Enhancing natural cycles in agro-ecosystems to boost plant carbon capture and soil storage

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Abstract

One of society's greatest challenges is sequestering vast amounts of carbon to avoid dangerous climate change without driving competition for land and resources. Here we assess the potential of an integrated approach based on enhancement of natural biogeochemical cycles in agro-ecosystems that stimulate carbon capture and storage while increasing resilience and long-term productivity. The method integrates plant photosynthesis in the form of (cover) crops and agroforestry which drives carbon capture. Belowground plant-carbon is efficiently stored as stable soil organic carbon (SOC). Aboveground crop and tree residues are pyrolyzed into biochar, which is applied to the soil reducing carbon release through decomposition. Enhanced weathering of basalt powder worked into the soil further captures and stores carbon, while releasing nutrients and alkalinity. The integrated system is regenerative, through enhanced virtuous cycles that lead to improved plant capture, biomass storage and crop yield, the prerequisites for large-scale carbon sequestration along with food security.

Introduction

Human-induced climate change has significant adverse impacts on our environment, economy, and way of life. Reductions of carbon dioxide emissions alone are no longer sufficient to avoid dangerous impacts (Allen et al., 2018; Hansen et al., 2017), and capture plus long-term storage of atmospheric carbon (so-called "negative emissions") will be required.

Large-scale carbon sequestration is possible through a range of options, each with its own advantages and drawbacks (Fuss et al., 2018; Hepburn et al., 2019; Roe et al., 2019; Smith, 2016; Smith et al., 2016). One family of methods centres on enhancing natural biogeochemical processes. These techniques also have positive environmental impacts (Beerling et al., 2020; Fuss et al., 2018; Paustian et al., 2016; Smith, 2016; Smith et al., 2016), and could (partly) pay for themselves by increasing natural capital and agricultural productivity (Hepburn et al., 2019). Examples include (1) boosting the growth and standing carbon stock in plants in cropping and pasture systems through cover- and inter-cropping (e.g., agroforestry); (2) re-establishing and/or enhancing soil organic carbon (SOC) stocks (Pausch & Kuzyakov, 2018); (3) production of biochar, which is plant biomass transformed at elevated temperatures under oxygen-limited conditions (pyrolysis) into a recalcitrant form that withstands decomposition for many decades/centuries to possibly even millennia (Lehmann & Joseph, 2015); and (4) increasing the inorganic carbon sink in soils via Mg and Ca silicate weathering by working finely ground rock (basalt) into soils (Beerling et al., 2018).

The combined global carbon sequestration potential of such measures has been estimated at 0.3-6.8 Gt C yr^{-1} (Smith et al., 2020). The potential of each technique independently has been reported in Smith et al. (2020), who compiled the full range of literature values:

(1) Agroforestry: ~0.03-1.55 Gt C yr^{-1}
(2) Soil carbon sequestration (SOC): 0.14-1.36 Gt C yr^{-1}
(3) Biochar: 0.01-1.80 Gt C yr^{-1}
(4) Enhanced rock (basalt) weathering: 0.14-1.1 Gt C yr^{-1}.

Large-scale carbon sequestration is an enormous challenge in itself, and doing so without competition for land and resources among different carbon sequestration techniques and with food production is an even greater one (Fuhrman et al., 2020; Smith et al., 2013, 2019). Here we evaluate an integration of the aforementioned land-based carbon sequestration techniques in agricultural systems on the same land area (Figure 1). This avoids competition for land and resources among drawdown methods, and further helps to build resilient and regenerative agro-ecosystems. Importantly, we contend that interactions between methods and with soil processes can set up synergistic virtuous cycles that further enhance the potential for carbon sequestration. This study aims to (i) discuss the key limitations of individual carbon sequestration techniques by themselves, a prerequisite to maximise their potential; (ii) assess interactions and synergies between the techniques; and (iii) define conditions and strategies that allow for integration and large-scale carbon sequestration in agro-ecosystems.

Definitions and key limitations of individual techniques

Plants

Plants are the central players in the assessed land-based carbon sequestration system (Figure 1). They capture CO_2 and convert it into sugars that are translocated throughout the plant and soil. Eventually, plant carbon enters the soil from aboveground litter, and from roots and their rhizodeposits (Figure 2). Typically, practices that increase aboveground biomass also accumulate SOC; plant productivity and the size of the SOC pool are linked (Smith et al., 2008).

Adding trees to agricultural land and consequently conversion of crop- and grassland into agroforestry, a form of inter-cropping (the integration of at least two plant species in the same area), can increase aboveground biomass more than 10-fold and has been found to increase SOC stocks by 25% and 19% globally, respectively (Cardinael et al., 2018; Chapman et al., 2020). Cover crops (the establishment of plants for the purpose of protecting the soil) also boost aboveground carbon stocks throughout the year and can increase SOC stocks by 0.1-1 t ha^{-1} yr^{-1} (Blanco-Canqui et al., 2015; Poeplau & Don, 2015). Plant and soil carbon storage increases with plant species-richness due to higher niche partitioning, and thus nutrient and water use efficiencies (Bayala & Prieto, 2019; Brooker et al., 2015; Cardinale et al., 2007).

Globally, plant biomass accumulation is limited by nutrients and water (Fatichi et al., 2020; Terrer et al., 2019). Plant carbon can accumulate quickly, but the system then starts to saturate (Luyssaert et al., 2008) and the captured carbon dioxide can even be released, for example by fires, land-use change, and climate change (Table 1) (Law et al., 2018). If undisturbed, however, plant carbon is stable for >100 years (Luyssaert et al., 2008), the typical timescale for climate-change predictions (IPCC, 2013).

