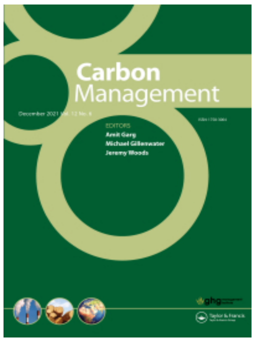

Carbon Management

ISSN: (Print) (Online) Journal homepage: https://www.tandfonline.com/loi/tcmt20

Soil carbon sequestration potential bounded by population growth, land availability, food production, and climate change

Sonja G. Keel, Daniel Bretscher, Jens Leifeld, Albert von Ow & Chloé Wüst-Galley

To cite this article: Sonja G. Keel, Daniel Bretscher, Jens Leifeld, Albert von Ow & Chloé Wüst-Galley (2023) Soil carbon sequestration potential bounded by population growth, land availability, food production, and climate change, Carbon Management, 14:1, 2244456, DOI: 10.1080/17583004.2023.2244456

To link to this article: https://doi.org/10.1080/17583004.2023.2244456

CARBON MANAGEMENT
2023, VOL. 14, NO. 1, 2244456
https://doi.org/10.1080/17583004.2023.2244456

Taylor & Francis
Taylor & Francis Group

OPEN ACCESS Check for updates

Soil carbon sequestration potential bounded by population growth, land availability, food production, and climate change

Sonja G. Keel[a], Daniel Bretscher[a], Jens Leifeld[a], Albert von Ow[b] and Chloé Wüst-Galley[a]

[a]Climate and Agriculture Group, Agroscope, Zurich, Switzerland; [b]Economic Modelling and Policy Analysis Group, Agroscope, Ettenhausen, Switzerland

ABSTRACT

Improving soil management to enhance soil carbon sequestration (SCS)—a cost-efficient carbon dioxide (CO_2) removal approach—can result in co-benefits or trade-offs. Here we address this issue by setting up a modeling framework for Switzerland that combines soil carbon (C) storage, food production and agricultural greenhouse gas (GHG) emissions. The link to food production is crucial because crop types and livestock numbers influence soil organic C (SOC) stocks, through soil C inputs from plants and manure. We estimated SCS rates for the years 2020–2050 for three scenarios, each with two variants for biochar: cover cropping (0.30 t CO_2 equivalents [CO_2-eq] $ha^{-1} yr^{-1}$), biochar addition (0.36–1.8 t CO_2-eq $ha^{-1} yr^{-1}$) and agroforestry-biochar addition (2.2–2.3 t CO_2-eq $ha^{-1} yr^{-1}$). Different limiting factors (land and biomass availability, population growth) affected SCS rates and indicated that they cannot be sustained until 2100 under all scenarios (cover cropping: 0.10 t CO_2-eq $ha^{-1} yr^{-1}$ [2051–2100]; biochar addition: 0.35–1.8 t CO_2-eq $ha^{-1} yr^{-1}$; agroforestry-biochar addition: 1.0–1.7 t CO_2-eq $ha^{-1} yr^{-1}$). This information together with the associated GHG emissions is critical for planning net zero strategies and highlights the importance of integrated assessments that capture links between SCS and the food system.

ARTICLE HISTORY
Received 11 January 2023
Accepted 16 July 2023

KEYWORDS
Net zero; carbon dioxide removal; dietary change; biochar; nature-based solutions; agriculture; modelling; greenhouse gas mitigation

Introduction

More than 70 countries have announced net zero emission targets (https://www.un.org/en/climate-change/net-zero-coalition). To reach this goal, most will rely on carbon dioxide removal (CDR) or negative emissions to offset emissions that are unavoidable or difficult to reduce [1]. Because capacities for different approaches and emission sources differ largely [2], it will be necessary to develop country-specific strategies to reach net zero that favor domestic compensation. Policymakers therefore need realistic potentials for different CDR approaches that account for local conditions.

Soil carbon sequestration (SCS) is a cost-efficient CDR method [3, 4] with many co-benefits [5]. It is defined as a net uptake of atmospheric carbon dioxide (CO_2) that leads to an increase in soil organic carbon (SOC) storage within the same region from which CO_2 was taken up by photosynthesis [6]. Although carbon (C) is not permanently stored, natural C sinks including afforestation and reforestation have an atmospheric cooling effect [7, 8]. Owing to their high technology readiness level, they could make an important contribution to rapid upscaling of CDR, which is critical to limit peak temperatures [9].

In recent years, a wealth of studies has been published related to SCS, especially after the 4p1000 initiative was launched (https://www.4p1000.org/). The initiative highlighted that an annual increase of 4‰ in SOC in the first 30 to 40 cm of soil would significantly reduce atmospheric CO_2 concentrations. This statement initiated a debate on whether SCS could indeed make a significant contribution to climate change mitigation. Important concerns related to insufficient nitrogen availability to form new soil organic matter [10–12], potential trade-offs caused by enhanced nitrous oxide (N_2O) emissions [13, 14] or the reversibility of C sinks were raised [11, 15]. Furthermore, assessing potentials for SCS on agricultural land is especially

CONTACT Sonja G. Keel sonja.keel@agroscope.admin.ch
Supplemental data for this article can be accessed online at https://doi.org/10.1080/17583004.2023.2244456.

challenging. In addition to environmental factors such as soil conditions and climate, SOC stocks are influenced by a wide array of effects linked to management. These factors include the type of land use (cropland or grassland [16]), which crop is cultivated [17], the duration of soil cover, whether crop residues are left on the field [18] and whether organic fertilization (e.g. manure or slurry) is applied [12], as these determine the amount of C that is returned to the soil. In the case of organic fertilization, the quality of C also matters [19] because organic fertilizers are usually stored and eventually processed (composted, digested) before being applied. During these processes, C can be lost and fractions more resistant to decomposition may remain or form. This results in lower C turnover rates of these substrates when applied to soils as compared with additions of plant material. An extreme case is biochar. This C-rich material produced through thermal conversion of biomass can be applied as a soil amendment and has a high intrinsic stability, minimizing C losses through decomposition [20].

To determine how much additional C can be stored in soils, the use of a valid baseline is crucial. This baseline should include the current management, so that SOC storage resulting from soil conservation practices that are already applied will not be accounted for twice (i.e. the actual effect should be additional). Furthermore, it is important for realistic projections that the baseline accounts for expected environmental and main societal changes. While climate change effects have been considered in earlier studies [21], population growth and increasing food demand have received little attention in this context. Additionally, limitations in biomass availability due to its low production or competing uses should be taken into account [21, 22]. Together, the variability in these factors within or across regions or countries also underpins the need to provide country-specific estimates and approaches.

To quantify SCS and account for these multiple factors, we have set up a modeling framework that links soil C simulations with a food supply model that considers population growth (Figure 1). The food supply model simulates calorie production, can optimize diets and considers future demand on agricultural area and livestock. Livestock numbers are used to estimate the amount of farmyard manure as C input to soil, methane (CH_4) emissions from enteric fermentation and manure management as well as N_2O emissions resulting from the storage and application of manure to soils by using a GHG model. This model allows trade-offs or benefits of changes in agricultural production and diets related to GHG emissions to be estimated. This framework is set up for Switzerland, a densely populated country.

By the year 2100, the Swiss population is expected to have grown by 43% (https://www.bfs.admin.ch/bfs/en/home/statistics/population/population-projections/national-projections.html). We first estimate minimal changes in diets that would allow sufficient food for a larger population to be provided until the year 2100. Import and export of food and feed are assumed to remain the same. This forms the baseline scenario. Direct effects of climate change projected by nine different model chains and three emission scenarios on SOC stocks are included in this baseline and all other scenarios.

Next, we assess the SCS rates for three separate scenarios (Figure 1) by comparing their SOC stocks with those of the baseline scenario. The cover crop scenario represents an increased use of a well-known soil conservation practice. Cover crops, also named inter-crops or catch crops, are planted between two main crops when soils would otherwise be bare. They contribute to higher C inputs and can reduce C losses. Additionally, we quantify SCS rates for biochar additions. We use national estimates for sustainably available biomass (from landscape maintenance or low-quality so-called forest energy wood [23]) to calculate how much biochar could be produced with domestic resources that are not from agricultural land ("biochar scenarios"). Furthermore, we assess the potential of growing additional biomass for biochar production on agricultural land that is made available through reductions in feed production as a result of changes in diets ("agroforestry-biochar scenarios"). On this "free cropland", fast-growing trees are planted alongside agricultural fields and are harvested for biochar production. For these scenarios we estimate annual and cumulative SCS rates for the years 2020–2100.

