Optimierung von Bewirtschaftungsstrategien für Dauergrünland in der gemäßigten Zone mit dem Ziel, die Kohlenstoffspeicherung zu erhöhen, Ertragsanforderungen zu erfüllen und die Widerstandsfähigkeit gegenüber künftigen Klimaänderungen zu gewährleisten

– Auswertungen auf der Grundlage von Langzeitsimulationen –

Dissertation
zur Erlangung des Doktorgrades Dr. sc. agr.
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vorgelegt von

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Optimising management strategies on permanent grassland in the temperate zone in order to increase carbon sequestration, meet yield demands and ensure resilience to future climate change

– Evaluations based on long-term simulations –

Dissertation

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Kurzfassung

Der künftige Klimawandel wird landwirtschaftliche Bewirtschaftungsstrategien erfordern, die sich an die steigenden Lufttemperaturen und die sich ändernden Niederschlagsmuster anpassen und gleichzeitig Nahrungsmittel, Futtermittel und Biomasse bereitstellt sowie eine Vielzahl von Ökosystemleistungen erhalten. Die Speicherung von atmosphärischem Kohlendioxid in die Atmosphäre als organischer Kohlenstoff (Corg) im Boden ist dabei eine wichtige Maßnahme, um nicht nur die Bodenqualität zu verbessern, sondern auch die mit der landwirtschaftlichen Produktion einhergehenden Emissionen durch beispielsweise Methan oder Lachgas auszugleichen und so die globalen und nationalen klimapolitischen Ziele einzuhalten.

In dieser Dissertation wurden Strategien für eine nachhaltige, klimaangepasste Landnutzung für politische Entscheidungsträger und Landwirte erstellt. Dabei wurden die Auswirkungen einer langfristigen Acker- und Grünlandnutzung auf die chemischen Bodeneigenschaften verglichen, welche die Fähigkeit des Bodens bestimmen, Nährstoffe zu speichern und bereitzustellen, Emissionen von Treibhausgasen (THG) und die Auswaschung von Nährstoffen zu reduzieren, sowie seine Fruchtbarkeit zu verbessern. Im Rahmen einer Literaturrecherche wurden mehrere Bewirtschaftungspraktiken zur Verbesserung dieser Eigenschaften identifiziert, wie die Einführung einer artenreichen Dauervegetation, der Anbau von Leguminosen und Zwischenfrüchten, die Einführung von Grünlandphasen in einer Ackerfruchtfolge, die Verringerung der Bodenbearbeitung auf Ackerland, die Bevorzugung der Beweidung gegenüber dem Mähen von Grünland, und die Erhöhung des Eintrags organischer Stoffe, z. B. durch die Rückhaltung von Ernterückständen oder den verstärkten Einsatz organischer Düngemittel.

Um das Potenzial der Corg-Speicherung sowohl auf nationaler als auch auf regionaler Ebene zu prüfen, wurden Simulationen mit dem CANDY-Modell durchgeführt, bei denen insgesamt 18 Szenarien des künftigen Klimawandels bis zum Jahr 2100 sowie unterschiedliche Bewirtschaftungssysteme an mehreren repräsentativen Standorten angewendet wurden. Bei den Simulationen auf nationaler Ebene wurden Regionen mit hoher Priorität identifiziert, die entweder ein hohes Risiko für Corg-Verluste bei suboptimaler Bewirtschaftung oder ein Potenzial für hohe Corg-Gewinne aufweisen, was als Leitfaden für künftige Entscheidungen in der Umweltpolitik dienen kann. Besonders sandige oder schluffige Standorte mit geringen Sommerniederschlägen sind einem hohen Risiko von Corg-Verlusten ausgesetzt. Im Gegensatz dazu werden vor allem Standorte in Mittelgebirgs- bis Alpenregionen vom zukünftigen Klimawandel und den damit zu erwartenden steigenden Lufttemperaturen profitieren, da erwartungsgemäß in solchen Regionen selbst unter den extremsten Klimaszenarien kein Trockenstress

auftreten wird. Die dadurch verlängerte Dauer der Vegetationsperiode wird das Pflanzenwachstum und die Rhizodepisition fördern. Das Corg-Bindungspotenzial ist auf tonreichen Böden, sowohl auf regionaler als auch nationaler Betrachtung, bei intensiver Bewirtschaftung, insbesondere bei Beweidung, am höchsten. Intensives Grünland auf tonreichen Böden birgt den Simulationen zufolge jedoch auch das höchste Risiko von Corg-Verlusten durch Bodenverdichtung und ist damit ein Standort mit hoher Priorität. Ferner kann unter steigender Intensivierung ein Rückgang des Artenreichtums einhergehen, welches wiederum in vielen Regionen eine hohe Priorität genießt und darüber hinaus sowohl zur Verbesserung der Bodenqualität als auch Corg-Speicherung beiträgt. Daher stellen die hier gewonnenen Erkenntnisse einen groben Leitfaden dar, wobei konkrete Empfehlungen der Bewirtschaftungsintensität stets unter Berücksichtigung potenzieller Kompromisse auf standortebene auszusprechen sind.

