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Soil carbon sequestration and biochar as negative emission technologies

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Abstract

Despite 20 years of effort to curb emissions, greenhouse gas (GHG) emissions grew faster during the 2000s than in the 1990s, which presents a major challenge for meeting the international goal of limiting warming to <2 °C relative to the preindustrial era. Most recent scenarios from integrated assessment models require large-scale deployment of negative emissions technologies (NETs) to reach the 2 °C target. A recent analysis of NETs, including direct air capture, enhanced weathering, bioenergy with carbon capture and storage and afforestation/deforestation, showed that all NETs have significant limits to implementation, including economic cost, energy requirements, land use, and water use. In this paper, I assess the potential for negative emissions from soil carbon sequestration and biochar addition to land, and also the potential global impacts on land use, water, nutrients, albedo, energy and cost. Results indicate that soil carbon sequestration and biochar have useful negative emission potential (each 0.7 GtCeq. yr^{-1}) and that they potentially have lower impact on land, water use, nutrients, albedo, energy requirement and cost, so have fewer disadvantages than many NETs. Limitations of soil carbon sequestration as a NET centre around issues of sink saturation and reversibility. Biochar could be implemented in combination with bioenergy with carbon capture and storage. Current integrated assessment models do not represent soil carbon sequestration or biochar. Given the negative emission potential of SCS and biochar and their potential advantages compared to other NETs, efforts should be made to include these options within IAMs, so that their potential can be explored further in comparison with other NETs for climate stabilization.

Keywords: biochar, carbon, negative emission technology, sequestration, soil

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Introduction

Greenhouse gas (GHG) emissions grew faster during the 2000s than in the 1990s (Le Quéré *et al.*, 2013), despite 20 years of effort to curb emissions. This continuing increase in emissions will present a major challenge for meeting the international goal of limiting warming to <2 °C relative to the preindustrial era, particularly if stringent climate policies are not introduced rapidly (Peters *et al.*, 2013; Edenhofer *et al.*, 2014; Tavoni *et al.*, 2015). To avoid warming of more than 2 °C with a >50% chance, most recent scenarios from integrated assessment models (IAMs) include the large-scale deployment of negative emissions technologies (NETs), that is technologies that result in the net removal of CO_2/GHGs from the atmosphere (Edmonds *et al.*, 2013; Rogelj *et al.*, 2013; van Vuuren *et al.*, 2013; Clarke *et al.*, 2014; Edenhofer *et al.*, 2014; Fuss *et al.*, 2014; Krey *et al.*, 2014; Riahi *et al.*, 2015). As society must decide which mitigation pathways are desirable to tackle climate change, information on the potential risks and opportunities afforded by all NETs is necessary.

Smith *et al.* (2015) recently reviewed and analysed the biophysical and economic limits to NET implementation for a number of NETs: (i) bioenergy (BE; Creutzig *et al.*, 2015) with carbon capture and storage (CCS; together referred to as BECCS; Obersteiner *et al.*, 2001), (ii) direct air capture of CO_2 from ambient air by engineered chemical reactions (DAC; Keith, 2009; Socolow *et al.*, 2011), (iii) enhanced weathering of minerals (EW; Schuiling & Krijgsman, 2006; Köhler *et al.*, 2010; Hartmann & Kempe, 2008; Kelemen & Matter, 2008) where natural weathering to remove CO_2 from the atmosphere is accelerated, and the products stored in soils, or buried in land/deep ocean and (iv) afforestation and reforestation (AR) to fix atmospheric carbon in biomass and soils (Canadell & Raupach, 2008; Jackson *et al.*, 2008; Arora & Montenegro, 2011). For reasons of tractability, the analysis of Smith *et al.* (2015) did not consider (v) manipulation of uptake of carbon by the ocean either biologically (i.e. by fertilizing nutrient limited areas; Sarmiento *et al.*, 2004; Joos *et al.*, 1991) or

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chemically (i.e. by enhancing alkalinity; Kheshgi, 1995), or other land-based options, such as (vi) changed agricultural practices (which include activities such as less invasive tillage with residue management, organic amendment, improved rotations/deeper rooting cultivars, optimized stocking density, fire management, optimized nutrient management and restoration of degraded lands; Smith *et al.*, 2008; Smith, 2012; Powlson *et al.*, 2014), or (vii) converting biomass to recalcitrant biochar, for use as a soil amendment (Woolf *et al.*, 2010). IAMs have so far focused primarily on BECCS (Azar *et al.*, 2010; Kriegler *et al.*, 2013; van Vuuren *et al.*, 2013) and AR (Strengers *et al.*, 2008; Wise *et al.*, 2009; Reilly *et al.*, 2012; Humpenöder *et al.*, 2014). Although soil carbon is starting to be considered by some IAMs, it has so far only been considered as a by-product of other land management actions, such as BECCS and AR. In this study, we assess the NET potential for, and biophysical and economic limitations of, the two land-based NETs not considered in Smith *et al.* (2015), that is soil carbon sequestration (SCS) and soil amendment with biochar. This is timely given the proposal of the French Government to attempt to increase global soil carbon stocks by 0.4% as a climate mitigation measure. Table 1 provides a summary of the abbreviations used.