Soil organic carbon (SOC)

Microorganisms degrade plant carbon (respiring CO_2), but also foster conversion into stable forms of SOC (Castellano et al., 2015; Kallenbach et al., 2016) (Figure 2a). Both processes are affected by the activity, abundance, and community composition of microorganisms and are soil dependent (Luo et al., 2017). To achieve long-term sequestration of plant-derived carbon, a simple increase in total SOC content is insufficient. Instead, an increase is needed in persistent SOC stocks, through protection in soil microaggregates (aggregate occlusion), and/or carbon-binding to clay and silt particles (mineral-associated SOC/matrix stabilisation) (Cotrufo et al., 2013; Smith et al., 2008). Therefore, the soil needs to possess sink strength in the form of available minerals or soil aggregation to build stable SOC (Figure 2). Aggregate protection typically stabilises SOC on decadal time scales, while mineral matrix stabilisation can protect SOC for centuries (Cotrufo et al., 2013).

Similar to plant biomass, SOC levels reach saturation and can be disturbed, for example through overgrazing, land-use change, and climate change (Castellano et al., 2015; Luyssaert et al., 2008; Poeplau & Don, 2015; Smith et al., 2008). Besides biomass input and the availability of sink strength, stable SOC accumulation depends on the conversion efficiency of plant carbon into SOC, here defined as the carbon sequestration efficiency (CSE) (Figure 2), and the rate of SOC degradation (Anthony et al., 2020; Smith, 2008). The microbial growth efficiency (carbon use efficiency) defines the proportion of plant carbon that is converted into microbial biomass and stored, *versus* the proportion that is decomposed and released as CO_2 via heterotrophic respiration (Anthony et al., 2020). The microbial carbon can subsequently be stabilised into other forms of SOC (mineral-associated SOC mainly) (Sokol et al., 2019). Both processes combined make up the CSE as defined here. In most agricultural systems, only a small proportion of aboveground plant carbon is transformed into (stable) SOC by biological processes; the CSE is low at only ~8% (Jackson et al., 2017) reflected in Figure 2a as 4% of the overall plant carbon stabilised (8% of the 45% carbon as shoot biomass).

Biochar

During biochar production (pyrolysis), biomass is heated in the absence of oxygen, which directly converts the atmospheric carbon that was captured by plants into a form that is stable for centuries (Lehmann & Joseph, 2015) (Figure 2b). The process results in an initial release of ~45% of the plant carbon stored in agricultural and forestry residues (mean over different temperatures) (Crombie & Mašek, 2015) and, hence, in greater carbon emissions in the first few years of biochar production, relative to regular biomass decomposition (negative values in Sup Figure 1). However, over subsequent years, this is offset, as further decomposition emissions are avoided, and net carbon-negative conditions develop. The mean residence time of biochar has been estimated at 500-1000 years, several orders of magnitudes greater than that of unpyrolysed biomass (Budai et al., 2016; Lehmann et al., 2015; Wang et al., 2016). Assuming a ~60-times lower degradation rate of biochar than unpyrolysed biomass (Budai et al., 2016), biomass pyrolysis becomes net carbon negative after ~3-5 years (Sup Figure 1).

Biochar use is limited by biomass feedstock availability and processing costs. For example, it can be essential to leave crop residues in the field to reduce soil erosion and evaporative losses in water-limited regions (Klocke et al., 2009). In other cases, some (bioenergy) crop and forestry residues are well suited for biochar production (Sessions et al., 2019). Globally, wheat, for example, had annual grain yields of 0.4-9.4 t ha^{-1} in 2019 (mean 3.3 t ha^{-1}; range/mean of all countries listed in database) (Food and Agricultural Organization of the Unites Nations, 2020). With a typical harvest index of 0.5 (50% of biomass in grain, 50% into stem and leaves) (Fischer, 2007), 0.4-9.4 t ha^{-1} of wheat straw residue is produced annually on-farm. Tree plantations can produce ~10-100 t ha^{-1} of residue over a 30-40 year rotation, equivalent to 0.25-3.3 t ha^{-1} yr^{-1} (Ghaffariyan & Apolit, 2015; Gregg & Smith, 2010; Wrobel-Tobiszewska et al., 2015). The biochar yield from woody and grass feedstocks is ~25% on average across different pyrolysis temperatures (Mašek et al., 2019). Hence, pyrolysis of wheat straw and pine plantation residues produces 0.1-2.4 and 0.06-0.8 t ha^{-1} yr^{-1} of biochar, respectively (mean 0.8 and 0.4 t ha^{-1} yr^{-1}). We thus infer that limited on-site availability of biomass residues in agriculture and neighbouring forestry systems will initially enable biochar application rates of ~1 t ha^{-1} yr^{-1}, which corresponds to 0.73 t C ha^{-1} yr^{-1} (at a mean biochar carbon content of 73% (Crombie & Mašek, 2015). To make more accurate assessments, alternative uses of residues need to be considered locally and biomass availability (e.g., forestry sites) mapped to biochar use (agricultural sites).

Basalt weathering

Enhanced weathering is the acceleration of the natural process of rock dissolution by crushing Mg- and Ca-rich silicate rocks before application to soil. During weathering, carbon dioxide is captured and initially stored in the form of dissolved bicarbonate (HCO_3^-). Further reactions convert the bicarbonate into Ca and Mg carbonates, which deposit in the marine environment where they remain sequestered for millennia (Beerling et al., 2018). Basalts are the preferred rock types because they are rich in elements beneficial to plant growth (P and K) and contain low concentrations of elements potentially toxic for plants, such as Cr and Ni (Beerling et al., 2018).

Actual basalt weathering rates and hence carbon drawdown potential remain uncertain, depending strongly on particle size (limited by grinding cost), climatic and soil conditions, and biological activity (Amann et al., 2020; Beerling et al., 2020; Kelland et al., 2020). Water flow is critical because mineral surfaces have to be in contact with water for the dissolution reaction to take place, and disturbed for the reaction to continue (Li et al., 2016). Therefore, wet and warm climates demonstrate the highest weathering rates by far (Brady et al., 1999; A. F. White & Blum, 1995).