Abstract

Future climate change will require agricultural management to adapt to increasing air temperatures and changing precipitation patterns while simultaneously providing food, fodder and biomass, and maintaining a variety of ecosystem services. The sequestration of atmospheric carbon dioxide into the atmosphere as soil organic carbon (SOC) is thereby a key measure not only in improving the soil quality but also to counterbalance emissions of e.g. methane or nitrous oxide associated with agricultural production, thereby adhering to global and national climate policy goals.

This thesis provides a guideline for sustainable land use for policy-makers and farmers by comparing the impact of long-term arable and grassland use on the chemical soil properties, which determine the soil's ability to store and provide nutriens, limit emissions of greenhouse gases (GHG) and leaching of nutrients, as well as its overall fertility. As part of a review, several management practices aimed at improving these properties were identified, e.g. introduction of a biodiverse, permanent vegetation, growing legumes and catch crops, implementing grassland phases in an arable rotation, reduction of tillage on arable land, prefering grazing over mowing on grassland, and increasing inputs of organic matter by e.g. retaining harvest residues or increasing inputs of organic fertilisers.

In order to explore the SOC sequestration potential on a national and regional scale, simulations using the CANDY model were performed by applying 18 future climate change scenarios until the year 2100 and contrasting management regimes on several representative sites. The simulations identified regions of high priority due to either high risks for SOC losses under suboptimal management or a potential for high SOC gains, which may guide future decision-making in environmental policies. Particularly sandy or silty sites with low summer precipitation are at high risk of SOC losses. In contrast, particularly sites in low-mountainous to alpine regions will benefit from future climate change and therewith expected increasing air temperatures due to no dryness stress occuring in such regions even under the most extreme climate scenarios. The therereby extended duration of the vegetation period will promote plant growth and rhizodepisition. The SOC sequestration potential on both the national and regional scale is highest on clay-rich soils under intensive, particularly grazing, management. However, intensive grasslands on clay-rich soils also have the highest risk of SOC losses as a result of soil compaction according to the simulations, rendering them sites of high priority. Furthermore, high intensification can be accompanied by a decline in biodiversity, which in turn is a high priority in many regions and also contributes to improving both soil quality and SOC storage. The findings of this thesis therefore represent a rough guideline, whereby specific recommendations for management intensity must always be made taking into account potential trade offs at site level.

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Abbreviations

BAT: Biologic active time

BD: Bulk density

C: Carbon

CANDY: Carbon and Nitrogen Dynamics (model)

CO_2: Carbon dioxide

GHG: Green house gas

K: Potassium

LCA: Life-cycle assessment

LU: Livestock units

LUI: Land use intensity

OM: Organic matter

SOC: Soil organic carbon

SOM: Soil organic matter

N: Nitrogen

NO_3: Nitrate

P: Phosphorus

1. Introduction

1.1. Background of the thesis research project

This doctoral thesis was written as a part of the "SattGrün" project at the Julius Kühn Institute of Crop and Soil Science within a working group led by Dr. Katrin Kuka and supervised by Prof Dr. Jörg Michael Greef. The project combined remote sensing data and simulation models with the overarching goal of providing precise and up-to-date information in order to optimise mowing dates, fertilisation and other management measures in grassland sites in Germany. The aim was to prioritise the economic aspects of agriculture, selected ecosystem services and soil functions in the decision-making process in order to ensure the profitability of grassland farms, the resilience of sites in the face of climate change, and the protection of soil and groundwater as the natural basis of life.