Figure 1 depicts the main flows of carbon among atmospheric, land, ocean and geological reservoirs for fossil fuel combustion (Fig. 1a), BE (Fig. 1b) and CCS (Fig. 1c), and the altered carbon flows for BECCS (Fig. 1d), for soil carbon sequestration (Fig. 1e), for biochar addition to soil (Fig. 1f) and for biochar addition to soil as part of BECCS (Fig. 1g).

Using data from the latest available literature, I first assess the impact on land use (area required) and GHG emissions (total GHG balance), water (water use per unit of negative emissions), nutrients (nitrogen: N, phosphorus: P and potassium: K associated with sequestered carbon), biophysical climate impacts (represented by surface albedo), energy produced or demanded (per unit of negative emissions) and cost (cost per unit of negative emissions), of SCS and biochar, all per unit of negative emission (i.e. per-t-Ceq.). I then use global estimates of potential implementation of these two NETs (GtC yr^{-1}), to estimate global impacts on each of these biophysical and cost parameters, using similar methods to those used recently by Smith *et al.* (2015).

Table 1 Abbreviations for negative emissions technologies used in this paper

Abbreviation	Full name of negative emission technology
NET	Negative Emission Technology
DAC	Direct Air Capture
EW	Enhanced Weathering
AR	Afforestation/Reforestation
BE	Bioenergy
CCS	Carbon Capture and Storage
BECCS	Bioenergy with Carbon Capture and Storage
SCS	Soil Carbon Storage
Biochar	Biochar addition to soil

Materials and methods

Sources of data used to estimate impacts of SCS and biochar on a per-t-Ceq. removal basis are summarized in Table 2. Papers were selected by Web of Knowledge search for the impacts considered in this paper.

Per-t-Ceq. impact values were then scaled to the global level by multiplying per-t-Ceq. impact values by global implementation levels from Smith *et al.* (2008) for SCS and from Woolf *et al.* (2010) for biochar, similar to the approach used by Smith *et al.* (2015) for other NETs.

Results

Estimating impacts of SCS and biochar on a per-t-Ceq. removal basis

Land use and GHG impacts. SCS rates vary considerably by soil type and climate region (Smith, 2012), but a global meta-analysis derived mean values for croplands and grasslands ranging from 0.03 (the lowest mean cropland SCS rate) to around 1 tC ha^{-1} yr^{-1} (for restoration of degraded land; Smith *et al.*, 2008). Inverting these values gives a land requirement per-t-Ceq. of negative emissions, that is a range of 1–33 ha tC^{-1}. Given that biochar is more recalcitrant than soil organic matter (Lehmann *et al.*, 2015), and that application rates to soil can be high (e.g. 30–60 t ha^{-1}: Genesio *et al.*, 2012; Zhang *et al.*, 2010), negative emissions per ha can be much higher, giving land requirements for biochar of <1 ha tC^{-1}. However, as SCS and biochar addition can be applied to all managed land without changing its current use, there is no competition for land associated with these NETs, unlike AR and BECCS, which require land use change and compete for land for other uses (Smith *et al.*, 2010). Net removal of CO_2 from the atmosphere could be further increased by increasing plant productivity. Lal (2004) reports that an addition of 1 tC ha^{-1} on degraded cropland soils could increase crop yield by 0.5 kg ha^{-1} (cowpeas) to 40 kg ha^{-1} (wheat). If this additional yield is removed and consumed as food or feed, however, it would not contribute to net CO_2 removal from the atmosphere.

In addition to the impacts on soil carbon storage, SCS and biochar can impact other GHGs. While increased soil organic matter under SCS is considered to have

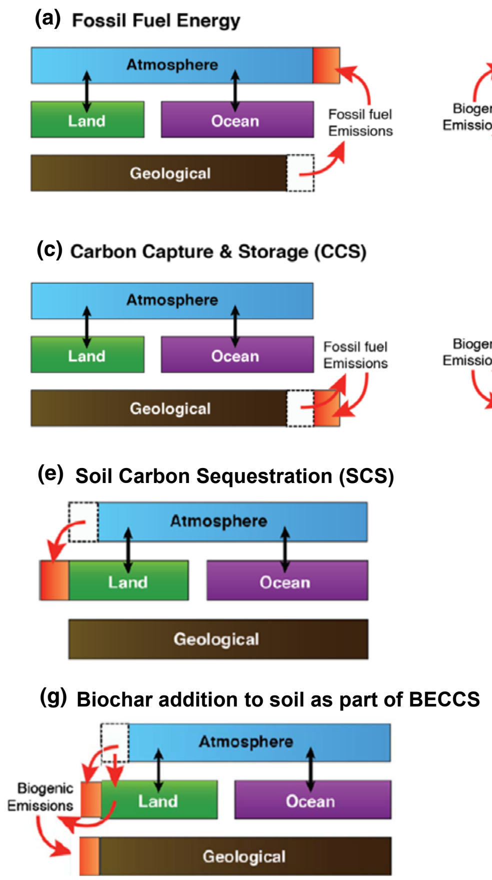

Fig. 1 The main flows of carbon among atmospheric, land, ocean and geological reservoirs for fossil fuel combustion (a), BE (b) and CCS (c), and the altered carbon flows for BECCS (d), for SCS (e), for biochar addition to soil (f) and for biochar addition to soil as part of BECCS (g). Adapted from Smith *et al.* (2015).

