based methods and techniques used for the measurement, reporting, and verification of ERW efficacy, including approaches based on effluent water or pore water using any proxy for dissolution (alkalinity, cations, conductivity, etc.).
- **Discusses sensitivity of MRV approaches/signal to noise issues:** Examining the degree to which MRV methods are sensitive to changes in ERW-related signals and their ability to distinguish true signals from background noise or confounding factors, considering challenges such as data variability, uncertainty, and measurement limitations.
- **Soil heterogeneity as a challenge to MRV:** Explores the extent to which natural variability in soils is an obstacle to accurate quantification of CDR using ERW. This includes studies assessing the impact of different protocols to pool samples to mitigate soil heterogeneity in measurements, the amount of minimum samples required for sufficient signal to noise ratios, as well as statistical or numerical approaches to deal with soil heterogeneity.
- **Cost of MRV (per ton of CDR):** Explores the cost of MRV for ERW related to the cost of 1 t of CDR, as well as the extent to which related costs may be prohibitive for ERW to scale or how MRV costs could be brought down. Mention the absolute costs as well as relative cost compared to the price of CDR if given.

S3.13.8 Synergies and antagonisms with other CDR approaches and regenerative processes

- **ERW in combination with biochar**: Exploring synergies, trade-offs, and potential combined effects when integrating ERW with biochar (both as a carbon dioxide removal strategy as well as for heavy metal management etc.).
- **ERW in combination with BECCS**: Exploring synergies, trade-offs, and potential combined effects when integrating ERW with BECCS.
- **ERW in combination with agroforestry**: Exploring synergies, trade-offs, and potential combined effects when integrating ERW with agroforestry.
- **ERW in combination with afforestation/reforestation**: Exploring synergies, trade-offs, and potential combined effects when integrating ERW with afforestation/reforestation.
- **ERW in combination with SOM management**: Exploring synergies, trade-offs, and potential combined effects when integrating ERW with maximization of soil organic matter content. This category is quite broad and can encompass for example discussion of synergies with no-till agriculture, use of perennial crops, cover crops, etc.
- **ERW in combination with other CDR approaches**: Exploring synergies, trade-offs, and potential combined effects when integrating ERW with other carbon dioxide removal strategies not explicitly mentioned above.

- **ERW in combination with phytomining**: Exploring synergies, trade-offs, and potential combined effects when integrating ERW with phytomining – the intentional and economic recovery of metals from soils using hyper accumulating plants.
- **Discussion of how ERW ties in with other regenerative agriculture approaches**: Exploring the intersections between ERW and practices aimed at restoring soil health, biodiversity, and ecosystem services within agricultural systems not explicitly mentioned above.
- **ERW in comparison to other CDR approaches**: Explores strengths and limitations of ERW in comparison to other CDR approaches and its potential to contribute to mitigating climate change as part of a wider CDR portfolio.

S3.13.9 Other

- **Uses/discusses natural analogues**: Drawing comparisons or insights from natural geological processes or occurrences relevant to ERW, such as weathering rates in specific geological formations or landscapes.
- **Urban Soils:** The potential for engineered or natural urban soils to serve as carbon sinks, either by enhancing their carbon storage capacity. This includes examining the role of urban soils influenced by factors such as concrete residues and assessing their current contribution to carbon sequestration in urban environments, as well as how this could be optimized in the future.
- **Climate Change Resilience:** The role of Enhanced Rock Weathering (ERW) in boosting the capacity of ecosystems, agriculture, and communities to adapt to and cope with the impacts of climate change. ERW could enhance climate resilience, e.g., by improving soil water holding capacity, health, increasing nutrient availability, and supporting sustainable agricultural practices that help ecosystems withstand climate-related stresses such as drought, flooding, and temperature extremes. This category includes how ERW contributes to climate-smart agriculture, adaptation strategies, and long-term sustainability in the face of changing environmental conditions.
- **Economic/Social Resilience:** The ability of economic and social systems to absorb shocks, adapt to disruptions, and maintain stability and growth. This includes diversifying resources (e.g., fertilizer sources), creating new employment opportunities, and fostering more resilient supply chains and local economies through sustainable practices such as ERW. It focuses on ensuring that communities and economies can thrive amid external pressures or changes.
- **Other**: Add anything that has not been captured by the discussion topic. The indicator should be at least 2 to warrant this category.

S3.14 Comment

If you feel like there is something important that has not been captured by the database, feel free to add any comment here. This can either be for a whole paper, or specific do an individual line of data. Please try to be brief.