1.2. Challenges posed by future climate changes on agriculture

In the next few decades the expected climate change will pose several challenges to agriculture (Chang et al., 2017; Furtak and Wolińska, 2023). Warming climates in the temperate regions, coupled with changing precipitation patterns and more frequent weather extremes (Chang et al., 2017; DWD & EWK, 2020; Fischer et al., 2018; Furtak and Wolińska, 2023; Swemmer et al., 2007), will require land management practices to adapt to never before encountered conditions in oder to ensure food security and provision of a variety of ecosystem services (Anderson et al., 2020; Boix-Fayos and Vente, 2023; Conijn et al., 2018; Lin et al., 2024). Such ecosystem services include, but are not limited to, maintaining biodiversity and soil structure, filtering and retention of water, storage and provision of nutrients, provision of renewable ressources, and sequestration of carbon dioxide (CO_2) from the atmosphere into the soil (Baer and Birgé, 2018; Banerjee and van der Heijden, 2023; Basset et al., 2023; Hartmann and Six, 2023; Hasan et al., 2020; Steinhoff-Knopp et al., 2021; Toor et al., 2021).

However, especially under changing climatic conditions, unsustainable land management may lead to degrading soil quality, such as erosion, compaction, greenhouse gas (GHG) emissions, nutrient leaching, and decline of soil organic matter (SOM), thus compromising the ability of agricultural soils to provide the aforementioned ecosystem services (Borrelli et al., 2014; Borrelli et al., 2023; Shaheb et al., 2021; Tesfai et al., 2015; van Beek and Tóth, 2012). Particularly on the long term this decline in soil health and fertility affects crop yields, further exacerbating already present environmental and economic stress on agriculture (Baddeley et al., 2017; Boix-Fayos and Vente, 2023; Nevens and Reheul,

2003). The pressure on agriculture is even expected to grow in the future, considering the still increasing demand for animal products, and the high environmental impact and requirement for land area associated with their production (Boke Olén et al., 2021; Guyomard et al., 2021; Makkar, 2018). Thus, it is crucial to identify management practices and site conditions that foster and promote soil health as well as the provision of ecosystem services and simultaneously meet yield demands, while considering the impact of future climate change (Boix-Fayos and Vente, 2023; Makkar, 2018; Taube et al., 2014).

1.3. The role of soil organic carbon as a means to sustainable and climate-smart agriculture

In the soil, SOC fulfills a variety of functions related to the provision of ecosystem services and soil health in general. It acts as a binding agent within soil aggregates, thus increasing the resistance to erosion and its water holding capacity, reduces succeptibility to compaction, stores nutrients, limits losses of nitrogen (N) via leaching as well as gaseous emissions, regulates the soil pH, and is one of the largest global carbon (C) pools (Abbasi et al., 2007; Abbasi and Rasool, 2005; Jones et al., 2005; Kodešová et al., 2009; Lal, 2008; Linsler et al., 2015; McGechan et al., 2005; Meurer et al., 2024; Wang et al., 2012; Zhang et al., 2013). Consequently, sequestration of SOC via land use practices is a key measure to both promoting soil fertility and counterbalancing GHG emissions associated with agriculture and thus contributing to national and global climate protection efforts (Albritton et al., 2001; Arneth et al., 2010; IPCC, 2015; Lal, 2023; Nazir et al., 2024). Identifying farming practices that meet the yield demands and also sequester atmospheric C while minimising the impact of agricultural land use on terrestrial habitats, particularly considering the increasing competition for agricultural land area, is thus of paramount importance (Guyomard et al., 2021; Harvey and Pilgrim, 2011; IPCC, 2006, 2015; Jeanneret et al., 2021; Taube et al., 2014; van de Ven et al., 2021).

1.4. The impact of management, climate and soil properties on soil carbon dynamics

Both arable and grassland sites have a certain SOC sequestration potential, which depends on the exact management practices (Conant et al., 2017; DeGryze et al., 2004; Lugato et al., 2014; Poeplau, 2021; Post and Kwon, 2000; Ramesh et al., 2019), climatic conditions (Bai et al., 2023; Lugato and Berti, 2008; Puissant et al., 2017) and soil properties (Don et al., 2009; Rodríguez-Albarracín et al., 2023; Scartazza et al., 2023). In terms of management and land cover, this potential is highest under a perennial, undisturbed vegetation with regular C inputs via residues and manure (DuPont et al., 2014; Maltas et