small/negligible impacts on soil methane (CH_4) emissions (Smith *et al.*, 2008), increased soil C storage results in more organic nitrogen in the soil, which could be mineralized to become a substrate for nitrous oxide (N_2O) production, although the effect is difficult to quantify (IPCC, 2006). Biochar addition, on the other hand, has quantifiable impacts on non-CO_2 GHG emissions, with CH_4 emissions increasing significantly and N_2O emissions decreasing substantially when biochar is added to paddy rice soils (Zhang *et al.*, 2010). Other studies in pasture systems show the opposite effect (increased N_2O; decreased CH_4; Scheer *et al.*, 2011) and others no effect (Castaldi *et al.*, 2011). The prevention of CH_4 and N_2O emissions from biomass decay when biomass is used to produce biochar is also expected to contribute to the emission reduction delivered by biochar (Woolf *et al.*, 2010). Given the uncertainty in the relationships between both soil organic matter content and biochar addition and CH_4 and N_2O emissions, the impact on non-CO_2 GHGs for both SCS and biochar is assumed here to be zero.

Water use. Water is not used in quantity in the production of biochar (with the exception of hydrothermal carbonization; Meyer *et al.*, 2011). In terms of impacts on soil water when applied, per-t-Ceq. water impacts of increased soil organic matter storage and biochar additions to soil are difficult to quantify, although it is widely accepted that both improve soil water holding capacity (Lal, 2004; Lehmann & Joseph, 2009; Woolf *et al.*, 2010), thereby increasing retention of water in the soil/plant system. Unlike other NETs, then, SCS and biochar are likely to have a net beneficial impact on water footprint (and no negative impact), so the per-t-Ceq. impact on water use is <0. Additional water used for irrigation to grow the biochar feedstock is not included; if the biomass feedstocks needed irrigation, this would have a water footprint.

Nutrient impacts. Soil organic matter from SCS provides additional nutrients for plant uptake, as the stoichiometry of the organic matter means that for every t C ha^{-1} of soil organic matter added, nutrients, that is N, P and

Table 2 Sources of data used to estimate impacts of SCS and biochar on a per-t-Ceq. removal basis

Impact	Data type	Source
Land use and GHG impact	Land use intensity for SCS	Smith *et al.* (2008)
	Increased productivity from SCS	Lal (2004)
	N_2O emissions from SCS	IPCC (2006)
	CH_4 and N_2O emissions from biochar	Zhang *et al.* (2010); Scheer *et al.* (2011); Castaldi *et al.* (2011); Woolf *et al.* (2010); Meyer *et al.* (2011)
Water use	Qualitative data on improved soil water holding capacity for SCS and biochar	Lal (2004); Lehmann & Joseph (2009); Woolf *et al.* (2010)
Nutrient impacts	Nutrient content: nitrogen (N), phosphorus (P) and potassium (K) for SCS	Lal (2004)
	Nutrient content: nitrogen (N), phosphorus (P) and potassium (K) for biochar	Roberts *et al.* (2015)
Albedo impacts	Qualitative impact of SCS management practices on albedo	Luyssaert *et al.* (2014); Daughtry *et al.* (2006); Pacheco & McNairn (2010)
	Quantitative impact of biochar application on albedo	Meyer *et al.* (2012); Genesio *et al.* (2012)
Energy	Range of energy requirements for SCS management	Smith *et al.* (1998, 2008)
	Range of energy generation potentials for biochar	Roberts *et al.* (2015); Shackley *et al.* (2011); Meyer *et al.* (2011); Woolf *et al.* (2014)
Cost	Costs of SCS management actions	Smith *et al.* (2008); McKinsey & Co. (2009)
	Costs of biochar production/addition	Shackley *et al.* (2011); Meyer *et al.* (2011); Dickinson *et al.* (2014)

K, would increase by 80 kgN ha^{-1}, 20 kgP ha^{-1} and 15 kgK ha^{-1} (calculated from values in Lal, 2004). Although these nutrients need to be added to maintain stable soil organic matter, it is assumed that these can be obtained through a combination of organic matter addition and nitrogen fixation. Biochar is credited with reducing nutrient losses from soils (Lehmann *et al.*, 2003; Steiner *et al.*, 2008). Although biochars are very variable, depending on feedstock and pyrolysis temperature (Lehmann & Joseph, 2009), using the average of ranges of C, N, P and K content given for lignocellulosic biochars in Roberts *et al.* (2015), biochar containing 1 tC ha^{-1} would contain ~30 kgN ha^{-1}, ~10 kgP ha^{-1} and ~70 kgK ha^{-1}. Nutrient benefits are therefore quantifiable for SCS and biochar, and in both cases are positive. Additional fertilizers to grow the biochar feedstock are not included; if the biomass feedstocks required more fertilizer than the previous land use, this would have a negative impact on nutrients.