Materials and methods

Description of national-scale soil organic carbon inventory

To estimate changes in SOC stocks, we used a model-based, national-scale SOC inventory for agricultural, mineral topsoils (0–30 cm depth) that was

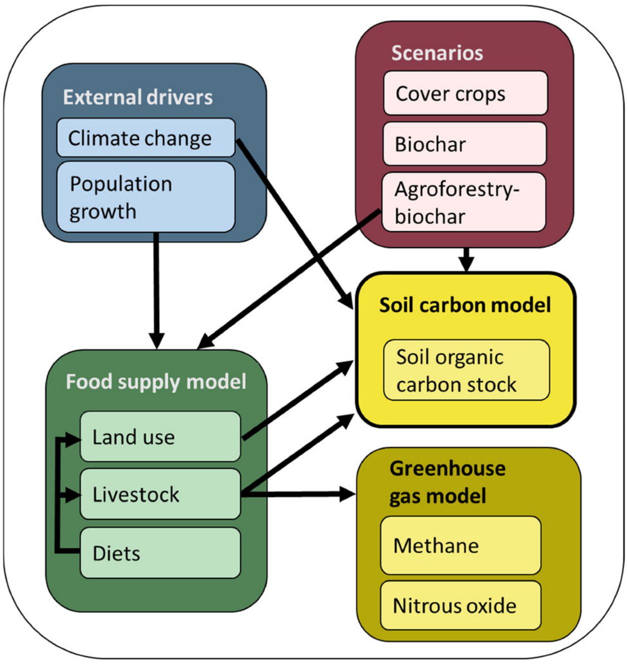

Figure 1. Modeling framework for assessment of SCS in agricultural soils including trade-offs and benefits related to changes in food production.

developed for GHG reporting under the United Nations Framework Convention on Climate Change [24]. The system is based on the SOC model RothC and incorporates the management of the nineteen most important crop types grown in Switzerland and six agricultural grassland categories of varying management intensity. We used the function RothCModel in the R package SoilR [25], modified by adding two biochar pools (see section "Parameterization of biochar decomposition and estimation of biochar yield"). The code will be made available on Zenodo. Residue management, cover crops and applications of organic amendments are accounted for—the latter incorporating livestock numbers as well as crop and grassland types.

An allometric equation [26], adapted to measurements made in Switzerland [27, 28], is used to derive the amount of plant C inputs to the soil based on measured annual yields. The clay content of the soil is derived from a soil suitability map. To calculate initial SOC stocks, an approach that relates SOC stocks to clay content, elevation and land use type is used [24, 29]. The initial sizes of the different C pools in the RothC model are estimated using a pedo-transfer function [30], which proved to be a good alternative to spin-up estimations [24]. Upscaling to the national level is done by stratifying the country into 240 regions with similar pedo-climatic conditions and agricultural production types.

Monthly resolved, crop-specific application rates of organic amendments (OrgAm) were calculated as a function of annual OrgAm availability and distribution among nineteen crop types and six grassland categories [24]. OrgAm availability in Switzerland is a function of herd sizes, animal-specific OrgAm excretion rates, animal housing (determining the type of OrgAm), straw addition (used in deep litter systems), net removals to anaerobic digesters and OrgAm-C loss due to storage. OrgAm application to different crops or grassland types was calculated as a function of farmer behavior, the movement of cattle, horses, sheep, goats and pigs to summer pastures, and the permissible fertilization rates of the different crops and grasslands, the latter also incorporating management intensity and elevation. OrgAm on grasslands also incorporates legislation on the allowed fertilization of extensive meadows and summer pastures. Crops assumed to receive OrgAm are cereals, maize (silage and grain), potatoes, beets, sunflowers and rape seed (oil). For simulations for 2020 onwards, new OrgAm application rates were calculated: These calculations used future herd sizes, crop and grassland surfaces and straw availability (as a function of cereal cultivation), as projected by the food supply model DSS-ESSA (see section "The DSS-ESSA food supply model"). All other parameters (given above) were held constant, using the 2019 values.

Climate change and emission scenarios

Simulations were run for the period 1990–2100. Daily mean air temperature (at 2 m aboveground) and precipitation values for this entire period were obtained from the CH2018 Climate Scenarios [31], more specifically the DAILY-LOCAL data product, which contains values for individual weather stations. The Climate Scenarios are based on regional climate models (RCMs) from the European Coordinated Regional Downscaling Experiment climate simulations [32]. The RCMs are themselves derived from model chains comprising different global climate models and RCMs. Downscaling to the individual weather stations is done using quantile mapping, using station observations as reference observations (utilizing data from between 85 and 399 stations for temperature variables and precipitation, respectively), and using the period 1981–2010 as a reference period. For this period, the statistical distribution of the (simulated) daily values matches that of the observed values, but there is no temporal correspondence of the predictions with observations. This means, for example, that a drought as strong as the 2003 drought is not necessarily predicted to occur in 2003.

Monthly mean temperature (T_{mean}) and monthly precipitation sums (*PPN*) were derived from the Climate Scenarios data product. Monthly evapotranspiration (*ET*) sums were calculated using the Hargreaves function (*ETH*) [33], using daily temperature minimum (T_{min}) and maximum (T_{max}) values also obtained from the Climate Scenarios data product. The Hargreaves function is known to result in biased values under certain meteorological conditions. We checked for biases, comparing the *ETH* values with *ET* values as calculated using the Priestley and Taylor function [34] (*ETP-T*)—which has been shown to provide an unbiased *ET* estimate in the Swiss plateau [35]—for the period 1990–2019 (data required to calculate *ETP-T* were available for this period but not beyond it) for four agriculturally important regions in Switzerland. A bias was indeed found and was corrected as follows: For each station (see last paragraph of this section) a linear regression of *ETH* values against *ETP-T* values was performed, and the resulting regression coefficients were used to correct the *ETH* values.

The mean values from nine model chains were used in this study (DMI-HIRHAM_ECEARTH_EUR11, KNMI-RACMO_HADGEM_EUR44, MPICSC-REMO1_MPIESM_EUR11, MPICSC-REMO2_MPIESM_EUR11, SMHI-RCA_ECEARTH_EUR11, SMHI-RCA_HADGEM_EUR44, SMHI-RCA_MPIESM_EUR44, SMHI-RCA_MIROC_EUR44, SMHI-RCA_NORESM_EUR44), considering the representative concentration pathways (RCPs) 2.6, 4.5 and 8.5.

The 24 regions of this study were represented by one weather station each. Comparison of T_{mean} and *PPN* parameters (1990–2019) between the agriculturally most important six regions and individual weather stations showed that this is an appropriate strategy. In the Climate Scenarios data product, temperature (T_{mean}, T_{min} and T_{max}) and *PPN* have been predicted for 82 weather stations. Stations representing the 24 regions were chosen out of these 82 stations based on the following criteria: Regions containing only one weather station (fewer than 10 cases) were allocated that station; where multiple weather stations occur in a region, the station furthest away from the edges of that region was chosen; weather stations with atypical elevation (mostly mountain tops) were not chosen.

Parameterization of biochar decomposition and estimation of biochar yield

To parameterize biochar decomposition, we followed a concept that describes biochar as composed of two C pools, a small labile and a large stable pool, which both decompose following first-order kinetics with a respective intrinsic rate constant [36]. The size of the smaller, labile biochar pool can be estimated from biochar properties and CO_2 release during shorter-term decomposition studies. It was shown based on incubation data [37] that the amount of C released during one year is closely related to the thermal stability of the biochar [38] (called "recalcitrance index" by these authors; Supplementary information, Fig. S1).

According to federal regulations, biochar quality must conform to the European Biochar Certificate to be amendable to Swiss agriculture. Samples from Fig. S2 (Supplementary information) with this certification have molar ratios of hydrogen (H) to C (H/C ratios) of 0.25 (±0.03 SE), which is slightly above those reported for hardwood and coniferous woodchip biochars [39]. We used this H/C ratio for our calculations, which corresponds to a recalcitrance index of 0.53. Using the equations presented in Figs. S1 and S2 (Supplementary information), the size of the labile pool for a reference temperature of $+9.3\,^{\circ}C$ (corresponding to a temperature rate modifying factor of 1 in RothC) is **0.2**%. This pool fully decomposes within one year, and its rate constant $k1$ per year at $9.3\,^{\circ}C$ is set to

10.0, corresponding to the decay constant of the decomposable plant material pool in RothC. The size of the labile pool was smaller than the estimate of 5–10% in a previous study [36]; however, some of the studies from which these authors had derived pool sizes utilized more labile biochars, i.e. biochars produced at lower temperatures. The approach followed here, instead, provides a universal analytical measure for estimating the labile pool size.

Correspondingly, the size of the stable pool was 100% − 0.2% = **99.8**%. Its decomposition rate constant was derived from a double exponential decay model

$$C(t) \ = C1^{-k1t} + C2^{-k2t} \tag{1}$$

with $C1$ and $C2$ being the sizes and $k1$ and $k2$ the rate constants of the labile and the stable pool, respectively. To get $k2$, the equation was fitted using the data in the Intergovernmental Panel on Climate Change guidelines for national GHG inventories [40], where the amount of biochar-C remaining after 100 years of decay is reported as a function of pyrolysis temperature. Because in practice pyrolysis temperature is often not known, we again followed an approach to derive this important parameter from measurable biochar properties. We used data compiled in a previous study that reports both pyrolysis temperature and the biochar H/C ratio [41] (Supplementary information, Fig. S3). More than 63% of the variability in predicted pyrolysis temperature could be explained by the H/C ratio of the produced biochar. The H/C ratio of 0.25 derived above corresponds to a pyrolysis temperature of about 660 °C, which translates into 90% of biochar-C remaining after 100 years. Fitting Equation 1 to these data gives a rate constant $k2$ of 0.0009 per year for a decomposition temperature of 20 °C, corresponding to a $k2$ of **0.00047** when calculated for a temperature of 9.3 °C (the temperature where the rate modifier in RothC equals one). This parameter value, which is the most relevant for simulating the fate of biochar in soil in the long term, is in the range of rate constants reported in earlier studies [36, 37, 42–44]. The advantage of our approach is that all relevant parameters for simulating biochar decomposition can be derived from just the molar H/C ratio, allowing the parameters to be adjusted to the biochar at hand.