al., 2018; Poeplau, 2021; Six et al., 1998; Wiesmeier et al., 2013). Particularly repeated vegetation regrowth and therewith associated increased rhizodeposition is beneficial to SOC sequestration, given sufficient nutrient and water availability (Adewopo et al., 2014; Nüsse et al., 2018; Poeplau, 2021). The long-term stabilisation of SOC, however, depends on climatic conditions and the soil microclimate, i.e. soil moisture, aeration and temperature, as they determine the activity of soil biota and thus the biochemical decomposition of organic matter (OM) and release of CO_2, which is then lost from the soil (Chen et al., 2018; Fu et al., 2023; Tang et al., 2019). Physical soil properties, mainly the texture and bulk density (BD), thereby determine the water holding capacity and aeration, and thus impact decomposition rates and SOC losses (Moyano et al., 2011; Pallandt et al., 2022; Tang et al., 2017). Furthermore, since SOC is mainly stored in mineral-associated aggregates, the available specific surface area of mineral particles limits the maximum SOC storage potential (Don et al., 2009; Rodríguez-Albarracín et al., 2023; Scartazza et al., 2023).

1.5. Challenges and solutions in predicting changes in soil organic carbon

The complex interactions of the three main factors influencing SOC sequestration, namely management, climate and soil properties, render predicting long-term SOC changes a “three body problem” in environmental and soil sciences, particularly when considering spatial and temporal heterogeneity, the magnitude of total C inputs and outputs in relation to absolute changes in SOC contents, and a multitude of sampling issues (Jacobs et al., 2020; Puissant et al., 2017; Schädler et al., 2019; Smith et al., 2020): the change of one site property might heavily change the impact, magnitude and interactions of other site properties. This is moreso relevant considering there’s still a lacking consensus on whether some management practices (e.g. grazing) are more beneficial than others (e.g. mowing and grazing exclusion) to SOC sequestration, despite a large body of available studies on the topic (e.g. Acharya et al., 2012; Eze et al., 2018; Ferreira et al., 2016; Franzluebbers and Stuedemann, 2009; Gao-Lin et al., 2017; Guo et al., 2018; Li et al., 2016; Luo et al., 2015; Ma et al., 2016; Oates and Jackson, 2014; Riedo et al., 2000; Rogiers et al., 2005; Schonbach et al., 2012; Shi et al., 2013; Xiong et al., 2016). As a result, field experiments, while crucial for identifying essential drivers for sequestration, offer only limited insight into long-term SOC dynamics (Jones and Donnelly, 2004; Schädler et al., 2019; Smith et al., 2020). Furthermore, changes in SOC contents due to recent management change typically become apparent after several years to decades and former land use may impact soil properties for as long as decades to centuries (Conant et al., 2001; DuPont et al., 2010; DuPont et al., 2014; Forey and

Dutoit, 2012; Jones and Donnelly, 2004; McLauchlan, 2006). These issues severly limit the robustness of management recommendations based exclusively on field data.

Process-oriented simulation models offer a solution to these issues, allowing to predict SOC changes under a variety of management options, climate scenarios and site properties (Bai and Cotrufo, 2022; Chang et al., 2017; Hinge et al., 2018; Hinge et al., 2021; Jones and Donnelly, 2004; Moss et al., 2010; Witing et al., 2019). This in turn allows to formulate reliable management recommendations that are in line with current climate policy frameworks or identify trade-offs associated with certain sequestration goals (Chang et al., 2017; Gosling et al., 2017; IPCC, 2006, 2015; Taube et al., 2014). However, while there is a considerable body of studies presenting results of SOC simulations based on climate data, soil properties and management, the majority of available studies do not sufficiently consider contrasting management options, sites or future climate scenarios, or are simply limited by a short simulation duration (e.g. Abdalla et al., 2016; Anindita et al., 2023; Carauta et al., 2021; Chang et al., 2017; Congreves et al., 2016; Falloon et al., 2006; Hinge et al., 2021; Liu et al., 2017; Lugato and Berti, 2008; Riedo et al., 2000; Valkama et al., 2020). Consequently, there is still a substantial lack in robust predictions of long-term SOC changes and therefrom arising management recommendations.

1.6. Thesis structure and research objectives

This thesis aims at providing management recommendations for a sustainable and profitable use of agricultural land while considering ecosystem services provided by the respective land area in order to secure the natural basis for life for the coming decades, with a particular focus on SOC sequestration.

As such, in order to provide management recommendations to maintain soil health, productivity and sustainability to farmers and decision makers, chapter 2 reviewed the long-term impact of arable and grassland use on chemical soil properties, such as contents of SOC, N, phosphorus (P), and other essential nutrients, as well as the soil's C to N ratio, pH level, electrical conductivity and cation exchange capacity. These properties primarily relate to the soil's ability to store and provide nutrients as well as limit nutrient leaching and GHG emissions. The study particularly examined the SOC sequestration potential, the risk and pathways of N losses, as well as key processes that are responsible for major gains and losses of C and N under arable and grassland use respctively. Consequently, the study identified strategies that on the one hand allow for reducing the required fertiliser application rate, thus saving time and money, without compromising fertility or yields, and on the other hand have the highest potential for SOC sequestration, thus contributing to climate protection efforts via reduction and storage of GHGs.