Albedo impacts. Although soil organic carbon content is unlikely to impact directly upon albedo to a significant extent, some practices that are used for SCS (e.g. residue management) can impact upon albedo (Luyssaert *et al.*, 2014), to the extent that remote sensing can be used to detect residues on fields (Daughtry *et al.*, 2006; Pacheco & McNairn, 2010). It is conceivable that light coloured residues (e.g. cereal straw) left upon the surface of an otherwise bare soil could increase albedo, thereby providing an additional climate benefit to soil C storage. The magnitude of this impact, however, cannot currently be quantified due to uncertainty over the mix of management practices used for SCS and the albedo impacts of each practice. It is safe to assume, however, that there will be no negative impacts on albedo from SCS, as there could be from AR (Betts *et al.*, 2007).

Because biochar material tends to be dark and can be applied in large quantities, it can darken the soil surface. Biochar application at 30–60 t ha^{-1} to soil decreased surface albedo over the crop season by up to 40% relative to controls, which, in turn, increased soil temperature (Genesio *et al.*, 2012). Control albedo measurements in this experiment were around 0.2 (early growing season) to 0.3 (late growing season), so a 40% reduction represents absolute albedo decreases of ~0.08–0.12 (Genesio *et al.*, 2012). These albedo changes are consistent with the mean annual albedo reduction of 0.05 calculated for application of 30–32 t ha^{-1} biochar to a test field in Germany by Meyer *et al.* (2012).

Energy. Most practices that promote SCS are close to current practice (e.g. improved rotations, residue management) and would not incur a significant energy cost. Indeed, some practices (such as reduced/zero tillage) may save energy by reducing the energy input to farm operations, such as diesel use in tillage (Frye, 1984; Kern & Johnson, 1993; Smith *et al.*, 1998). Some practices, however, may incur an energy cost (e.g. additional energy for pumping irrigation water), but given that SCS can be achieved in most cases with energy inputs that do not differ significantly from baseline practices, SCS is assumed to be energy neutral.

Assuming energy contents for lignocellulosic biochars of 16.4–35.3 MJ kg^{-1} from Roberts *et al.* (2015), and allowing for 10% and 20% process energy cost and energy loss in pyrolysis plants, respectively (from Shackley *et al.*, 2011), then expressing the range as per-t-C, we arrive at an energy generation potential of ~20–50 GJ tC^{-1} of biochar.

Cost. The cost of cropland and grazing land sequestration given by McKinsey & Co. (2009), which was recalculated from Smith *et al.* (2008), ranges from about −165 $ tCeq. $^{-1}$ (for tillage and residue management) to around $40 tCeq. $^{-1}$ The cost of cropland and grazing (for degraded land restoration and improved agronomy – values converted from € tCO$_2$-eq. $^{-1}$, all $ values are US$ for decade 2000–2010).

Estimates for costs of production to application of biochar (assuming UK conditions) are about −830 to around 1200 $ tCeq. $^{-1}$ (converted from GB£ tCO2-eq. $^{-1}$ values in Shackley *et al.*, 2011). The mean of this range is 185 $ tCeq. $^{-1}$ A tighter cost range is derived from values in Meyer *et al.* (2011), using costs of 51 to 386 $ tbiochar^{-1} (for yard waste and retort charcoal, respectively), giving cost ranges of 54–406 $ tCeq. $^{-1}$ if 95% C content from slow pyrolysis is assumed, or 100–757 $ tCeq. $^{-1}$ if 51% C content from torrefaction is assumed. The mean of these ranges are ~230 and 430 $ tCeq. $^{-1}$, respectively. There may be economic benefits from biochar application that offset some of these costs (Dickinson *et al.*, 2014), but the benefits differ by region.

All per-t-Ceq. impacts of SCS and biochar on land use, water, nutrients, albedo, energy and cost are shown in Fig. 2.

Global estimates of impacts of SCS and biochar

Global implementation levels for SCS range from 0.4 to 0.7 GtCeq. yr^{-1} at carbon prices ranging from ~70 to 370 $ tCeq. $^{-1}$ (equivalentto20–100 tCO$_2$-eq. $^{-1}$; Smith *et al.*, 2008), with a technical potential of 1.3 GtCeq. yr^{-1} Woolf *et al.* (2010) report sustainable potential for avoided emissions for biochar, after accounting for competition for nonwaste biomass, to be ~1 GtCeq. yr^{-1}, with a maximum technical potential of 1.8 GtCeq. yr^{-1}. Of these avoided emissions, about 30% derives from fossil fuel displacement, and about 70% is from biochar storage in soil – so the negative emission component of biochar is 0.7 GtCeq. yr^{-1}, with a maximum technical potential of 1.3 GtCeq. yr^{-1}. The global impacts on land use, water, nutrients, albedo, energy and cost for these levels of implementation are presented below.