Besides precipitation and runoff, soil hydrology plays a crucial role in mineral weathering (Maher, 2011). In all climate zones, heavy clays and compacted soils will likely limit the dissolution rates of added basalt severely due to low saturated hydraulic conductivity (poor water flow through soil) and a prevalence of preferential water flow pathways through cracks in soil that minimise interaction with basalt minerals (Maher, 2010; Swoboda-Colberg & Drever, 1993). Under natural conditions, flow in soil generally affects only 0.1-10% of the soil matrix (Velbel, 1993), so that most of the available mineral surfaces cannot exchange solutes, which limits dissolution. Poor contact between pore water and mineral surfaces could explain the 2-3 orders of magnitude difference in weathering rates that is measured in field (poor contact) *vs*. lab (maximum contact) experiments (Swoboda-Colberg & Drever, 1993; Art F. White & Brantley, 2003).

Using sorghum plants and highly controlled experimental conditions with constant irrigation (2,330 mm yr^{-1}), drainage, and assuming permanent exposure of mineral surfaces to water, basalt weathering rates were estimated to drive carbon sequestration at 0.63-0.82 t C ha^{-1} yr^{-1} for 100 t ha^{-1} basalt application, using a reactive transport model (Kelland et al., 2020). This is equivalent to around 10% of its total theoretical carbon sequestration potential (~0.08 t C t^{-1} rock) (Renforth, 2012). Mesocosm studies with wheat and barley, a precipitation of 800 mm yr^{-1}, and natural processes such as drying cycles, preferential water flow, and mineral precipitation, found the carbon sequestration potential of olivine (more rapid theoretical weathering and more total sequestration potential than basalt) to be much lower, 0.006-0.013 t C ha^{-1} yr^{-1} at an application rate of 220 t ha^{-1} (Amann et al., 2020). It confirms the discrepancy of rock weathering rates between controlled lab conditions and natural conditions brought forward by other authors (Swoboda-Colberg & Drever, 1993; Art F. White & Brantley, 2003).

According to the rather limited body of existing studies, enhanced basalt weathering rates might be too low under realistic field conditions (range of 0.01 t C ha^{-1} yr^{-1}) to sequester significant amounts of carbon dioxide on a societally relevant time scale (~100 years). Yet, modelling studies predict significant carbon capture potential in areas where hydrological and climate conditions are suitable (Beerling et al., 2020). This highlights an urgent need for more studies that assess mineral weathering in the field under realistic conditions, and strategies to increase the weathering rate (some of which are discussed in this article).

Mechanistic interactions and synergies among techniques

Nutrient retention, availability, and acquisition

Nutrient leaching and low nutrient use efficiency in agricultural systems (Figure 3a) are significant environmental and economic issues. SOC has a very high cation exchange capacity (CEC), so that building up SOC helps to retain positively charged nutrients, such as Ca, Mg and K (Blume et al., 2016a). Biochar and basalt application mainly affect the CEC in acidic soils through an increase in soil pH, although the direct provision of negatively charged surface sites may also have a positive influence (Anda et al., 2015; Gillman et al., 2002; Jien & Wang, 2013). Enrichment of biomass with inorganic nutrients before pyrolysis or application of biochar with nutrient-rich organic or inorganic materials offers slow-nutrient release potential that provides synergistic improvements on plant growth (Buss et al., 2020; Buss, Jansson, & Mašek, 2019; Hagemann et al., 2017; Mašek et al., 2019; Schmidt et al., 2015).

A global meta-analysis demonstrated that 50% less N fertiliser (typically comprising positively charged ammonium and negatively charged nitrate) was needed for wheat and maize when the SOC content was increased from 0.5 to 1% (Oldfield et al., 2019). Better plant growth feeds more carbon into the soil, helping to build SOC, which then supports further nutrient retention. Biochar ageing could also help to retain nitrate (Hagemann et al., 2017). Intercropping and cover cropping increase N, P, and micronutrient use efficiency, while resource sharing of plants and mycorrhizal fungi facilitates nutrient acquisition, with positive effects on crop growth (Abdalla et al., 2019; Bayala & Prieto, 2019; Brooker et al., 2015; Hartwig & Ammon, 2002).

Plants and microorganisms can mine nutrients from (added) basalt and hence increase nutrient availability and basalt weathering rates by exudation of organic ligands, such as acetate and propionate (Bormann et al., 1998; Kantola et al., 2017). These acids lower the reaction pH, increasing the rate of dissolution, and can also precipitate and form complexes with basalt dissolution products, which enables further dissolution. In addition, uptake of already-dissolved nutrients by plants shifts reaction equilibria towards the products (Bormann et al., 1998; Kantola et al., 2017). In various studies plants increased rock weathering rates by a factor of 1-10 compared to an unplanted control (Bormann et al., 1998; Cochran & Berner, 1996; Haque et al., 2020; Hinsinger et al., 2001; Moulton et al., 2000). It highlights the effect biological activity can have on basalt dissolution and the need to consider the entire plant-soil-climate system to evaluate weathering rates and plant nutrient provision from basalt.

Soil hydraulic functions

Ideally, precipitation is captured in soil through rapid infiltration and high water retention. Soil texture (particle size distribution; sand-silt-clay content), has long been considered the key factor in soil hydrology. In clay-rich soils a low saturated hydraulic conductivity restricts water infiltration and movement within soil. In contrast, saturated hydraulic conductivity is high but water retention is low in sandy soils. Modifying soil texture is challenging because it needs very high application rates of minerals, such as basalt (Amann et al., 2020).

Soil structure (aggregation) is likely to be at least as important as soil texture for soil hydraulic functions (Fatichi et al., 2020; Robinson et al., 2019). Increased SOC content, root biomass, and the abundance of soil organisms have been correlated with high soil aggregation (Miller et al., 2000; Robinson et al., 2019). Ca, often a significant part of basalt, also facilitates soil aggregation and SOC stabilisation, in particular in clay-rich soils (it reduces soil slaking and dispersion) (Baldock et al., 1994; Blume et al., 2016b; Wiesmeier et al., 2019). Therefore, accumulation of SOC and basalt application can help water infiltration and retention. Intercropping facilitates water use efficiency through complementary root architecture, enhanced soil aggregation, and hydraulic lift, i.e., wicking of soil water from deep zones through roots to drier, upper soil. These features can significantly boost plant biomass and yields (Bayala & Prieto, 2019; Bogie et al., 2018; Brooker et al., 2015), which in turn helps build up SOC, highlighting virtuous interactions.