Chapters 3 and 4 in turn were dedicated to the simulation-based assessment of the SOC sequestration potential under contrasting management options under varrying climate scenarios on multiple sites. Both chapters thereby focused on temperate grassland. While chapter 3 followed a broad and generalised approach on a national scale, chapter 4 investigated the changes on a regional scale in a representative grassland region.

A central goal of chapter 3 was identifying representative grassland regions across Germany which have the highest potential for long-term SOC sequestration and the highest risk of SOC losses. Furthermore, the study investigated the magnitude of the effect of soil properties, management or climate on the long-term sequestration potential and which climate metrics, i.e. annual means, long-term changes or seasonality, are most relevant on a national scale. For this, the CANDY model (Carbon and Nitrogen Dynamics; Franko et al., 1995; Franko et al., 1997) was applied. The model was parametrised using data from field experiments and then validated with data form various long-term soil survey sites under representative, typical management across Germany. From all available management data, a total of ten grassland management regimes with increasing intensity, five of which integrated grazing periods in addition to annual mowing, were developed and validated using remote sensing data. Following this, 24 grassland sites across Germany, representing different pedoclimatic regions, were selected. On these 24 sites, each of the developed ten management regimes were simulated under a total of 18 climate datasets in a baseline (1990 – 2019) and a future (2020 – 2099) period. The study succesfully identified preferential grassland regions with high SOC sequestration potential, as well as management practices fostering SOC sequestration on the long-term. However, the large scale of the approach left several open questions regarding the magnitude and impact of meteorological metrics on the regional scale. Furthermore, due to the representative nature and national coverage of the sites, the impact of BD and initial SOC content on sequestration potential could not be explored.

Chapter 4 tackled these questions by applying the CANDY model to a dataset consisting of 50 grassland sites in a typical mid-altitude grassland region in the Hainich Biodiversity exploratory in Thuringia, in central Germany. The model was validated using recent management and ambient climate data (2008 – 2017) as well as soil properties. Following this, the reported management in the Hainich region was used to derive six representative management regimes, each three for mown and grazed grassland along an intensification gradient, which were again applied to a total of 18 future climate datasets (2018 – 2100) to increase the robustness of the predictions as well as to identify central interactions between management, soil properties, and climate. The high data density in the Hainch region allowed not only to identify optimal grassland management strategies for a typical, cool, mid-altitude grassland

region, but also to quantify the impact of grazing and mowing among a gradient of different soil properties, precipitation amounts and mean air temperatures. Furthermore, the dataset allowed to quantify the effect of increasing BD and particularly initial SOC content, and thus the remaining available specific particle surface for SOC storage, on the overall sequestration potential.

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2. Comparison of chemical soil properties of temperate grassland and arable land – a review

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Abstract

Chemical soil properties contribute to the resilience of the soil ecosystem. Healthy soils with optimal nutrient levels, balanced pH and good organic matter contents are better able to withstand environmental stresses, such as drought, disease or pest pressure. When comparing the chemical soil properties of temperate grassland and arable land, several differences can be observed due to differences in soil cover and management practices. Grasslands typically sequester more carbon, limit nitrogen leaching and have lower nitrous oxide emissions due to less soil disturbance and a more closed nutrient cycle. In contrast, arable land has higher nutrient losses through harvest, leaching, gaseous emissions and erosion due to regular tillage, frequent bare phases and sequesters less carbon, typically due to higher mineralisation rates and lower nutrient returns. Monitoring and managing chemical soil properties through regular soil testing, appropriate nutrient management, addition of organic matter such as organic fertilisers, inclusion of grassland phases and catch crops in crop rotations, incorporation of crop residues into the topsoil after harvest and further sustainable agricultural practices are essential to promote soil health. By optimising chemical soil properties, farmers and land managers can improve productivity, conserve natural resources and support the long-term sustainability of the soil ecosystem.