Land use. For implementation levels of SCS at C prices of up to ~370 $ tCeq. $^{-1}$, 0.7 GtCeq. yr^{-1} of SCS would require 700 (min) and 23100 (max) Mha. For implementation at maximum technical potential, the land area would be 1300 (min) to 42900 (max) Mha. Biochar implemented at 0.7 (constrained) and 1.3 (maximum theoretical) GtCeq. yr^{-1} of negative emissions would have a land footprint for spreading of biochar (assuming application rates of 50 t ha^{-1}; Genesio *et al.*, 2012) of 14 and 26 Mha, respectively.

For SCS and biochar, however, this land can remain in its current land use, and both SCS and biochar can be practiced without competing for land. The requirement for feedstock to produce biochar, however, does have a land footprint. Of the biochar potential of ~0.7 GtCeq. yr^{-1}, 0.3 GtCeq.yr is derived from dedicated biomass crops (Woolf *et al.*, 2010), which is proposed to occur on 50% of abandoned, degraded cropland that is not used for other purposes (Woolf *et al.*, 2010). This 0.3 GtCeq. yr^{-1} of negative emissions from biochar would require 40–260 Mha, assuming biomass yields of lignocellulosic energy crops (e.g. Miscanthus: 5.8–8.6 t Ceq. ha^{-1} yr^{-1}; Smith *et al.*, 2015), and that 20% (fast pyrolysis) to 88% (torrefaction) of the biomass C will be converted to biochar C (Meyer *et al.*, 2011). If yields were lower on this degraded land, the area required to deliver these negative emissions from biochar would increase proportionally.

Water use. Water use by SCS and biochar is assumed to be negligible, so global impact on water of both technologies is estimated to be ~zero.

Nutrient impact. For implementation levels of SCS at 0.7 GtCeq. yr^{-1} of negative emissions, SCS would add an additional 56 Mt N yr^{-1}, 14 Mt P yr^{-1} and 10.5 Mt K yr^{-1} to the soil. Equivalent values for 1.3 GtCeq. yr^{-1} of negative emissions from SCS are 104, 26 and 19.5 Mt yr^{-1} of N, P and K, respectively. Biochar implemented at 0.7 GtCeq. yr^{-1} of negative emissions would add 21 Mt N yr^{-1}, 7 Mt P yr^{-1} and 49 Mt K yr^{-1} to the soil. Equivalent values for 1.3 GtCeq. yr^{-1} of negative emissions from biochar are 31, 13 and 91 Mt yr^{-1} of N, P and K, respectively.

Albedo. Global impacts on albedo of SCS are zero, and for areas on which biochar is spread (14 Mha for implementation at 0.7 GtCeq. yr^{-1}), albedo could be reduced