Biochar yields were estimated based on pyrolysis efficiencies and C contents of biochar. Pyrolysis efficiencies (i.e. amount of biochar produced per unit feedstock material, w/w) for woody materials of 23.1% (±1.0% SE; $n = 28$) were found for processes >500 °C when produced at the laboratory scale [41]. We omitted pyrolysis data from processes with lower temperatures because biochar used in Swiss agriculture is processed at higher temperatures. The yield of 23.1% is slightly below pyrolysis efficiencies for wood in medium-scale pyrolysis units [45] as the ones used in Switzerland (**26.4**% ± 4%). The corresponding C contents of the produced biochars from wood at the laboratory scale were **75.3**% ± 1.7% [41], corresponding to C-pyrolysis efficiencies of biochar production of approx. 41%, assuming a C content of wood of 48%. For the medium-scale unit, the C- pyrolysis efficiency was 44% ± 7% [45]. For our calculations, we used the values highlighted in bold that are typical for biochar production in Switzerland (C-pyrolysis efficiency for biochar production is not needed for the calculation).

The DSS-ESSA food supply model

We used the food supply model DSS-ESSA ("DSS" stands for decision support system, while "ESSA" is a German acronym for "Ernährungs-Sicherungs-Strategie Angebotslenkung", which means food security strategy supply management) to simulate potential changes in crop shares and livestock numbers associated with shifts in dietary composition [46]. These shifts occur as a consequence of increasing food demand due to a growing population and additionally due to replacement of feed crops by agroforestry in case of the agroforestry-biochar scenarios. DSS-ESSA is an optimization model that simulates the Swiss food supply sector including crop and livestock production, food processing and storage, foreign trade and food consumption (Supplementary information, Fig. S4). It identifies the optimal allocation of the production resources based on the available quantities of resources including public and private sector food stocks, simultaneously taking into account production constraints and nutritional requirements. Essential food production activities, including foreign trade, food processing and stock management, are depicted in the model.

DSS-ESSA is a linear programming (LP) model. An LP model maximizes or minimizes an objective function whilst fulfilling a series of constraints (Equations 2 and 3). In DSS-ESSA, these constraints or restrictions include mainly interdependencies within production processes, such as minimal feeding requirements or maximal yields in food processing. When solving the optimization problem with an algorithm, a mathematical optimization solver is needed to identify the variable values of the optimal solution. DSS-ESSA uses the Gurobi 9.1 solver (https://www.gurobi.com/), in which the variables are assigned to those real and non-negative numerical values that lead to the maximum (or minimum) objective function value.

$$\text{Objective function}: \sum_{j=1}^{J} c_j x_j \rightarrow \max \qquad (2)$$

x_j Variables (quantities)
c_j Contribution to objective value (per unit x_j)
J Number of variables

$$\text{Constraints}: \sum_{j=1}^{J} a_{nj} x_{nj} \leq b_n \ (\text{for } n = 1, \ldots, N) \qquad (3)$$

a_{nj} Production coefficients (quantity of resource n per unit of x_j; e.g. labor needs per hectare of wheat)
b_n Capacities (quantities) or fixed factors
N Number of capacities or relationships

DSS-ESSA is a dynamic recursive LP model, which means that the value of a model variable (typically an annual value) is linked to its value in the previous period. Thus, the model takes into account development over time, e.g. considering the rearing of young animal stock by means of transferring the population from one age category to the next. The main data sources of the model are the official Swiss agricultural and foreign trade statistics [47, 48]. Specific technical data have been taken from agricultural and food industry reports [49, 50]. Multi-annual averages are calculated in the case of annually fluctuating data such as yields and import quantities. Data that are unlikely to change in the short term are only updated on occasion (e.g. nutritional recommendations [51]); most of these data are from 2015 [46].

The assumptions made for the simulations (especially population growth, limited area and constant import quantities) required a change in domestic agricultural production and food consumption until 2100 in order to meet the increasing demand. The diet was also changed so that the discrepancy between the current diet and the national nutritional recommendations would be reduced.

Description of baseline scenario

Simulations of SOC stocks were performed for the years 1990–2100. The years 1990–2019 were used as a warm-up phase, and only data for 2020–2100 are presented. Some basic input data were kept constant for simulations of future years, including total grassland and cropland areas (year 2017 was used as a reference) and, except for the cover crop scenario, C inputs to the soil from cover crops (up to 0.71 t C $ha^{-1} yr^{-1}$ with the amount depending on the rotation [lowest value: 0.52 t C $ha^{-1} yr^{-1}$]; only added after summer crops, distributed over five months). Crop yields were calculated as the average of yields from 2014 to 2018. Changes in crop shares simulated with DSS-ESSA affect the simulation of SOC stocks because different crops are associated with different amounts of C inputs from plants and from OrgAm, the latter according to the common management in Switzerland (i.e. some crops mostly receive organic fertilizer, while others mainly receive mineral fertilizer that does not contain any C). Additionally, C inputs to the soil from organic fertilizer changed through time, responding to variations in herd sizes and thus amounts of excreta.

Description of C sequestration scenarios

For the SCS scenarios, we focus on cropland only because potentials to store additional C in Central Europe and globally are highest on cropland [52, 53] and some measures (e.g. cover crops) cannot be applied to grassland. The SCS scenarios were run as described for the baseline scenario (above) except for changes described below, which were applied from 2020 onwards. The amount of SOC that could be sequestered during the years 2020–2100 was calculated as difference between SOC stocks of a specific scenario and the SOC stocks of the baseline scenario (Supplementary information, Fig. S5). Here, SCS represents the combined effect of additional CO_2 uptake and compensation of potential SOC losses that occur over the simulated period, which when defined in this way does not follow the definition cited in the introduction [6].

Cover crop scenario

For the cover crop scenario, we assumed that the abundance and biomass production of cover crops could be increased. Cover crops can be ploughed under as green manure before sowing the next main crop or be harvested for feed. They enhance overall biomass input to the soil and reduce decomposition and erosion because the soil remains covered. We assumed C inputs could be increased from 0.52–0.71 t C $ha^{-1} yr^{-1}$ to 2.5 t C $ha^{-1} yr^{-1}$ through the use of more diverse and productive seed mixtures and more frequent use of cover crops. This value was estimated based on the most frequently sold cover crop mixture in Switzerland and biomass data for single cover crops [54]. Compared with the baseline scenario, C inputs to the soil from cover crops were 3.6- to 4.7-fold higher, depending on their current abundance.

Biochar scenarios

For the two biochar scenarios, we estimated potential biochar production based on sustainable potentials of domestic biomass use [23]. For the biochar I scenario, we included only biomass from the category "wood from landscape maintenance" (wood from pruning in pastures and meadows, gardens, parks, road and track sides). For the biochar II scenario, we additionally included green waste from households and landscape (0.2 Mt yr^{-1}), the untreated fraction of waste wood (0.024 Mt yr^{-1}; assuming 24% is untreated, Vanessa Burg, personal communication) and forest energy wood (parts of stem and branch wood, brushwood and bark; 0.6 Mt yr^{-1}). To estimate biochar production, a pyrolysis efficiency of 26.4% ± 4% and a C content of the produced biochar of 75.3% was assumed. To estimate biochar application rates (Supplementary information, Table S1), the amount of biochar that could be produced annually (constant for 2020–2100) was divided by the annual cropland area receiving OrgAm (between 376,290 and 379,750 ha). For SOC simulations, we applied biochar at annual rates. Although biochar can be used on grassland, we focused on the application on cropland only, because biochar can be ploughed under to reduce losses by erosion. Potential effects of biochar on GHG emissions (e.g. reduced N_2O emissions from soils [20]) were not accounted for.

Agroforestry-biochar scenarios

For the two agroforestry-biochar scenarios, enhanced woody biomass production on agricultural land was made possible through changes towards more plant-based diets. We maintained the current per person calorie supply and did not allow food imports to increase. The objective function of the food supply model was set to maximize the cropland area no longer needed for feed or food production ("free cropland"). The "free cropland" surface that became available as a result of reduced feed production was assumed to be used for agroforestry. We calculated two versions of this scenario: For the agroforestry-biochar I scenario, we assumed that a minimum share of 20% grass–clover leys has to be maintained in the rotation (current share is 31%). Leys are temporary grasslands that are typical elements of Swiss crop rotations and are important to prevent erosion. In the agroforestry-biochar II scenario, no more leys were present in crop rotations to minimize feed production on cropland. The goals of both scenarios were to i) maximize the amount of wood that can be allocated to processing biochar, and ii) apply easily removable forms of agroforestry given that the available area for wood production varies over time. Short rotation coppice (SRC) was considered the best form of agroforestry in this context. The annual aboveground increment has been estimated at 3.2 ± 1.3 t C $ha^{-1} yr^{-1}$ with yields slightly increasing in the second and subsequent cycles [40]. The increment was independent of species, and the most suitable trees in the temperate zone were willow (*Salix* spp.), poplar (*Populus* spp.) or black locust (*Robinia pseudoacacia*), with typical rotation lengths of five years [55]. A more recent estimate for various SRC poplar varieties in Germany [56] revealed aboveground wood yields of up to 5.5 t C $ha^{-1} yr^{-1}$ (based on a C content of 50% dry matter [d.m.]), with many varieties reaching 4.0 t C $ha^{-1} yr^{-1}$. For our calculation, we used the latter value. We calculated the wood growth per hectare, assuming trees planted as hedgerows (e.g. narrow field strips with one to two rows of trees) along agricultural fields rather than field-wide monocultures. The same assumptions were used for pyrolysis efficiency and the C content of agroforestry-biochar as described above for the biochar scenarios.