Keywords:

chemical soil properties, temperate grassland, arable land, soil health, carbon sequestration, sustainable management

2.1. Introduction

In addition to biological and physical properties, good chemical conditions are essential for healthy soils (Raghavendra et al. 2020; Toor et al. 2021). The chemical properties and ecosystem services that can be provided by the soil depend on site conditions, such as climate and parent material, and particularly how it is managed to produce food, fodder or renewable biomass (Baer and Birgé 2018; Hasan et al. 2020; Rillig et al. 2023). Ecosystem services that are related to the chemical status of the soil are nutrient cycling, carbon (C) sequestration and thus, climate regulation (Baer and Birgé 2018; Hasan et al. 2020; Toor et al. 2021). Conserving and managing soil ecosystems in a sustainable way is crucial to maintain these services and ensure long-term environmental sustainability and human well-being (Hasan et al. 2020; Mayel et al. 2021; Toor et al. 2021; Banerjee and van der Heijden 2023). If the soil is not managed sustainably, there is a risk of e.g. increased greenhouse gas (GHG) emissions, nutrient leaching, heavy metal contamination, decline of organic matter (OM), acidification or salinisation (Wuana et al. 2011; van Beek and Tóth 2012; Tesfai et al. 2015; Yadav et al. 2020; Singh

2021). The risk of this varies depending on the plant cover and management practices (van Vooren et al. 2017; Paudel et al. 2021).

In general, grasslands provide more non-productive ecosystem services than arable land; farmers do not directly benefit from these services, although they are essential for the maintaining of soil quality, biodiversity and regulating nutrient and water cycles (Taube et al. 2014; Gosling et al. 2017; Baer and Birgé 2018; Reinsch et al. 2018a; Bengtsson et al. 2019). Extensive grasslands provide more regulation than production ecosystem services, although intensive grasslands with a higher focus on production ecosystem services are necessary to ensure high fodder quality (Costa et al. 2013; Kauppinen et al. 2013; Jaeger et al. 2019; Zhao Y et al. 2020; Paudel et al. 2021; Hoekstra et al. 2023). Arable land, in turn, fulfils the primary role of producing food and energy crops, although several practices also aim to improve the provision of ecosystem services (Takatsuka et al. 2005; Sandhu et al. 2010; van Vooren et al. 2017; Fischer et al. 2018; Vidaller and Dutoit 2022). However, especially high-productivity systems on both arable and grassland under suboptimal management (e.g. frequent tillage, heavy field traffic, overstocking, residue removal) can have negative environmental impacts such as loss of soil organic carbon (SOC), GHG emissions or nitrogen (N) leaching (Goossens et al. 2001; Freibauer et al. 2004; Skiba et al. 2009; Power 2010; Papini et al. 2011; Kunrath et al. 2015; Buchen et al. 2017; Chen et al. 2017; van Vooren et al. 2017; van Hal et al. 2019; Wang et al. 2020). The demand for animal products is expected to grow by 60 to 70 % by 2050 (Makkar 2018) and considering the GHG emissions from arable soils and livestock, current C accumulation rates are insufficient to accomplish sustainable and climate-neutral agriculture (Post and Kwon 2000; Janssens et al. 2003; Schulze et al. 2009; Soussana et al. 2010; Boke Olén et al. 2021; Launay et al. 2021). Reducing the GHG emissions and nutrient losses, and increasing C sequestration, which depends on appropriate management practices, are therefore key objectives of current agri-environmental policies (IPCC 2006, 2015; Brilli et al. 2017; Wiesmeier et al. 2020).

In order to provide a basis for policy makers and farmers, this literature review compares chemical soil properties of mineral soils under temperate grassland and arable land to identify conditions favorable for C sequestration, mitigation of N leaching and GHG emissions, and maintenance of soil fertility and resilience to climate change. The primary area of interest are temperate grasslands and arable land in Europe, although studies from more distant regions and climates will be considered as supporting information where relevant. In order to compare the effects of land use on soil properties, this review will mainly focus on the topsoil (upper 0 – 30 cm). This layer has the highest plant root density, soil organic matter (SOM) and nutrient contents, and is the main zone of management effects (Franzluebbers et al. 2000; Conant et al. 2001; Poeplau and Don 2013; Gregory et al. 2016; Walia et al.

2017). However, while the topsoil is most vulnerable to change, there is a considerable C storage below the topsoil (Chapman et al. 2013; Skadell et al. 2023). C accumulation in the subsoil is more sustainable than in the topsoil, because it is more stabilised and has a slower turnover, making the subsoil the more appropriate long-term C sink (Poeplau and Don 2013; Skadell et al. 2023), which may also apply to soil N (MacDonald et al. 2011). Therefore, the processes that transport nutrients and water from the topsoil to the subsoil will also be investigated in this review.