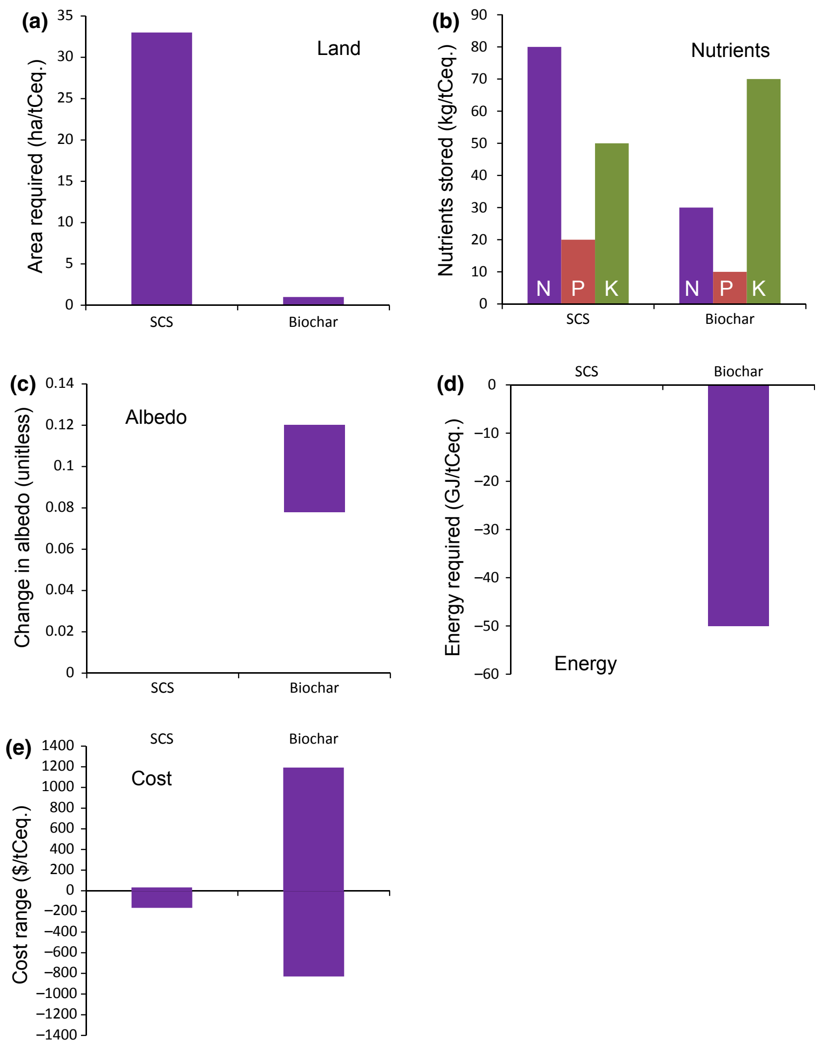

Fig. 2 Impacts of SCS and biochar on land use (a), nutrients (b), albedo (c), energy (d) and cost (e), expressed on per-t-Ceq. basis. Additional water use not shown as it is zero for SCS and biochar.

by 0.08 to 0.12. To contextualize this change in albedo, the mean annual albedo reduction of 0.05 calculated for application of 30–32 t ha^{-1} biochar to a test field in Germany by Meyer *et al.* (2012) was estimated to reduce the overall climate mitigation benefit of biochar application by 13–22% Meyer *et al.* (2012).

Energy. Global energy costs/benefits from SCS are zero, whereas biochar implemented at 0.7 GtCeq. yr^{-1} of negative emissions could produce 14–35 EJ yr^{-1} energy, with maximum theoretical deployment at 1.3 GtCeq. yr^{-1} of negative emissions producing up to 65 EJ yr^{-1} energy.

Cost. About 20% of the mitigation from SCS is realized at negative cost (−165–0 \$ tCeq. $^{-1}$) and about 80% realized is between 0 and 40 \$ tCeq. $^{-1}$ (calculated from values in McKinsey & Co., 2009; global marginal abatement cost curve for the agriculture, which was recalculated from values in Smith *et al.*, 2008). Implementation of SCS at 0.7 GtCeq. yr^{-1} of negative emissions (i.e. at C price up to 370 \$ tCeq. $^{-1}$) would save 7.7 B\$, comprising 16.9 B\$ of savings, and 9.2 B\$ of positive costs (values from Smith *et al.*, 2008). A marginal abatement curve is not available for biochar so total costs/savings cannot be calculated, but the global range for implementation at 0.7 GtCeq. yr^{-1} of negative emissions would range from −581 B\$ (if all costs were at the low end of cost estimate range) to 1560 B\$ (if all costs were at the high end of cost estimate range). Assuming the cost to be the mean of the range of per-t-Ceq. costs (185 \$ tCeq. $^{-1}$) given in Shackley *et al.* (2011), costs of implementation for biochar would be 130 B\$.

Discussion

SCS and biochar have lower negative emission potential (0.7 GtCeq. yr^{-1}) than some other NETs, with BECCS able to deliver 3.3 GtCeq. yr^{-1} by 2100 (mean from IPCC AR5 database; Smith *et al*., 2015) and DAC also able to deliver this level of negative emissions, although the potential is comparable to that of AR (1.1 GtCeq. yr^{-1} by 2100) and is greater than that for EW (0.2 GtCeq. yr^{-1} by 2100; Smith *et al*., 2015). Considering soil-based NETs is timely, given the proposal of the French Government to attempt to increase global soil carbon stocks by 0.4% as a climate mitigation measure, and their ability to improve soil quality and productivity (Lal, 2004).

Additional land requirements are low for SCS or biochar application, as it is possible to implement SCS and spread biochar without competition for land, although land required to grow biomass feedstock for biochar could be 40–260 Mha of land. This is greater than the land requirement for DAC (negligible excluding land for renewables to provide energy) and EW (2 Mha), but ranges from an order of magnitude smaller to similar, to the land required to implement 3.3 and 1.1 GtCeq. yr^{-1} of negative emissions through BECCS (380–700 Mha) and AR (320 Mha), respectively (Smith *et al*., 2015).

While the full GHG balance of biochar application to soils is very feedstock and technology specific, and uncertain (between −144 and 17 kgC tC^{-1} in biomass feedstock, assuming C content of biomass to be 50% of dry matter; calculated from values in Meyer *et al*., 2011), the magnitude and uncertainty of GHG reductions/emissions is not large enough to prevent use of biochar as a NET, although more comprehensive studies are required to assess the full climate impact (e.g. by including albedo impacts, see below, or addition of black carbon (soot) to the atmosphere; Meyer *et al*., 2011).