For simulations of SOC stocks, we additionally estimated annual plant C inputs to the soil from these agroforestry systems. Young poplar stands were estimated to release 0.17 t C $ha^{-1} yr^{-1}$ *via* fine root turnover [57]. In addition, weeds, which are abundant in SRC, released 0.45 t C $ha^{-1} yr^{-1}$, resulting in a total belowground input of 0.62 t C $ha^{-1} yr^{-1}$. The latter is close to the estimate of 0.56 t C $ha^{-1} yr^{-1}$ for silvopasture systems in temperate Europe [40]. Annual aboveground litter in poplar SRC was 1.5 ± 0.3 t C $ha^{-1} yr^{-1}$ in central Spain [58], between 2.1 and 2.6 t C $ha^{-1} yr^{-1}$ (based on a litter C content of 40%) in central Italy [59] and between 1.4 and 1.5 t C $ha^{-1} yr^{-1}$ in central France [60] (second year of cultivation). The latter study also revealed strong differences in litter amounts between the first and second year of production. We considered an annual rate of 1.4 t C $ha^{-1} yr^{-1}$ as conservative. In terms of decomposition rates, residue inputs from SRC (above- and below-ground) were treated the same as any other input from regular annual or perennial agricultural crops.

Calculation of changes in methane and nitrous oxide emissions

We estimated changes in the dominant sources of CH_4 and N_2O for the baseline and the agroforestry-biochar I and II scenarios, as these are strongly affected by changes in animal numbers and diets. For the cover crop and the biochar I and II scenarios, CH_4 and N_2O emissions are assumed to be equal to the baseline as also diets are the same. To estimate emissions, the model of the Swiss national agricultural GHG inventory was used [61]. The emission estimates thus comprise all sources from agricultural activities on farms as covered by the respective system boundaries such as CH_4 emissions from enteric fermentation and N_2O emissions from agricultural soils [62]. For each scenario, livestock populations were adjusted according to the respective outputs of the DSS-ESSA model. For reasons of simplicity all other parameters were held constant (e.g. emission factors). Emissions from feed production abroad were not covered here.

Results

Projected changes until 2100 in the baseline scenario

We first estimated least changes in diets and agricultural production necessary to cover the food demand of a larger population until the year 2100 without increasing imports of food and feed. This estimate forms the baseline scenario. Under this condition, the intake of energy-rich food commodities (i.e. sugar, oil and alcoholic beverages), which currently makes up about half of the calorie intake (blue colors in Figure 2a), was reduced by 30%, while potato and dairy product consumption increased (Figure 2a, “Base” vs. “Curr”). This led to an increase in the area used for bread grain and potato cultivation and a reduction in the extent of fodder cereals and rapeseed (Figure 2b). The existing grassland was used for an increase in dairy farming at the expense of the husbandry of other cattle (e.g. rearing of cattle, suckler cow husbandry). Although feed imports remained unchanged, this resulted in an increase in livestock numbers dominated by dairy cows and other ruminants and a smaller increase in pigs (Figure 2c, “Base” vs. “Curr”). Accordingly, livestock emissions increased by 25% in the case of CH_4 and by 14% for N_2O (Figure 2f). The increase in livestock numbers also led to 12% higher C inputs to soils through organic matter additions (mainly farmyard manure, but also straw from animal bedding) (Figure 3). Although this is an important source of C for soils, the soil C model predicted significant losses in SOC stocks for the baseline scenario mainly in response to increasing temperatures as also climate change is accounted for (0.11 t C $ha^{-1} yr^{-1}$ on permanent grassland and 0.064 t C $ha^{-1} yr^{-1}$ on cropland for the emission pathway RCP8.5 for the years 2020–2100; Supplementary information, Fig. S6).

Soil carbon sequestration rates on Swiss cropland soils

For the cover crop scenario, we assumed cover crops were planted more often and had higher productivity than at present. All other conditions such as land use, climate change and livestock numbers remained as in the baseline scenario. Hence, resulting GHG emissions are unchanged (Figure 2f). SCS rates were rather small (Table 1; Figure 4a) and were lowest for RCP8.5 (i.e. highest temperature increase; Supplementary information, Fig. S7) compared with RCP4.5 or RCP2.6. Values are already above zero in 2020 because the model has a monthly time step but we present SCS rates as annual averages. Towards the year 2100, a new steady state was reached and annual SCS rates were around zero, even including negative rates in some years (Figure 4b).

For the biochar scenarios, we quantified SCS rates for the application of biochar produced with domestic, non-agricultural biomass. All other conditions (e.g. land use, diets, climate change) were the same as in the baseline simulation. Compared with the cover crop scenario, SCS rates were twice as high for the biochar I scenario and 10 times higher for biochar II (Table 1; Figure 4c), even taking account of losses during biochar manufacture. These higher rates can be explained by enhanced C inputs and the high stability of biochar. Compared with the cover crop scenario, differences between different climate model chains were minimal and SCS rates hardly changed over time

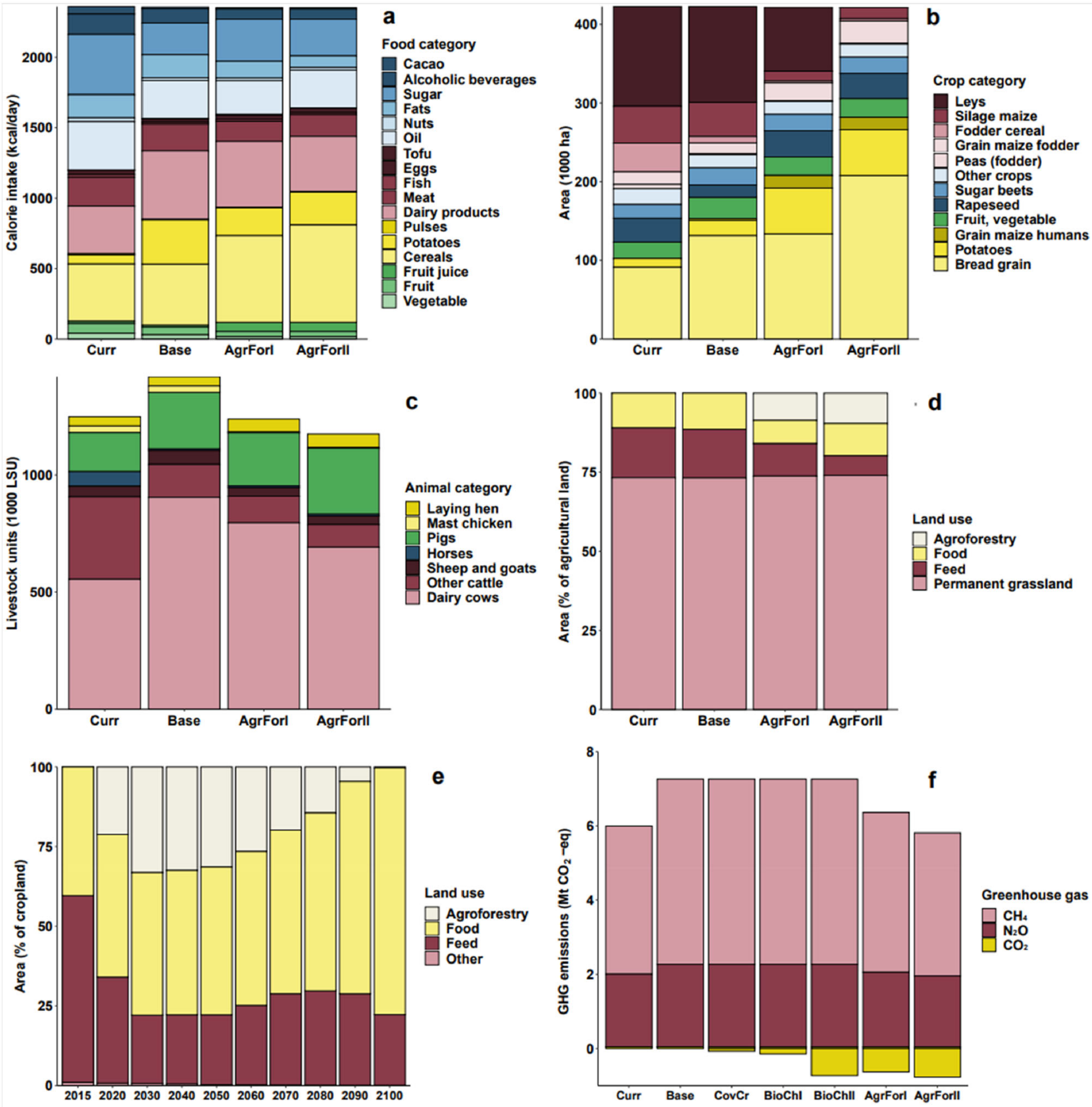

Figure 2. **Diets (a), cropland use (b), livestock numbers (c), and agricultural land use (d) projected by the food supply model for current conditions (curr), a baseline simulation for the year 2100 (base) and two variants of the agroforestry-biochar scenario in the year 2100 (AgrForI and AgrForII). For the AgrForII scenario, we additionally show the temporal change in cropland use (e). Agricultural GHG emissions (f) estimated based on livestock numbers, fertilizer use and SCS rates (i.e. negative emissions) for the year 2100 of three additional soil management scenarios: cover crops (CovCr), biochar additions (BioChI, BioChII). For the latter three scenarios, CH_4 and N_2O emissions are the same as for the baseline because no changes in diets are assumed. SCS rates of AgrFor scenarios account for changes in C inputs to the soil affected by biochar additions, changes in cropland use and livestock numbers. Results in panels a–d for the cover crop and biochar-only scenarios would be identical to the baseline scenario and are therefore not shown. Please note that diets are presented in calories and not in portions (as is more common). This explains why the share of fruits and vegetables is very low.**

(Figure 4d). We did not account for potential reductions in SOC at the place of biomass origin, i.e. system boundaries were not extended. Therefore, the presented SCS rates for biochar-only scenarios likely represent overestimates.