For each section, this review provides a tabular summary of the results presented in the cited literature. The comparison of a given property between grassland and arable land is simplified into the categories of being significantly larger on grassland, greater on arable land or not significantly different. The tables also include supplementary information about the sampling strategy, number of sites, study region, and sampled depth/horizon. Soils were identified according to the world reference base for soil resources (WRB; IUSS Working Group WRB 2022). Fig. 1 also provides a simplified visualisation of the inputs, outputs and transformation processes of C and N, which are the main nutrient pools in agricultural soils.

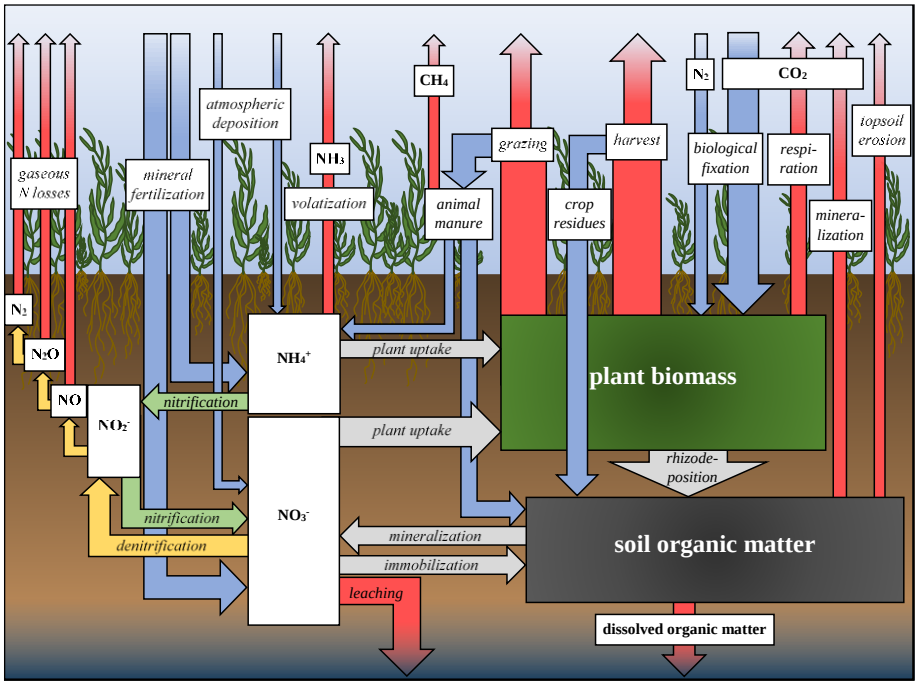

Fig 1: Simplified soil carbon and nitrogen cycle of agricultural soils. Major inputs (blue) and outputs (red) as well as transformation and transportation processes (grey, green and yellow) are denoted in cursive, major nutrient pools and transformation products are bold. Information were gathered from Attard et al. (2016), Buchen et al. (2016), Chapman et al. (2013), Di and Cameron (2002), Haynes and Francis (1990), Horrocks et al. (2016), MacDonald et al. (2011), McLauchlan (2006), Mori and Hojito (2007), Panettieri et al. (2017), Post and Kwon (2000), Rumpel et al. (2015), Schlesinger et al. (2000), Skiba et al. (2009), Soussana et al. (2004), Wang et al. (2015) and Wiesmeier et al. (2012).

2.2. Soil organic carbon

The soil C pool is the third largest of all global C pools and interacts with the oceanic, biotic, geological and atmospheric pools through various transport and transformation processes (Lal 2008). Many important regulating ecosystem services are associated with the SOC content, such as water filtration and storage, maintenance and development of the soil structure, storage and filtration of water, nutrient availability and storage, regulation of pollutants and buffering of the soil pH (Abbasi and Rasool 2005; Jones et al. 2005; McGechan et al. 2005; Abbasi et al. 2007; Kodešová et al. 2009; Li et al. 2018). The SOC content is determined by the geological parent material, soil formation processes, climatic conditions and, in particular, management (Don et al. 2009) which determines the C input/output-ratio and affects the soil physical and chemical properties as well as microbial communities (Post and Kwon 2000; Guo and Gifford 2002; Freibauer et al. 2004; Smith et al. 2005; Chen et al. 2017). This, in turn, leads to changes in the turnover conditions that affect the SOC cycle, resulting in either C emissions or sequestration (Tang et al. 2019). A major part of SOC is located in the topsoil, where it is at risk of being lost due to erosion, decomposition and disturbance (Schlesinger et al. 2000; Skadell et al. 2023). Various processes, such as tillage or bioturbation, can transport SOC from the topsoil to the subsoil, where it is less vulnerable to disturbance and can be stored for the long-term (Post and Kwon 2000; Chapman et al. 2013). A summary of literature comparing SOC-related differences between arable and grassland as an effect of land use can be found in Table 1.