Water requirements for SCS and biochar are zero, which means that they are the only NETs unlikely to have a water footprint. EW has a low water use (0.3 $km^3\ yr^{-1}$), and DAC has moderate to high water use (10–300 $km^3\ yr^{-1}$), whereas AR (370 $km^3\ yr^{-1}$) and BECCS (720 $km^3\ yr^{-1}$) each has a very large water footprint when implemented at 1.1 and 3.3 GtCeq. yr^{-1}, respectively (Smith *et al*., 2015).

Unlike DAC and EW, nutrient retention from SCS and biochar is significant, with nutrient storage measured in Mt N, P or K yr^{-1}. In comparison, AR stores several orders of magnitude less nutrients, measured in kt N yr^{-1}. The impact of BE/BECCS on ecosystem nutrient storage is variable and depends on the biomass removal regime and the vegetation that the BE crop replaces (Smith *et al*., 2015). Although nutrient storage contributes to the (supporting) ecosystem service of nutrient cycling (UK National Ecosystem Assessment, 2011), and nutrient storage in soils helps to prevent water and air pollution (e.g. reducing ammonia emissions, and nitrate and phosphorus leaching to water courses; Smith *et al*., 2013), this nutrient immobilization could be considered a cost, if additional fertilizer is required to provide plant available nutrients for crop growth. In the long term, however, the additional nutrients in the organic matter provide a net ecosystem benefit (Smith *et al*., 2013).

SCS has no influence on albedo, but biochar can darken the soil surface due to the addition of substantial quantities of dark material to the soil (Genesio *et al*., 2012), which has been shown to increase soil temperature measurably. Increases relative to unamended soils are of the order of 0.08–0.12 and could cover an area of 14 Mha for 0.7 GtCeq. yr^{-1} of negative emissions. DAC and EW have no albedo effect while the impact of BE/BECCS is context specific (depending on the BE crop, the vegetation replaced and the latitude), although AR is known to decrease albedo significantly when conifers replace other vegetation at high latitudes (Calvin *et al*., 2014; Smith *et al*., 2015).

The energy impact SCS is negligible, but biochar can produce energy during its production by pyrolysis. Implementation at 0.7 GtCeq. yr^{-1} of negative emissions would deliver 14–35 EJ yr^{-1}, with maximum theoretical implementation at 1.3 GtCeq. yr^{-1} producing 65 EJ yr^{-1}. In this respect, both SCS and biochar are favoured over DAC and EW, each of which has a significant energy requirement, and similar to AR which is also neutral with respect to energy (Smith *et al*., 2015). However, it delivers around 5–12 times less energy than BE/BECCS, but can be combined with BE/BECCS (Woolf *et al*., 2010), so that BE/BECCS delivers energy and CO_2 capture (for BECCS), and biochar is produced in the process. There is, of course, a trade-off between energy yield and biochar production, with minimum energy penalties from biochar production of 21 to 33 GJ tC^{-1} when producing liquid and gaseous biofuels, respectively (Woolf *et al*., 2014).

SCS could be delivered at estimated costs between −165 and 40 \$ tCeq. $^{-1}$ (McKinsey & Co., 2009), and it is estimated that much of the negative emissions could be delivered at negative cost (−16.9 B\$ yr^{-1}), and the rest at low (9.2 B\$ yr^{-1}) cost, with an overall saving of 7.7 B\$ yr^{-1}. Assuming the cost to be the mean of the range of per-t-Ceq. costs for biochar (185 \$ tCeq. $^{-1}$), costs of implementation for biochar at 0.7 GtCeq. yr^{-1} would be 130 B\$ yr^{-1}. The cost saving from SCS is the only NET which appears to have the potential to be net negative in terms of cost, whereas the cost of biochar is

Table 3 A comparison of the global impacts of SCS and biochar, with other NETs reviewed in Smith *et al.* (2015). BECCS cost is investment need for electricity and biofuels by 2050; see Smith *et al.* (2015) for further details

NET	Realistic (max) global C removal (GtCeq. yr^{-1})	Additional land requirement (max) (Mha)	Additional water requirement (max) (km^3 yr^{-1})	Mean (max) nutrient impact (Mt N, P, K yr^{-1})	Albedo impact (unitless)	Energy requirement (max) (EJ yr^{-1})	Estimated cost (B$)	Reference
BECCS	3.3	380–700	720	Variable	Variable	−170	138/123	Smith *et al.* (2015)
DAC	3.3	Very low (unless solar PV used for energy)	10–300	None	None	156	≫ BECCS	Smith *et al.* (2015)
EW	0.2 (1.0)	2 (10)	0.3 (1.5)	None	None	46	>BECCS	Smith *et al.* (2015)
AR	1.1 (3.3)	320 (970)	370 (1040)	2.2 (16.8)	Negative; or reduced GHG benefit where not negative	Very low	≪BECCS	Smith *et al.* (2015)
SCS	0.7 (1.3)	0	0	N:56, P:14, K:10.5 (N:104, P:26, K:19.5)	0	0	−7.7	This study
Biochar	0.7 (1.3)	40–260	0	N:21, P:7, K:49 (N:31, P:13, K:91)	0.08–0.12	−14 to −35 (−65)	130	This study

comparable to the investment needs to BE/BECCS of 123–138 B$ yr^{-1} in 2050 (Smith *et al.*, 2015). Costs for DAC and EW are much higher, and the cost of AR is low (Smith *et al.*, 2015).