For the agroforestry-biochar scenarios, we first estimated how much agricultural land could be made available to grow additional woody biomass through changes in diets. Currently, 60% of the Swiss cropland is used for feed production (Figure 2d, e). A moderate reduction in meat consumption of less than 30% (see below) would allow up to 36% of the Swiss cropland to be replaced with agroforestry during the next decades (Figure 2e) as feed production is reduced (i.e. lower fraction of leys, silage maize and other fodder cereals such as barley). Results are only presented for agroforestry-biochar II, but are very

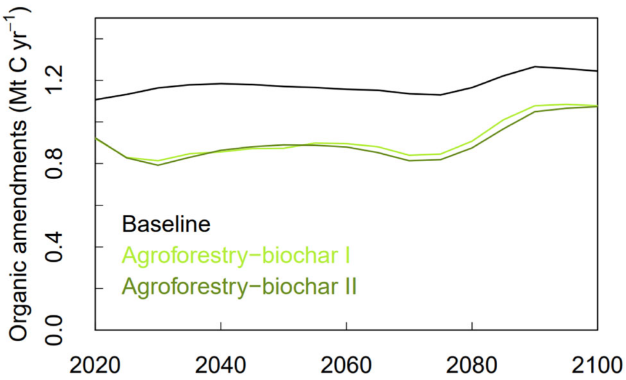

Figure 3. Amount of organic amendments (mainly farmyard manure, but also straw from animal bedding or digestate from biogas production) produced at national scale in response to changes in herd sizes associated with dietary changes and growing food demand (baseline). For the agroforestry-biochar scenarios, we additionally assumed that feed production and livestock numbers are reduced.

Table 1. SCS rates on Swiss cropland for different scenarios.

Scenario	Annual SCS (t C ha^{-1} yr^{-1})[a]	Relative SOC changes (‰)[b]	Annual SCS for total cropland area (Mt CO_2-eq yr^{-1})
Cover crops	0.05 (−0.01 to 0.20)	1.0	0.075
Biochar I (only from landscape maintenance)	0.10 (0.09–0.10)	1.9	0.14
Biochar II (additional biomass, e.g. from low-quality forest wood)	0.50 (0.47–0.51)	9.8	0.73
Agroforestry-biochar I (frequency of leys in crop rotations is reduced from 31% to 20%)	0.43 (−0.30 to 0.75)	8.5	0.63
Agroforestry-biochar II (no more leys)	0.53 (−0.18 to 0.88)	10.4	0.77

Notes: Mean annual SCS rates were calculated as differences in SOC stocks between the shown scenarios and the baseline for the years 2020–2100. Average results for nine model chains and three emission pathways are shown.
[a]Numbers in parenthesis are the ranges across 80 years. Negative numbers indicate SOC losses relative to the baseline.
[b]Change compared with the average SOC stock of 51 t C ha^{-1} (for the year 2020 at 0–30 cm depth) for cropland soils.

similar for agroforestry-biochar I. The results only consider the surface where trees are planted, so the total agroforestry area including cropland would be about 10 times larger. Owing to increasing land requirements for food production in response to population growth, the agroforestry area starts declining already after the year 2030; first at a slow, towards the year 2100 at a fast rate (Figure 2e).

In both agroforestry-biochar scenarios, the production and intake of meat projected for the year 2100 decreased compared with the baseline scenario (−25% and −19% for scenario I and scenario II, respectively; Figure 2a, "AgrFor" vs. "Base"), resulting in a lower demand for feed production on cropland (pink colors in Figure 2b). The extent of dairy farming was also lower than in the baseline scenario. Hence the number of dairy cows decreased (Figure 2c) and the intake of dairy products was 4% and 19% lower for scenarios I and II, respectively. In contrast, a larger part of cropland was used for the production of food for human consumption such as grains or potatoes (yellow colors in Figure 2b). Compared with the current situation, meat consumption decreased by 30% and 24% for agroforestry-biochar scenarios I and II, respectively (Figure 2a, "AgrFor" vs. "Curr"), whereas dairy consumption increased (by 38% and 16% for agroforestry-biochar scenarios I and II, respectively). Overall, diets were healthier (lower intake of sugar, oil and fat, and alcoholic beverages) and more starch-based (higher intake of cereals and potatoes).

As a result of these changes, agricultural GHG emissions were lower than in the baseline scenario. Under the agroforestry-biochar I scenario, reductions of 14% for CH_4 and 10% for N_2O were achieved, and even stronger reductions were found for scenario II (CH_4: −23%, N_2O: −14%). However, owing to population growth, GHG emissions slightly increased compared with the current situation under the agroforestry-biochar I scenario (CH_4: +8%, N_2O: +3%). With the agroforestry-biochar II scenario, it was possible to stabilize GHG emissions over time (CH_4: −3%, N_2O: −3%).

To estimate potential biomass production of agroforestry on this area, we assumed fast-growing trees (i.e. SRC) would be planted in narrow rows along agricultural fields (on cropland formerly used for feed production). Annually, they could provide up to 1.0 Mt d.m. yr^{-1} (agroforestry-biochar I scenario) or 1.1 Mt d.m. yr^{-1} (agroforestry-

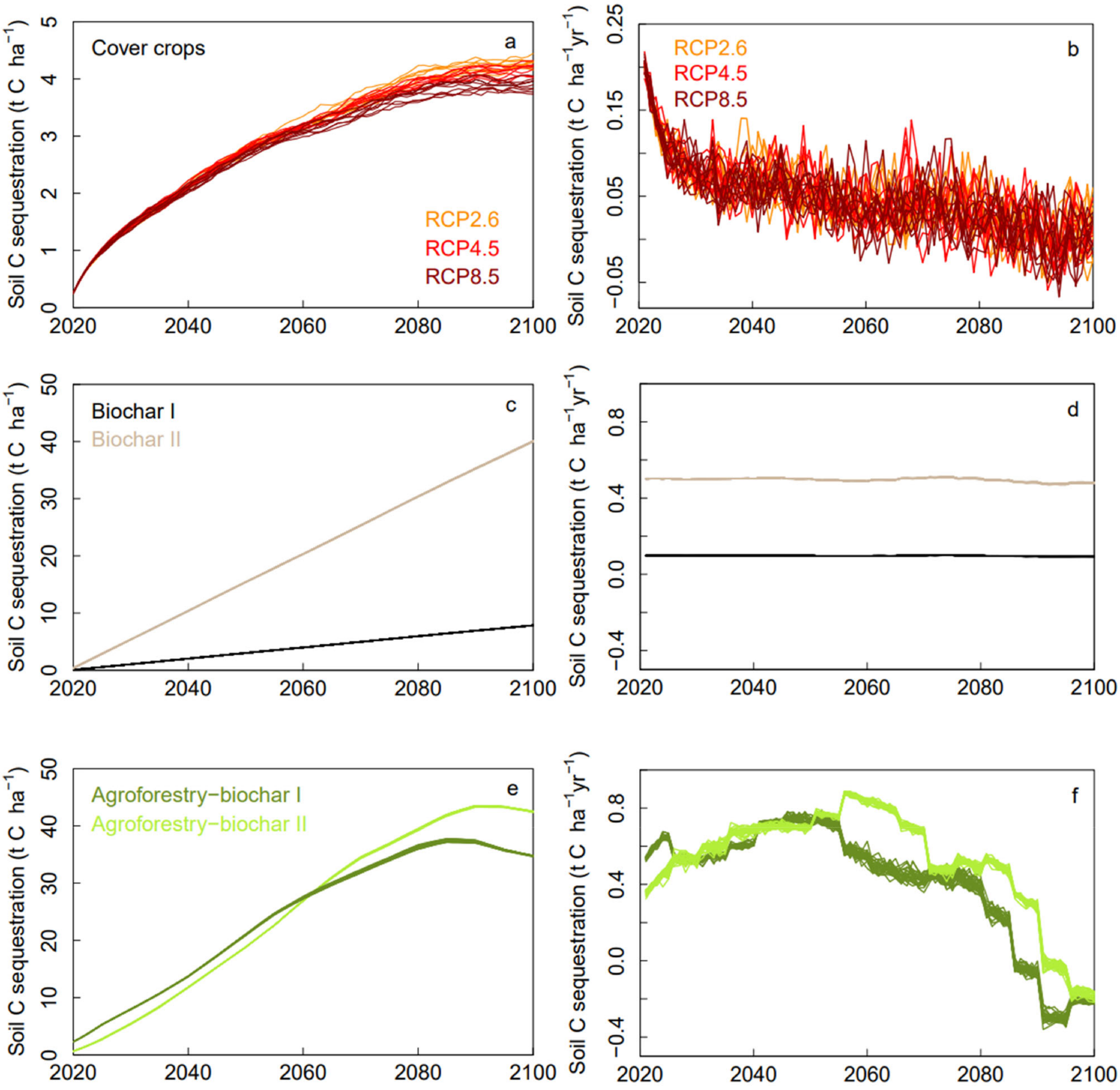

Figure 4. **Cumulative (left-hand side) and annual (right-hand side) SCS rates for three soil management scenarios: cover crops (a, b), biochar addition (produced from non-agricultural biomass) (c, d), biochar addition from agroforestry biomass (e, f). SCS rates are for mineral topsoils (0–30 cm) and were calculated against SOC stocks of a baseline scenario. Because SOC stocks are predicted to decline under the baseline scenario, the net sequestration (i.e. net atmospheric CO_2 removal) would be lower. Predictions for nine different climate model chains (lines) and three different emission pathways (different colors in a and b) are shown. Curves mostly overlay each other in the case of the biochar and agroforestry-biochar scenarios. For the biochar I scenario, only biomass from landscape maintenance was used to produce biochar. For biochar II, additionally forest energy wood (low-quality wood) and green waste from households were used. For the agroforestry-biochar scenarios, cropland is made available to grow trees for biochar production through moderate reductions in meat production and changes towards more plant-based diets. Please note the different scales on the y-axis.**

biochar II scenario) of biomass, which is about the same amount used in the biochar II scenario (1.0 Mt d.m. yr^{-1}). Technical assumptions regarding biochar production and distribution on cropland were the same as in the biochar-only scenarios. For SOC simulations we accounted for changes in plant-derived C inputs to the soil associated with changes in crop areas (Figure 2b) because crops differ largely regarding the amount of harvest residues. Owing to an expansion of rapeseed and grain maize cultivation—two crops with high amounts of residues—, the total amount of plant C input increased. In contrast, changes in livestock composition and slightly reduced herd sizes (Figure 2c) led to a decrease in manure and hence lower C inputs from organic fertilization (Figure 3).