Biochar application can likely change both soil texture via biochar particle size, and soil structure. Application of <30 t ha^{-1} of high surface-area biochar can increase hydraulic conductivity in clay-rich soils (Edeh et al., 2020). While a cumulative biochar application of 10 t ha^{-1} over 5-10 years will only marginally increase the plant-available water content of sandy soil, further application to >30 t ha^{-1} is expected to substantially increase the water-storage capacity (Edeh et al., 2020).

More available water can increase plant growth, which in turn helps to retain and re-circulate water locally (transpiration instead of runoff) (Green et al., 2013), and to improve the contact between water and minerals and, hence, the mineral weathering rate.

Aboveground plant carbon sequestration efficiency

Producing biochar from aboveground plant residues in high-biomass systems is key because it has a higher CSE than natural biomass decomposition on a century timescale. Optimising the biochar production system for maximum (stable) carbon yield decreases carbon losses further, and significantly improves the CSE. The carbon sequestration potential of woody biochar per unit biomass input can be increased by up to 45% by spraying low levels (2%) of alkali (and earth alkaline) metals onto the biomass, such as potassium or sodium (Mašek et al., 2019), or by incorporating wood ash (Buss, Jansson, Wurzer, et al., 2019) (Figure 2c). A significant part of basalt comprises alkali and earth alkaline metals (Table 2) that could also have the potential to catalyse biochar formation when incorporated into the biomass before pyrolysis, which in addition increases the nutrient content of biochar, providing further benefits for plant growth and carbon sequestration.

Biochar could be produced from biomass harvested from matured agroforestry systems, (Figure 3d), which are estimated to ultimately provide up to 10x higher biomass yields than simple cropping and pasture systems (Chapman et al., 2020) with estimates ranging from 0.3-15 t C ha^{-1} (Nair et al., 2010). Light limitation can cause tree growth to decline with age, so that tree pruning and thinning stimulate higher growth rates (Binkley et al., 2002; Forrester et al., 2013). The conversion of harvested agroforestry residues (average ~15% tree biomass pruning/thinning assumed per year) into biochar could make 0.05-2.3 t ha^{-1} yr^{-1} biomass available that supports the production of 0.01-0.6 t ha^{-1} of biochar per year (~0.01-0.4 t C ha^{-1} yr^{-1}), which is in addition to crop/forestry residue biochar. Other options to obtain biomass for biochar production on-farm are setting aside land for tree plantations (Werner et al., 2018) or fast growing bioenergy crops, harvesting woody weeds, which can yield up to 44 t ha^{-1} of biomass (Simmons et al., 2021), or increasing straw residues by planting crop varieties with lower harvest indices.

Dividing aboveground tree and shrub residues from agroforestry systems into N-rich green material and carbon-rich woody debris could further increase the CSE and improves N management (Figure 3b, d). During pyrolysis, N is mostly lost or converted into an unavailable form (Hossain et al., 2011), so that only the N-poor biomass fraction should be used for biochar production. Green, N-rich biomass is (biologically) converted into SOC more efficiently (higher CSE) than N-poor biomass (Anthony et al., 2020; Castellano et al., 2015; Cotrufo et al., 2013), which makes N-rich (composted) biomass ideal for building up SOC and providing N to plants (Figure 3b).

Importantly, more N is needed in the formation of mineral-associated SOC, relative to less persistent forms of SOC (aggregate carbon); more available N in soil increases SOC stability (Cotrufo et al., 2019). Consequently, the SOC pool is typically higher and more stable under N-rich plant species, e.g. legumes and N-fixing trees, than under N-poor species, which highlights the value of N-rich biomass as cover or inter crop (Angst et al., 2019; Anthony et al., 2020; Chenu et al., 2019) (Figure 3b, d).

Belowground plant carbon sequestration efficiency

Root biomass and rhizodeposition inject carbon deeper into soils than the soil surface plant-litter pathway, and offer a more efficient route for conversion into (stable) SOC (higher CSE) (~46% below ground in agricultural systems *versus* ~8% above ground; Figure 2a highlights these pathways including carbon partitioning within the plant) (Jackson et al., 2017; Sokol & Bradford, 2019). Further increase in the CSE could be achieved through management of the soil microbial community to increase the proportion of rhizodeposits that is converted into stable SOC (Kallenbach et al., 2019; Mukasa Mugerwa & McGee, 2017) (Figure 2b). 'Deep carbon' stocks (>20 cm) are also less influenced by climate than near-surface SOC, and so are less likely to be released in response to climate change (Mathieu et al., 2015).

Crops (annual – one season – plants), however, only supply belowground carbon within the first ~100 days, with a sharp decline after ~30 days (Pausch & Kuzyakov, 2018). Perennials supply belowground carbon (roots + rhizodeposits) over the entire vegetation period at levels equivalent to peak carbon supplies from annual crops. Therefore, a constant plant cover in form of perennial cover crops (living mulch), preferably a mix of legumes and non-legumes, or intercrops (e.g. agroforestry) can provide a continuous source of deep carbon, fostering both improved (stable) SOC formation and increased plant yields (Abdalla et al., 2019; Cardinael et al., 2018; Hartwig & Ammon, 2002). Elevated rhizodeposit input, e.g. through cover crops, however, could also result in loss (priming) of existing SOC stocks under some circumstances (Keiluweit et al., 2015; Tautges et al., 2019). Further investigation into locally optimized practices is needed to achieve the best net outcomes.