A summary of the global impacts of SCS and biochar, and comparison with the other NETs reviewed in Smith *et al.* (2015) is given in Table 3.

A summary graphic, comparing the impacts of SCS and biochar on land use, water, nutrients, albedo, energy and cost, and the comparison with other NETs, is presented in Fig. 3.

SCS and biochar therefore provide negative emissions with fewer potential disadvantages than many other NETs. Although the negative emission potential is lower than for DAC and BECCS, it is not insignificant and is comparable to the potential for AR and greater than for EW (Smith *et al.*, 2015). The potential for biochar could be enhanced by combining with BECCS, so that biochar is produced as a coproduct of energy generation from biomass (Woolf *et al.*, 2010), and the system can be modified to favour carbon allocation either to CO_2 for CCS or biochar for use as a soil amendment (Woolf *et al.*, 2014). Given that the same feedstock is used for biochar production and BE/BECCS, the negative emission potentials for the two options are not additive.

SCS and biochar negative emission potentials are assessed here independently and are not additive, as

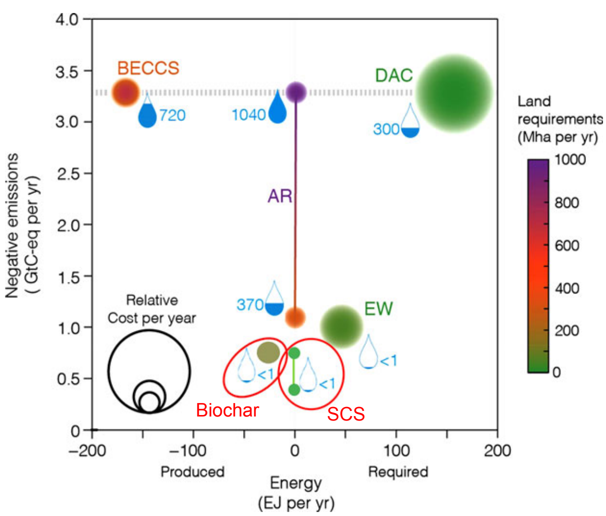

Fig. 3 Comparison of the global impacts of SCS and biochar and other NETs on land use, water, nutrients, albedo, energy and cost. NETs to which SCS and biochar are compared are BECCS, DAC, EW and AR as assessed by Smith *et al.* (2015). Water requirement shown as water drops with quantities in km^3 yr^{-1}; all other units are indicated on the figure. Adapted from Smith *et al.* (2015).

some SCS options rely on retention of crop/forest residues and additions of organic matter to the soil (Smith *et al.*, 2008; Smith, 2012), which means that these amendments would not be available as feedstocks for biochar production. On the other hand, if these residues are used for biochar production, they may cause a decline in soil organic carbon content (Woolf *et al*., 2010).

A drawback of SCS, which also affects AR but not other NETs, is that of sink saturation. We express SCS negative emission potential here as a yearly value, but the potential is time limited. SCS potential is large at the outset, but decreases as soils approach a new, higher equilibrium value (Smith, 2012), such that the potential decreases to zero when the new equilibrium is reached. This sink saturation occurs after 10–100 years, depending on the SCS option, soil type and climate zone (slower in colder regions), with IPCC using a default saturation time of 20 years (IPCC, 1997, 2006). As sinks derived from SCS are also reversible (Smith, 2012), practices need to be maintained, even when the sink is saturated so any yearly costs will persist even after the emission potential has reduced to zero at sink saturation. Sink saturation also means that SCS implemented in 2020 will no longer be effective as a NET after 2040 (assuming 20 years for sink saturation). The importance of this for NETs is that NETs are most frequently required in the second half of this century (Fuss *et al*., 2014; Smith *et al*., 2015), so SCS, like AR, may no longer be available after 2050, or will be less effective, if they are implemented for mitigation relatively soon.

The same issues associated in sink saturation apply partly to biochar, although the issue is less pronounced as biochar is more recalcitrant, and equilibrium (if it occurs) would be expected to take much longer, so that biochar should still be effective as a NET in the second half of this century even if implemented relatively soon. Some authors have suggested that biochar may induce a priming effect, such that organic matter decomposed more rapidly when biochar is add, but evidence for this effect is inconclusive (Zimmerman *et al*., 2011).

This analysis has shown that SCS and biochar have potential for delivering negative emissions and have some advantages over other NETs with respect to other biophysical, energy and cost impacts. However, neither SCS nor biochar production is represented in the current generation of IAMs. Given the negative emission potential of SCS and biochar and their potential advantages compared to other NETs, efforts should be made to include these options within IAMs, so that their potential can be explored further in comparison with other NETs.

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