Resulting SCS rates of the agroforestry-biochar scenarios were in the same range as the biochar II scenario if averaged over the years 2020–2100 (Table 1; Figure 4c, e). However, due to the temporal change in biochar production associated with the availability of land for agroforestry (Figure 2e), the temporal dynamics of SCS were different. Until about the year 2050, rates increased under the agroforestry-biochar scenarios and were mostly higher than in the biochar II scenario (Figure 4d, f). After 2050, SCS rates started to decrease owing to a conversion of agroforestry land to cropland and lower biochar production rates linked to less biomass.

Discussion

Our study provides quantitative and realistic national-scale estimates of SCS in agricultural, mineral topsoils of Switzerland. By expanding the system boundary to include the food supply system, we identified population growth as a potentially limiting factor in addition to biomass and land availability. Still, our results suggest that SCS could account for up to 13% of Switzerland's negative emission goal (Supplementary information, Table S1) and thereby significantly contribute to achieving net zero emissions in the year 2050. Thereafter, our results show that depending on which measure is chosen, SCS rates are predicted to decrease—an important information for designing sustained net zero emission strategies. While the reduction of SCS rates driven by reaching a new steady state of SOC for the cover crop scenario is a well-documented and expected outcome [63], potential reductions in response to population growth have so far received little attention. This knowledge gap suggests that comprehensive analysis—as the recent study by Sun et al. [64] that assessed GHG mitigation potentials of dietary changes for high-income countries—need to be expanded to account for the potentially growing food demand. Otherwise, such studies risk to overestimate the CDR potential of SCS. It is important to add that the effects of population growth we found are strongly driven by the assumption that food imports do not increase with population size. However, because of the war in the Ukraine and its impact on food supply in Europe and beyond, the topic of self-sufficiency has gained relevance and supports our assumption.

Extending the system boundaries to the agricultural sector allowed us to detect co-benefits for GHG mitigation associated with changes in land use and diets. While the intention of the agroforestry-biochar scenario was to make cropland available to produce more biomass, it also resulted in significant reductions in CH_4 and N_2O through lower livestock numbers due to reduced meat consumption. This result was not surprising, as it is well known that plant-based food is associated with lower GHG emissions [65–67]. However, so far studies that included all GHGs mainly focused on specific products or production systems [68, 69] but rarely included the entire agricultural sector of a country [70]. Quantifying SCS rates and agricultural emissions can support decision making, because comparisons as the one shown in Figure 2f identify the most effective measures. In contrast, studies that include additions of livestock for SCS but only report soil C balances [71] clearly overestimate the total climate benefit of such a management change, which might in reality lead to increases in total GHG emissions.

Though our results are specific to one country, a similar approach could be applied in other countries. We highlight the critical importance of including region-specific information and the value of integrated assessments. Global-scale approaches applied so far [64] likely use too general assumptions to assess realistic SCS rates for the following reasons: First, dietary changes need to account for local production conditions and food preferences. Owing to Switzerland's hilly topography, about 75% of the agricultural land in Switzerland can only be used as permanent grassland for grazing or fodder production. This explains why the Swiss diet clearly differs from the global, sustainable EAT-Lancet diet (e.g. higher fraction of dairy products in Switzerland) as applied in the study by Sun et al. [64] and will affect SCS rates through different predictions in land use. Second, our study emphasizes the importance of current soil management conditions (i.e. an appropriate baseline for estimating SCS rates). The fact that cover cropping is already partly used in Switzerland and is limited by current crop rotations (Swiss crop rotations are dominated by winter crops, which reduces bare soil phases available for cover crops) explains why our rates are six times lower than the rate of 0.32 t C $ha^{-1}yr^{-1}$ based on a global meta-analysis [72], but are in line with a study that applied a similar approach in Germany [73].

Average SCS rates of the biochar II and the two agroforestry-biochar scenarios were similar, but the underlying reasons were different and highlight the importance of linking a soil C model to a food system model. Whereas SCS in the biochar-only scenario was exclusively explained by biochar additions, the rates of the agroforestry-biochar scenarios were also the result of dietary changes influenced by changing crop areas. Owing to an expansion of grain maize (for human consumption) and rapeseed surfaces, C inputs to the soil increased because both crops produce large amounts of harvest residues. This example illustrates that we would have underestimated the SCS potential of the agroforestry-biochar scenarios, had we not linked the soil C with the food supply model. On the contrary, SCS rates would have been overestimated if we had not accounted for the lower C inputs from animal excreta (Figure 3) in response to reduced livestock numbers.

Overall, SCS rates for the biochar II and the two agroforestry-biochar scenarios are higher than expectations for Europe [74], and also higher than the goals of the 4p1000 initiative [75]. Biochar amendment was so far not discussed as a widespread measure, explaining this discrepancy. Theoretically the three management options we present are additive and even higher SCS rates could be achieved, but adjustments in the design of the scenarios would be needed to avoid even higher application rates of biochar per land area (e.g. expanding biochar additions to permanent grassland). Compared with increasing plant biomass additions, biochar amendments have several advantages: Theoretically, there is no upper limit in the resulting SOC stocks, the risk of C loss through mineralization is minimal, and the risk of priming is low. Indeed, negative priming has often been observed (i.e. reduced decomposition of soil organic matter), whereas positive priming has mainly been documented for nutrient-poor soils [76] (less applicable in Switzerland). However, a few open questions remain because field experiments or large-scale applications are scarce. This is critical because biochar persists in the soil for centuries. Although pyrogenic C occurs naturally in most soils and makes up 14% of SOC on average [77], studies are necessary to assess long-term effects on nutrient cycling and effects on soil fauna because negative results have been documented [78]. It is also important to note that biochar can contain toxic substances [79] and only quality-controlled biochar should be applied. Finally, large amounts of biomass are needed for biochar production, resulting in competing interests. Studies that compare C efficiencies of soil C storage including losses during production of biochar are urgently needed. Additionally, life cycle assessments for different types of biomasses (e.g. forest wood, urban waste) are required that consider local conditions (e.g. to account for transport distances) and show which the most sustainable type of use is.

Our findings potentially have important implications for the development of net zero strategies. The results we present aim to guide researchers in how to improve SCS estimates given local conditions by illustrating how soil C storage is linked to land use and food production. Thanks to a global effort led by the Food and Agriculture Organization of the United Nations, researchers worldwide were recently trained to use the same soil C model applied here, to estimate national-scale SCS rates (https://www.fao.org/soils-portal/data-hub/soil-maps-and-databases/global-soil-organic-carbon-sequestration-potential-map-gsocseq/en/). Furthermore, our results are intended to support farmers, consumers and governments in reducing GHG emissions in the agricultural sector.

Limitations of this study

This analysis considers SCS only on cropland, neglecting potentials on permanent grassland for which no suitable measures have been identified yet in the Swiss context. Biochar could be applied on grassland but, to minimize losses through erosion, only when they are renewed (i.e. ploughed), which is not a recommended practice.

A further limitation of our study is that simulations with the soil C model RothC are restricted to topsoil. Any potential measures that address SCS in deeper soil layers (e.g. deep rooting crops) could not be studied but might have a significant SCS potential [80, 81]. Additionally, estimates on biochar stability are prone to large uncertainties because most data on biochar decomposability rely on relatively short-term incubations of only a few years [82]. Lastly, we only focus on the direct climate change effect on SOC stocks and did not account for potential changes in yields. Because we express SCS rates as differences between specific scenarios and a baseline, both of which use the same yields as input data, any yield change would probably have minor effects. Furthermore, for the most important crops in Switzerland belowground C inputs, which generally dominate plant inputs to soil, are independent of yields [27, 28].

Conclusion

Rates of SCS relevant to reaching net zero emissions in Switzerland by the year 2050 can be achieved in Swiss agriculture, despite many limiting factors. Depending on the measure chosen, SCS rates could decrease strongly thereafter. This calls for a rapid upscaling of other CDR approaches and for stronger efforts to further mitigate agricultural GHG emissions in general. Our results clearly show that emissions from livestock in Switzerland are large compared with potential SCS rates and changes in diets would be an effective mitigation strategy. Furthermore, we showed that it is critical to consider both current (i.e. agricultural practices) as well as future conditions (population growth, climate change) for quantifying realistic SCS rates. Additionally, applying a multidimensional approach that accounts for the links of soil C storage with food production and GHG emissions is important to capture benefits or trade-offs associated with changes in land use or livestock numbers. The modeling framework we propose is a first step in this direction and can be complemented by additional dimensions such as economic modeling in future studies.

Acknowledgments

We thank Christina Schädel for helpful comments on an earlier version of the manuscript.

Disclosure statement

No potential conflict of interest was reported by the authors.

ORCID

Sonja G. Keel http://orcid.org/0000-0002-2645-273X
Jens Leifeld http://orcid.org/0000-0002-7245-9852
Chloé Wüst-Galley http://orcid.org/0000-0002-8517-782X

Data availability statement

The output of the RothC model (soil organic carbon stocks) is available on Zenodo (https://doi.org/10.5281/zenodo.6413955) as well as the version of RothC including two biochar pools and two exogenous organic matter pools (https://doi.org/10.5281/zenodo.8218886).