Genetic selection of annual crops for increased belowground carbon allocation may also increase the stable SOC pool (Jackson et al., 2017; Pausch & Kuzyakov, 2018; Sokol & Bradford, 2019). Although this can in the short term decrease crop yields owing to diversion of plant energy from grain to belowground mass, SOC levels up to 2% correlate positively with crop yields, which demonstrates that building up SOC eventually results in a net agronomic advantage (Oldfield et al., 2019). In addition, a higher carbon allocation in rhizodeposits can result in enhanced nutrient supply from microorganisms, since rhizodeposits directly feed microorganisms in exchange for nutrients (Sulman et al., 2017). Nurturing healthy soils by investing energy and resources belowground will bring benefits that allow farming systems to maintain yields in a changing climate, in stark contrast to a system purely focused on short-term optimization of carbon allocation into grains (Figure 1).

Longer-term SOC storage

Mineral-associated SOC storage depends on availability of appropriate sink minerals. Saturation takes place when the store of suitable minerals has been utilised, and leads to particularly low CSEs in some soils (Chenu et al., 2019; Lal, 2018). Basalt weathering supplies abundant Ca, Mg, Al, and Fe (Table 2) to the soil surface layer providing mineral surfaces for the formation of mineral-associated SOC, and improving soil aggregation (aggregate carbon) (Castellano et al., 2015; Chenu et al., 2019; Cotrufo et al., 2013; Wiesmeier et al., 2019) (Figure 2c). Application of goethite (an Fe-rich mineral) at 1.6 t ha^{-1} has been found to increase the CSE of rhizodeposits (Jeewani et al., 2020). Biochar application does not increase the mineral surface sink, but it can increase the conversion efficiency of rhizodeposits into mineral-associated SOC (higher CSE), decrease SOC degradation (negative priming), and foster the formation of microaggregates that promote further SOC stabilisation (Weng et al., 2017) (Figure 2c).

Biochar contains chemically and biologically recalcitrant carbon (Lehmann & Joseph, 2015) that does not easily degrade into low-molecular weight hydrocarbons, the form in which SOC sorbs to and is protected by minerals (Sokol et al., 2019). Therefore, removing carbon from the natural plant-SOC-atmospheric CO_2 cycle via pyrolysis helps to avoid SOC saturation of mineral surfaces. In particular in regions with soils close to their maximum SOC storage capacity (which reduces the CSE of plant litter into SOC (Chenu et al., 2019), crop, shrub, and tree residues should be pyrolyzed to avoid release of existing SOC (positive priming) (Anthony et al., 2020; Weng et al., 2020). The capacity to store carbon in the form of biochar in soils is likely unlimited (Table 1, Figure 2).

Strategies for integration in agro-ecosystems

On a global scale, plant growth is limited by P and N, although K can also limit productivity (Sardans & Peñuelas, 2015; Terrer et al., 2019). N for crop growth can be provided by microorganisms that live in natural symbiosis with plants, but P, K, and other nutrients are non-renewable and depleted in many ecosystems (Sardans & Peñuelas, 2015; Terrer et al., 2019). Basalts contain mineral nutrients in relevant quantities to satisfy plant demand, and therefore can (partly) replace conventional fertiliser application; on average basalts from four continents contained 0.2% P, 0.7% K, 5.3% Ca, and 3.7% Mg (Table 2) (Dumitru et al., 1999; Marsh, 1987; Mcdougall, 1976; Reichow et al., 2005).

Basalt application at 10 t ha^{-1} provides 8-53 kg P ha^{-1} and 19-426 kg K ha^{-1} (Table 2). Typical recommendations (depending on soil type, existing soil nutrients, etc.) are 40 kg P ha^{-1} and 133 kg K ha^{-1} for winter wheat, and 26 kg P ha^{-1} and 50 kg K ha^{-1} for improved rice varieties (FAO, 2006). This demonstrates that basalts can theoretically supply sufficient K and P to compensate for nutrients that are removed with the harvest. However, not all of the K and P is immediately plant available (Gillman et al., 2002; Kelland et al., 2020), and further research is needed to establish basalt-based nutrient supply in the short (immediate plant uptake), medium (one growing season), and long term (several growing seasons).

Basalts (and biochar) also contain Ca and Mg that can neutralise acidic soil (Biederman & Harpole, 2013; Gillman et al., 2002). Biochar and rock dust application at rates of 1 t ha^{-1} and 10 t ha^{-1}, respectively, supplies calcium carbonate equivalent to 0.9-3.7 t ha^{-1} of lime; woody biochar provides ~0.06 t ha^{-1} (Singh et al., 2017) and basalt 0.8-3.6 t ha^{-1} (Table 2). At such proposed application rates, the pH in soils of most textures and CECs will likely increase to 5.5-6.5, the ideal pH for most plants (Roy et al., 2006; Upjohn et al., 2005). Yet, the response of soil pH to biochar and basalt application is slower than that to conventional lime addition because of a lower solubility (Beerling et al., 2020; Jeffery et al., 2017). Still, silicate rocks can be a sustainable lime replacement that avoid the CO_2 emissions associated with lime production and application (Dietzen et al., 2018).

In semi-arid and arid areas rehydration strategies that supply water to plants will result in additional plant carbon and SOC accumulation (Trost et al., 2013) and likely basalt weathering. Given that severe droughts accelerated by climate change already affect many areas around the world, and are predicted to intensify and spread geographically (Grillakis, 2019), the development of efficient rehydration strategies will be key to climate change adaption and ecosystem resilience. Such strategies cannot be overly reliant on ponds/lakes/dams, given that shallow open waters with large surface areas are subject to disproportionate evaporative losses and can be a source of methane (Aguilera et al., 2019; Deemer et al., 2016). Instead, interventions to improve water retention within soils are critical.

Biogeochemical interventions through strategic application of biochar and basalt have the potential to spark virtuous cycles that increase water use efficiency, plant growth, and SOC accumulation (Figure 3b). In addition, cover cropping and landscape design through strategic tree planting, establishment of contour lines and soil terraces increase water infiltration and slow down the flow of water through the plant-soil system, and so help rehydrate the landscape (Bayala & Prieto, 2019; Blanco-Canqui et al., 2015; Dobes et al., 2013; George et al., 2012). To enable significant basalt weathering even under low water conditions, we propose banded basalt application and landscape contouring to align water flow with the buried basalt. This should be tested in future studies. Yet, cover crops increase water transpiration losses and can result in decreased yield in semi-arid environments, which calls for region-specific adaptation of practices (Bodner et al., 2007; Hartwig & Ammon, 2002).

In our proposed method increasing plant carbon capture and growth in agricultural systems with (perennial) ground cover and partial tree canopy cover is the first step (Figure 1). Improvements to water and nutrient supply enhance long-term soil properties and plant growth. The extra plant biomass then is managed through efficient conversion into biochar and stable SOC (Figure 2c). This enables virtuous cycles that further capture and storage water and carbon (Figure 1).

Outlook

Various unanswered issues arise as key future research questions, such as the weathering rate and plant nutrient-provision potential of basalt and the degree to which a specifically designed and regeneratively managed landscape can increase water use efficiency. Field trials and demonstration sites across climate and soil types are urgently needed to establish guidelines toward optimised carbon sequestration in productive agro-ecosystems.

Even more importantly, gaps between disciplines need to be bridged. First, to facilitate adoption of these concepts in practice, novel soil models with measurable soil carbon pools (Abramoff et al., 2018; Robertson et al., 2019) and improved representation of soil structure and associated hydrologic responses (Fatichi et al., 2020) need to be integrated into crop growth models, and calibrated to local conditions. Prediction tools will increase confidence in long-term sequestration benefits, which is required to garner further support from industry, government, and farmers. Second, application of biochar and basalt in different proportions and compositions needs to be incorporated into the models and combined with techno-economic analyses and decision-support tools. Detailed landscape mapping and analysis will allow further fine-scale modelling of nutrient and water flows and help in determining the ideal placement of trees and establishment of rehydration strategies in water-limited environments. Such fine-scale modelling and adaptations in heterogenous landscapes are essential for tailored approaches with respect to local to regional scale soil and climate conditions, which form the corner stone of successful implementations that safeguard our climate, environment, and food production.

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Author contributions

W.B. prepared the manuscript and designed the figures with input from the co-authors. K.Y., J.B. and E.J.R. edited and reviewed the manuscript. All authors worked on conceptualising the idea of integration of carbon sequestration techniques that was originally developed by J.B. and E.J.R.

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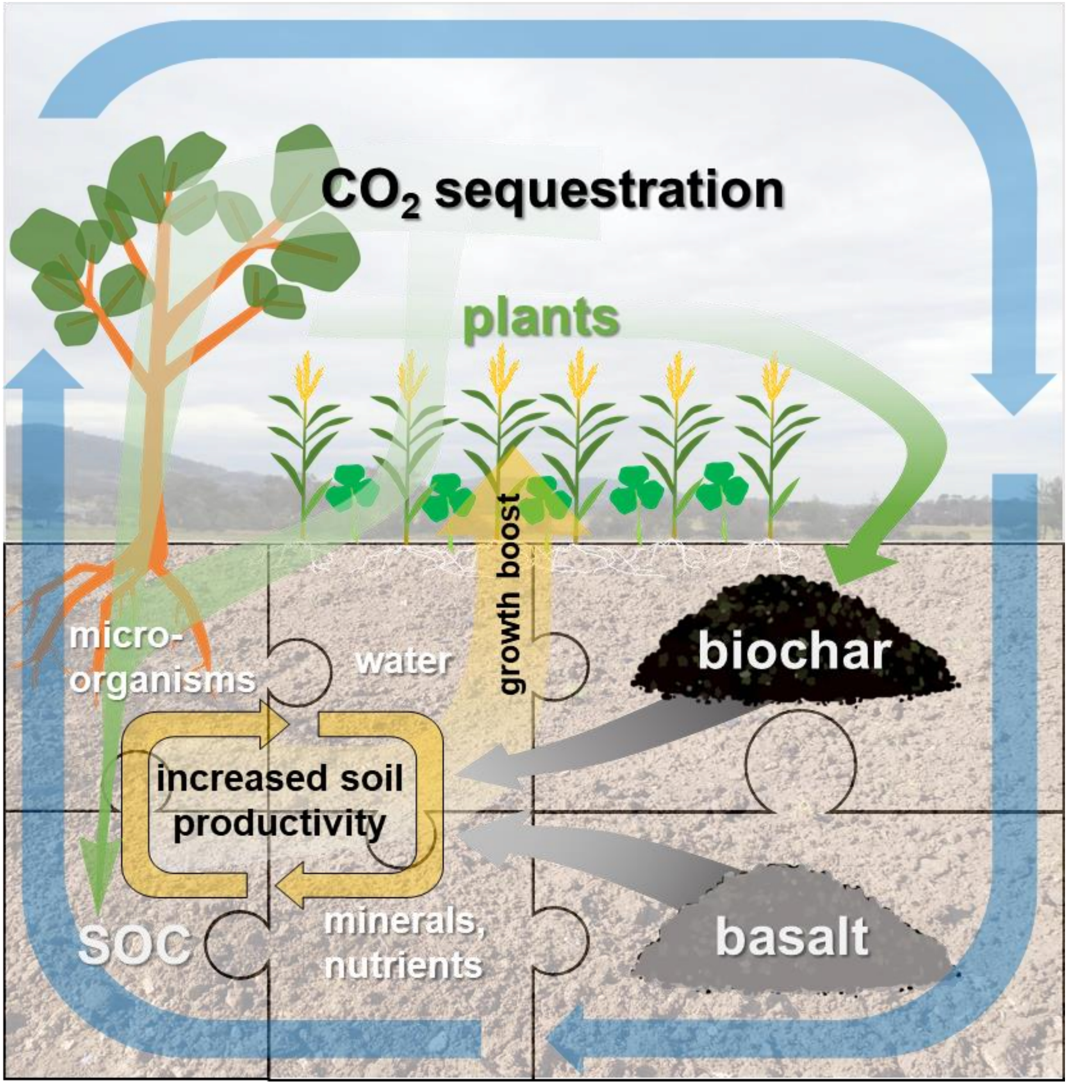

Figure 1: Integration of four land-based carbon sequestration techniques on the same land area. Improved soil conditions (microorganisms, water, minerals/nutrients and SOC content) boost plant growth. Addition of basalt and biochar can enhance a virtuous cycle of plant carbon capture and soil storage. Green symbolises plant carbon flow, blue is the hydrological cycle.