Author contributions

SGK designed the study with contributions from JL and CWG. CWG produced input files for soil carbon modelling, JL derived parameters for biochar and SGK ran the soil carbon simulations. Simulations with the food security model were performed by AVO and DB estimated agricultural greenhouse gas emissions. SGK analyzed the data and wrote the paper with contributions from all authors.

References

1. IPCC. Climate change 2022: mitigation of climate change. In: Contribution of working group III to the sixth assessment report of the intergovernmental panel on climate change. Shukla PR, Skea J, Slade R, et al., editors. Cambridge University Press, Cambridge, UK and New York, NY, USA. doi: 10.1017/9781009157926; 2022.
2. van Soest HL, den Elzen MGJ, van Vuuren DP. Net-zero emission targets for major emitting countries consistent with the paris agreement. Nat Commun. 2021; Apr 912(1):2140. doi: 10.1038/s41467-021-22294-x.
3. Fuss S, Lamb WF, Callaghan MW, et al. Negative emissions-Part 2: costs, potentials and side effects. Environ Res Lett. 2018; 13(6):063002. doi: 10.1088/1748-9326/aabf9f.
4. Smith P. Soil carbon sequestration and biochar as negative emission technologies. Glob Chang Biol. 2016; 22(3):1315–1324. doi: 10.1111/gcb.13178.
5. Lal R. Soil carbon sequestration impacts on global climate change and food security. Science. 2004; 304(5677):1623–1627. doi: 10.1126/science.1097396.
6. Olson KR, Al-Kaisi MM, Lal R, et al. Experimental consideration, treatments, and methods in determining soil organic carbon sequestration rates. Soil Sci Soc Am J. 2014;78(2):348–360. doi: 10.2136/sssaj2013.09.0412.
7. Leifeld J, Keel SG. Quantifying negative radiative forcing of non-permanent and permanent soil carbon sinks. Geoderma. 2022;423:115971. doi: 10.1016/j.geoderma.2022.115971.
8. Sierra CA, Crow SE, Heimann M, et al. The climate benefit of carbon sequestration. Biogeosciences. 2021;18(3):1029–1048. doi: 10.5194/bg-18-1029-2021.
9. Matthews HD, Zickfeld K, Dickau M, et al. Temporary nature-based carbon removal can lower peak warming in a well-below 2 °C scenario. Commun Earth Environ. 2022;3:65. doi: 10.1038/s43247-022-00391-z.
10. van Groenigen JW, van Kessel C, Hungate BA, et al. Sequestering soil organic carbon: a nitrogen dilemma. Environ Sci Technol. 2017; 51(9):4738–4739. doi: 10.1021/acs.est.7b01427.
11. Baveye PC, Berthelin J, Tessier D, et al. The "4 per 1000" initiative: a credibility issue for the soil science community? Geoderma. 2018;309:118–123. doi: 10.1016/j.geoderma.2017.05.005.
12. Poulton P, Johnston J, Macdonald A, et al. Major limitations to achieving "4 per 1000" increases in soil organic carbon stock in temperate regions: evidence from long-term experiments at rothamsted research, United Kingdom. Glob Chang Biol. 2018; 24(6):2563–2584. doi: 10.1111/gcb.14066.
13. Lugato E, Leip A, Jones A. Mitigation potential of soil carbon management overestimated by neglecting N_2O emissions. Nat Clim Change. 2018;8(3):219–223. doi: 10.1038/s41558-018-0087-z.
14. Guenet B, Gabrielle B, Chenu C, et al. Can N_2O emissions offset the benefits from soil organic carbon storage? Glob Chang Biol. 2021; 27(2):237–256. doi: 10.1111/gcb.15342.
15. Powlson DS, Whitmore AP, Goulding KWT. Soil carbon sequestration to mitigate climate change: a

critical re-examination to identify the true and the false. Eur J Soil Sci. 2011;62(1):42–55. doi: 10.1111/j.1365-2389.2010.01342.x.

16. Poeplau C, Don A, Vesterdal L, et al. Temporal dynamics of soil organic carbon after land-use change in the temperate zone – carbon response functions as a model approach. Glob Change Biol. 2011;17(7):2415–2427. doi: 10.1111/j.1365-2486.2011.02408.x.
17. Johnston AE, Poulton PR, Coleman K. Soil organic matter: its importance in sustainable agriculture and carbon dioxide fluxes. In: Advances in agronomy. Vol 101. San Diego: Elsevier Academic Press Inc; 2009. p. 1–57.
18. Bolinder MA, Crotty F, Elsen A, et al. The effect of crop residues, cover crops, manures and nitrogen fertilization on soil organic carbon changes in agroecosystems: a synthesis of reviews. Mitig Adapt Strateg Glob Change. 2020;25(6):929–952. doi: 10.1007/s11027-020-09916-3.
19. Levavasseur F, Lashermes G, Mary B, et al. Quantifying and simulating carbon and nitrogen mineralization from diverse exogenous organic matters. Soil Use Manage. 2022;38(1):411–425. doi: 10.1111/sum.12745.
20. Lehmann J, Cowie A, Masiello CA, et al. Biochar in climate change mitigation. Nat Geosci. 2021;14(12):883–892. doi: 10.1038/s41561-021-00852-8.
21. Wiesmeier M, Poeplau C, Sierra CA, et al. Projected loss of soil organic carbon in temperate agricultural soils in the 21st century: effects of climate change and carbon input trends. Sci Rep. 2016; 6(6):32525–32525. doi: 10.1038/srep32525.
22. Riggers C, Poeplau C, Don A, et al. How much carbon input is required to preserve or increase projected soil organic carbon stocks in German croplands under climate change? Plant Soil. 2021;460(1–2):417–433. doi: 10.1007/s11104-020-04806-8.
23. Burg V, Bowman G, Erni M, et al. Analyzing the potential of domestic biomass resources for the energy transition in Switzerland. Biomass Bioenergy. 2018;111:60–69. doi: 10.1016/j.biombioe.2018.02.007.
24. Wüst-Galley C, Keel SG, Leifeld J. A model-based carbon inventory for Switzerland's mineral agricultural soils using RothC. Agroscope Sci. 2020;105:1–110.
25. Sierra CA, Müller M, Trumbore SE. Models of soil organic matter decomposition: the SoilR package, version 1.0. Geosci Model Dev. 2012;5(4):1045–1060. doi: 10.5194/gmd-5-1045-2012.
26. Bolinder MA, Janzen HH, Gregorich EG, et al. An approach for estimating net primary productivity and annual carbon inputs to soil for common agricultural crops in Canada. Agric Ecosyst Environ. 2007; 118(1–4):29–42. doi: 10.1016/j.agee.2006.05.013.
27. Keel SG, Anken T, Büchi L, et al. Loss of soil organic carbon in Swiss long-term agricultural experiments over a wide range of management practices. Agric Ecosyst Environ. 2019;286:106654. doi: 10.1016/j.agee.2019.106654.
28. Hirte J, Leifeld J, Abiven S, et al. Below ground carbon inputs to soil via root biomass and rhizodeposition of field-grown maize and wheat at harvest are independent of net primary productivity. Agric Ecosyst Environ. 2018; Oct265(265):556–566. doi: 10.1016/j.agee.2018.07.010.
29. Leifeld J, Bassin S, Fuhrer J. Carbon stocks in Swiss agricultural soils predicted by land-use, soil characteristics, and altitude. Agric Ecosyst Environ. 2005; 105(1–2):255–266. doi: 10.1016/j.agee.2004.03.006.
30. Weihermüller L, Graf A, Herbst M, et al. Simple pedotransfer functions to initialize reactive carbon pools of the RothC model. Eur J Soil Sci. 2013;64(5):567–575. doi: 10.1111/ejss.12036.
31. CH2018 Project Team. CH2018 – Climate scenarios for Switzerland. National Centre for Climate Services. 2018. doi: 10.18751/Climate/Scenarios/CH2018/1.0.
32. Jacob D, Teichmann C, Sobolowski S, et al. Regional climate downscaling over Europe: perspectives from the EURO-CORDEX community. Reg Environ Change. 2020;20(2):51. doi: 10.1007/s10113-020-01606-9.
33. Hargreaves GH, Samani ZA. Reference crop evapotranspiration from temperature. Trans ASAE. 1985; 1(2):96–99.
34. Priestley CHB, Taylor RJ. On the assessment of surface heat flux and evaporation using large-scale parameters. Mon Wea Rev. 1972;100(2):81–92. doi: 10.1175/1520-0493(1972)100<0081:OTAOSH>2.3.CO;2.
35. Calanca P, Smith P, Holzkämper A, et al. Die Referenzverdunstung und ihre Anwendung in der Agrarmeteorologie. Agrarforschung Schweiz. 2011; 2(4):176–183.
36. Woolf D, Lehmann J. Modelling the long-term response to positive and negative priming of soil organic carbon by black carbon. Biogeochemistry. 2012;111(1–3):83–95. doi: 10.1007/s10533-012-9764-6.
37. Zimmerman AR. Abiotic and microbial oxidation of laboratory-produced black carbon (biochar). Environ Sci Technol. 2010;44(4):1295–1301. doi: 10.1021/es903140c.
38. Harvey OR, Kuo LJ, Zimmerman AR, et al. An index-based approach to assessing recalcitrance and soil carbon sequestration potential of engineered black carbons (biochars). Environ Sci Technol. 2012; 46(3):1415–1421. doi: 10.1021/es2040398.
39. Camps-Arbestain M, Amonette JE, Singh B, et al. A biochar classification system and associated test methods. In: Lehmann J, Joseph S, editors. Biochar for environmental management. London: routledge; 2015. p. 165–194.
40. IPCC. Refinement to the 2006 IPCC guidelines for national greenhouse gas inventories. In: Calvo Buendia E, Tanabe K, Kranjc A, et al. editors. Switzerland: IPCC; 2019.
41. Al-Wabel MI, Hussain Q, Usman ARA, et al. Impact of biochar properties on soil conditions and agricultural sustainability: a review. Land Degrad Dev. 2018;29(7):2124–2161. doi: 10.1002/ldr.2829.
42. Kuzyakov Y, Bogomolova I, Glaser B. Biochar stability in soil: decomposition during eight years and transformation as assessed by compound-specific 14C analysis. Soil Biol Biochem. 2014;70:229–236. doi: 10.1016/j.soilbio.2013.12.021.
43. Lehmann J, Abiven S, Kleber M, et al. Persistence of biochar in soil. In: Lehmann J, Joseph S, editors.