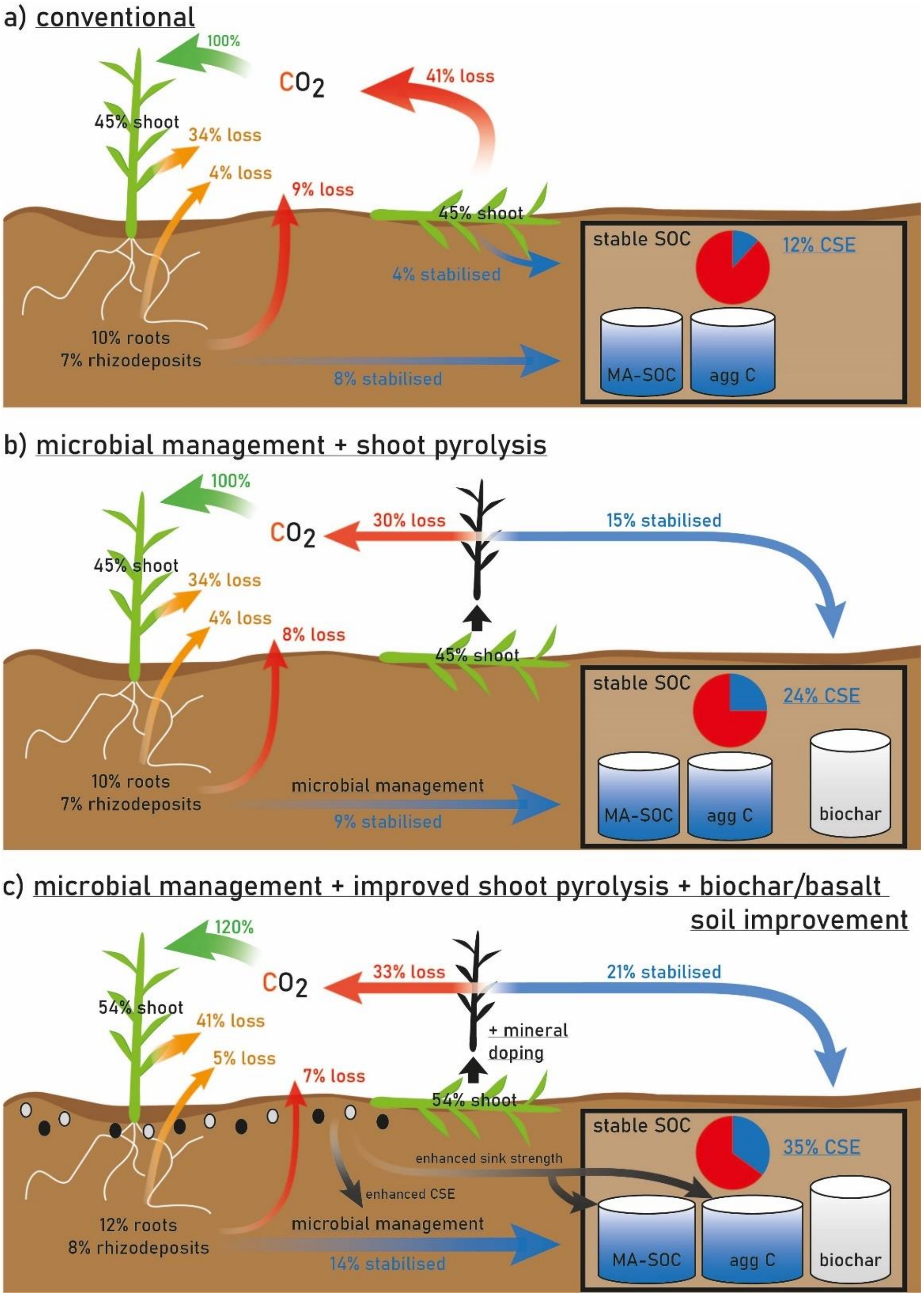

Figure 2: Relative carbon sequestration efficiency (CSE) of above- and belowground plant carbon into stable forms of soil carbon (MA-SOC, agg C, biochar). (a) Conventional cropping systems, (b) system with plant shoot pyrolysis (+ biochar soil application) and management of microbial composition for maximum stable SOC accumulation, and (c) system with mineral doping of feedstock before pyrolysis, microbial management, and improvements of soil properties through biochar and rock dust application, which increases plant growth and photosynthesis. Size and shading of the stable carbon

cylinders demonstrate the size of the carbon sink and level of saturation, with biochar having unlimited sink strength. Green arrows represent photosynthesis, orange autotrophic respiration, red heterotrophic respiration, and blue carbon stabilisation pathways. MA-SOC, mineral associated SOC; agg C, aggregate carbon. Percentage are example literature values presented for illustrative purposes, they vary according to the system under investigation (soil, plant type etc): plant carbon allocation (Pausch & Kuzyakov, 2018), conversion efficiency of plant litter and rhizodeposits into SOC (Jackson et al., 2017), concept of increased CSE of rhizodeposits into SOC (20% relative improvement assumed) (Mukasa Mugerwa & McGee, 2017), biochar CSE and improved CSE through mineral doping (mean across pyrolysis temperatures) (Mašek et al., 2019), increase in SOC storage capacity and CSE by biochar and basalt (combined relative CSE improvement of 20% assumed) (Jeewani et al., 2020; Weng et al., 2017).

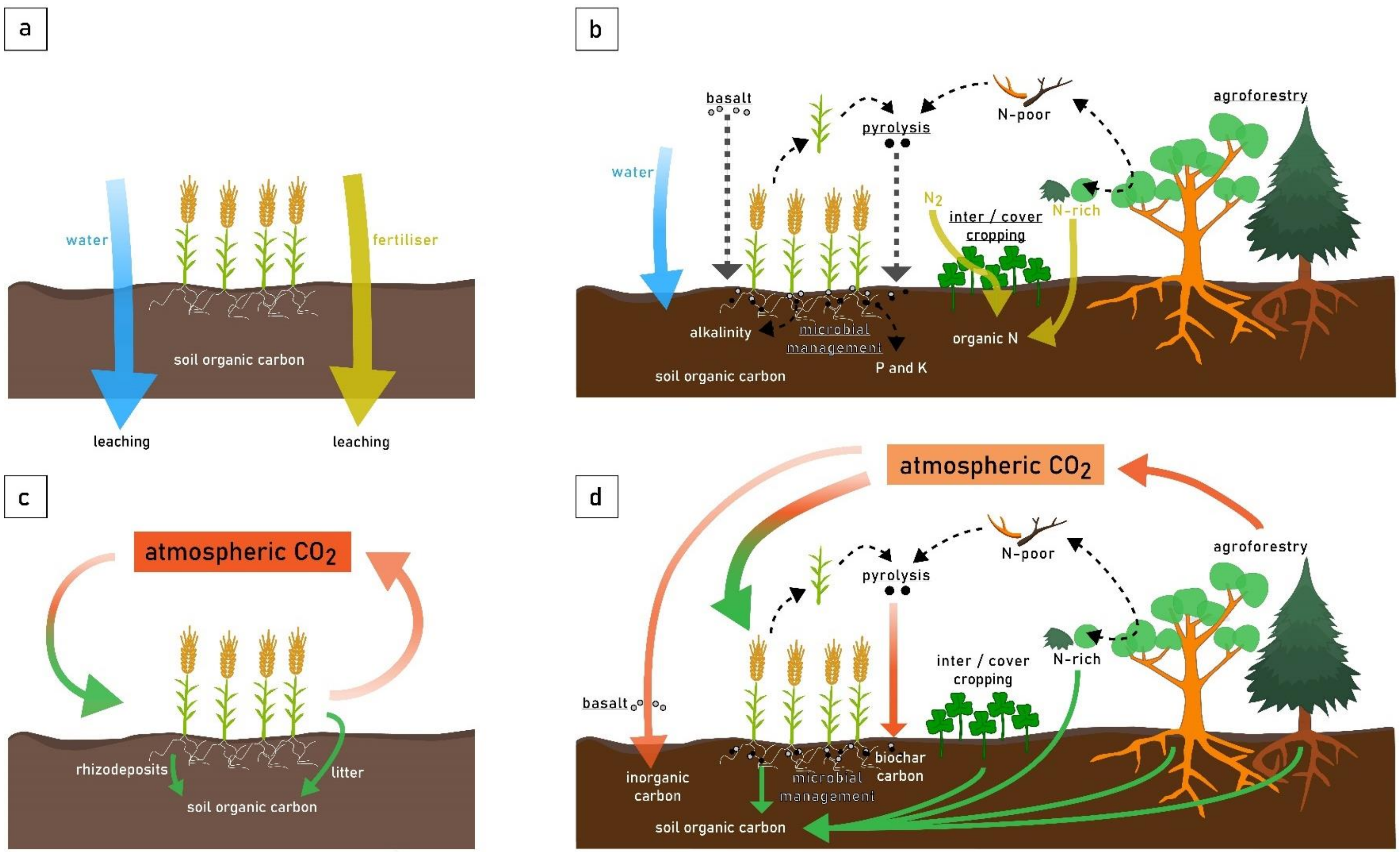

Figure 3: Nutrient and water (a, b) and carbon dynamics (c, d) in agro-ecosystems. Panels (a, c) represent a conventional cropping system and (b, d) enhanced system with land management options to maximise carbon sequestration and system resilience. In (a) soil holds fertiliser and water poorly. In (b) accumulation of SOC and improved soil aggregation enables improved water retention and inter/cover cropping and basalt provide nutrients. (c) describes a conventional system where carbon is provided to soil in the form of crop rhizodeposits and residues (litter) with most of the plant carbon lost as CO_2. In (d) CO_2 is captured by a diverse arrangement of plants and through basalt weathering.

Table 1: Key attributes of four land-based carbon sequestration techniques.

	plants	SOC	biochar	enhanced basalt weathering
carbon capture	✓	✗	✗	✓
“permanent” carbon sequestration (>100 years)	✓/ ✗ prone to perturbations	✓/ ✗ prone to perturbations	✓	✓
“unlimited” sink capacity	✗	✗	✓	✓
main carbon sequestration limitations	land area, nutrients, water	biomass input, microbial carbon conversion efficiency, microbial SOC decomposition	biomass, production costs	weathering rate, grinding and transport cost
improvement of soil properties	✓	✓	✓	✓

Table 2: Elemental contents of basalts (Dumitru et al., 1999; Marsh, 1987; Mcdougall, 1976; Reichow et al., 2005). CCE, calcium carbonate equivalency (%) compares lime with lime replacements in their ability to alter soil pH, based on 40% Ca content in $CaCO_3$.

		concentration (%)					dose in t ha^{-1} at basalt application rate of 10 t ha^{-1}				
	n	mean	SD	median	min	max	mean	SD	median	min	max
Mg as MgO	64	3.68	1.12	3.51	1.77	7.0	368	112	351	177	697
Ca as CaO	64	5.27	1.15	5.37	1.52	7.6	527	115	537	152	759
K as K_2O	64	0.71	0.58	0.64	0.19	4.3	71	58	64	19	426
P as P_2O_5	64	0.21	0.10	0.19	0.08	0.5	21	10	19	8	53
Al as Al_2O_3	64	9.35	1.12	9.41	5.40	11.5	935	112	941	540	1,150
Fe as Fe_2O_3	64	5.57	2.49	6.56	1.00	9.3	557	249	656	100	933
CCE (%)		22			8	36	2,236			822	3,639