Biochar for environmental management. Science, Technology and Implementation. Routledge; 2015. p. 235–282.
44. Jiang X, Denef K, Stewart CE, et al. Controls and dynamics of biochar decomposition and soil microbial abundance, composition, and carbon use efficiency during long-term biochar-amended soil incubations. Biol Fertil Soils. 2016;52(1):1–14. doi: 10.1007/s00374-015-1047-7.
45. Sørmo E, Silvani L, Thune G, et al. Waste timber pyrolysis in a medium-scale unit: emission budgets and biochar quality. Sci Total Environ. 2020; 718(718): 137335. doi: 10.1016/j.scitotenv.2020.137335.
46. von Ow A, Waldvogel T, Nemecek T. Environmental optimization of the Swiss population's diet using domestic production resources. J Cleaner Prod. 2020; 248:119241. doi: 10.1016/j.jclepro.2019.119241.
47. FSO. Federal statistical office. Farm structure census in 2019. Neuchâtel; 2020.
48. FCA. Swiss federal customs administration. Swiss-Impex foreign trade database. Berne; 2020.
49. Agristat. Statistical surveys and estimates on agriculture and nutrition, Swiss Farmers' Union. Brugg; 2020.
50. Swisspatat. Statistical data 2020 on potato cultivation and potato utilisation. Berne; 2020.
51. SSN. Swiss society for nutrition. Swiss food pyramid. Recommendations for a Healthy and Enjoyable Adult Diet. 2016;
52. Chen S, Arrouays D, Angers DA, et al. Soil carbon stocks under different land uses and the applicability of the soil carbon saturation concept. Soil Tillage Res. 2019;188:53–58. doi: 10.1016/j.still.2018.11.001.
53. Amelung W, Bossio D, de Vries W, et al. Towards a global-scale soil climate mitigation strategy. Nat Commun. 2020; 11(1):5427. doi: 10.1038/s41467-020-18887-7.
54. Wendling M, Büchi L, Amossé C, et al. Influence of root and leaf traits on the uptake of nutrients in cover crops. Plant Soil. 2016;409(1-2):419–434. doi: 10.1007/s11104-016-2974-2.
55. Hauk S, Knoke T, Wittkopf S. Economic evaluation of short rotation coppice systems for energy from biomass—a review. Renew Sustain Energy Rev. 2014;29: 435–448. doi: 10.1016/j.rser.2013.08.103.
56. Landgraf D, Carl C, Neupert M. Biomass yield of 37 different SRC poplar varieties grown on a typical site in North Eastern Germany. Forests. 2020;11(10):1048. doi: 10.3390/f11101048.
57. Berhongaray G, Janssens IA, King JS, et al. Fine root biomass and turnover of two fast-growing poplar genotypes in a short-rotation coppice culture. Plant Soil. 2013;373(1–2):269–283. doi: 10.1007/s11104-013-1778-x.
58. González I, Sixto H, Rodríguez-Soalleiro R, et al. Nutrient contribution of litterfall in a short rotation plantation of pure or mixed plots of *Populus alba* L. and *Robinia pseudoacacia* L. Forests. 2020;11(11): 1133. doi: 10.3390/f11111133.
59. Cotrufo MF, De Angelis P, Polle A. Leaf litter production and decomposition in a poplar short-rotation coppice exposed to free air CO2 enrichment (POPFACE). Global Change Biol. 2005;11(6):971–982. doi: 10.1111/j.1365-2486.2005.00958.x.
60. Guénon R, Bastien J-C, Thiébeau P, et al. Carbon and nutrient dynamics in short-rotation coppice of poplar and willow in a converted marginal land, a case study in Central France. Nutr Cycl Agroecosyst. 2016; 106(3):293–309. doi: 10.1007/s10705-016-9805-y.
61. Bretscher D. Agricultural sector of Switzerland's greenhouse gas inventory 1990-2019, federal office for the environment. Bern: FOEN; 2021.
62. IPCC. 2006 IPCC guidelines for national greenhouse gas inventories, prepared by the national greenhouse gas inventories programme. In: Eggleston S, Buendia L, Miwa K, et al. editors. Japan: Intergovernmental Panel on Climate Change; 2006.
63. Stewart CE, Paustian K, Conant RT, et al. Soil carbon saturation: concept, evidence and evaluation. Biogeochemistry. 2007; Oct86(1):19–31. doi: 10.1007/s10533-007-9140-0.
64. Sun Z, Scherer L, Tukker A, et al. Dietary change in high-income nations alone can lead to substantial double climate dividend. Nat Food. 2022;3(1):29–37. doi: 10.1038/s43016-021-00431-5.
65. Tilman D, Clark M. Global diets link environmental sustainability and human health. Nature. 2014; 515(7528):518–522. doi: 10.1038/nature13959.
66. Bajželj B, Richards KS, Allwood JM, et al. Importance of food-demand management for climate mitigation. Nature Clim Change. 2014;4(10):924–929. doi: 10.1038/nclimate2353.
67. Xu X, Sharma P, Shu S, et al. Global greenhouse gas emissions from animal-based foods are twice those of plant-based foods. Nat Food. 2021;2(9):724–732. doi: 10.1038/s43016-021-00358-x.
68. Hammar T, Hansson P-A, Röös E. Time-dependent climate impact of beef production – can carbon sequestration in soil offset enteric methane emissions? J Cleaner Prod. 2022;331:129948. doi: 10.1016/j.jclepro.2021.129948.
69. Liang C, MacDonald JD, Desjardins RL, et al. Beef cattle production impacts soil organic carbon storage. Sci Total Environ. 2020; 718:137273. doi: 10.1016/j.scitotenv.2020.137273.
70. Boehm M, Junkins B, Desjardins R, et al. Sink potential of Canadian agricultural soils. Clim Change. 2004; 65(3):297–314. doi: 10.1023/B:CLIM.0000038205.09327.51.
71. Rui Y, Jackson RD, Cotrufo MF, et al. Persistent soil carbon enhanced in mollisols by well-managed grasslands but not annual grain or dairy forage cropping systems. Proc Natl Acad Sci U S A. 2022; Feb 15119(7):1–6.
72. Poeplau C, Don A. Carbon sequestration in agricultural soils via cultivation of cover crops - a meta-analysis. Agric Ecosyst Environ. 2015; 200:33–41. doi: 10.1016/j.agee.2014.10.024.
73. Seitz D, Fischer LM, Dechow R, et al. The potential of cover crops to increase soil organic carbon storage in german croplands. Plant Soil. 2022; doi: 10.1007/s11104-022-05438-w.
74. Rodrigues L, Hardy B, Huyghebeart B, et al. Achievable agricultural soil carbon sequestration

across Europe from country-specific estimates. Glob Chang Biol. 2021;27(24):6363–6380. doi: 10.1111/gcb.15897.

75. Minasny B, Malone BP, McBratney AB, et al. Soil carbon 4 per mille. Geoderma. 2017; 292:59–86. doi: 10.1016/j.geoderma.2017.01.002.
76. Wang J, Xiong Z, Kuzyakov Y. Biochar stability in soil: meta-analysis of decomposition and priming effects. GCB Bioenergy. 2016;8(3):512–523. doi: 10.1111/gcbb.12266.
77. Reisser M, Purves RS, Schmidt MWI, et al. Pyrogenic carbon in soils: a literature-based inventory and a global estimation of its content in soil organic carbon and stocks. Front Earth Sci. 2016;4(80):1–14. doi: 10.3389/feart.2016.00080.
78. Briones MJI, Panzacchi P, Davies CA, et al. Contrasting responses of macro- and meso-fauna to biochar additions in a bioenergy cropping system. Soil Biol Biochem. 2020;145:107803. doi: 10.1016/j.soilbio.2020.107803.
79. Hilber I, Blum F, Leifeld J, et al. Quantitative determination of PAHs in biochar: a prerequisite to ensure its quality and safe application. J Agric Food Chem. 2012;60(12):3042–3050. doi: 10.1021/jf205278v.
80. Rumpel C, Chabbi A, Marschner B. Carbon storage and sequestration in subsoil horizons: knowledge, gaps and potentials; 2012. p. 445–464.
81. Alcantara V, Don A, Vesterdal L, et al. Stability of buried carbon in deep-ploughed Forest and cropland soils – implications for carbon stocks. Sci Rep. 2017; 7(1):5511–5511. doi: 10.1038/s41598-017-05501-y.
82. Woolf D, Lehmann J, Ogle S, et al. Greenhouse gas inventory model for biochar additions to soil. Environ Sci Technol. 2021;55(21):14795–14805. Oct 12. doi: 10.1021/acs.est.1c02425.