Table 1: Summary of results measuring the effect of land management on carbon dynamics, in particular contents of soil organic carbon (SOC) and soil organic matter (SOM), stable organic matter (OM), and light fraction OM. The comparison of a given property was simplified into categories as being larger on grassland (GL > AL), larger on arable land (GL < AL) or not significantly different (GL = AL).

Property	Relation	Number of sites	Sampling strategy	Sampling time	Depth [cm]	Soil type / texture class	Study region	Reference
SOC / SOM	GL = AL	2	paired sites, repeated sampling	June 2006 – June 2007	0 – 100	Cumulic Phaeozem	Ottawa County, Kansas, USA	(DuPont et al. 2010)
		6	chronose-quence	2008 – 2009	0 – 30	Silt, clay, clay loam, silty loam	Welles-bourne, England	(Gosling et al. 2017) 1

		8	paired sites, chronose-quence	March and July 2010 and 2011	0 – 10	Silty loam, Brown Earth	South Eastern Scotland	(Horrocks et al. 2016)
		12	paired sites	n. a.	0 – 30	Gleyic/ Haplic Leptosols, Haplic/ Stagnic Cambisols	Outer Western Carpa-thians, Poland	(Jozefowska et al. 2016)
	GL > AL	2	paired sites	July 2004 – May 2005	0 – 30	n. a.	Northeast Pakistan	(Abbasi et al. 2007)
		4	paired sites	n. a.	0 – 5	n. a.	Central Japan	(Arai et al. 2018)
		4	paired sites, repeated sampling	2005 – 2008 (36 months)	0 – 10	Cambisol	Lusignan, France	(Attard et al. 2016)
		1	repeated sampling	1991 – 2012	0 – 15	Dystric Cambisol	Aberdeen, NE Scotland	(Baddeley et al. 2017)
		1	repeated sampling	n. a.	0 – 20	Clay loam	Indagai Mountain Pass, Cankiri, Turkey	(Başaran et al. 2008)
		179	paired sites, repeated sampling	1978 – 1988, 2007 – 2009	0 – 100	Alluvial soils, Brown Earths, Gleys, Peats, Podsols and Rankers	Scotland	(Chapman et al. 2013)
		4	paired sites	May 2004	0 – 50	Stagnic Vertisol, Arenosol	Thuringia, Germany	(Don et al. 2009)

		2	paired sites	May & June 2008	0 – 20	n. a.	North Central Kansas, USA	(DuPont et al. 2014)
		2	chronose-quence	May 1997	0 – 20	Sandy loam, loam, clay loam	Watkins-ville, Georgia, USA	(Franzluebbers et al. 2000)
		18	paired sites	2008	0 – 30	Fluvisol	Kolubara Valley, Western Serbia	(Gajic 2013)
		n. a.	repeated sampling	1955 – 2005	0 – 30	Varying	Southern Belgium	(Goidts and van Wesemael 2007)
		74 studies	meta-analysis (paired sites, chronose-quence, repeated sampling)	n. a.	n. a.	n. a.	16 countries, focusing on Australia, Brazil, New Zealand, USA	(Guo and Gifford 2002)
		2	paired sites, repeated sampling	October 2008	0 – 75	Chromic Luvisol (silty clay loam)	Rotham-stedt, Harpen-den, UK	(Gregory et al. 2016)
		n. a.	review	n. a.	n. a.	n. a.	New Zealand	(Haynes and Francis 1990)
		3	repeated sampling	1987 – 2006	0 – 40	Orthic Luvisol (sandy loam)	Tänikon, Switzer-land	(Hermle et al. 2008)

		3	paired sites, repeated sampling	2014 – 2016	0 – 30	Haplic Luvisol	Uhřice, Czech Republic	(Horakova et al. 2017)
		n. a.	meta-analysis	n. a.	n. a.	n. a.	Europe	(Janssens et al. 2003)
		273 (65 studies)	meta-analysis (paired sites)	n. a.	0 – 20	Terrestrial mineral soils of the temperate zone	Northern Hemi-sphere (Central Europe, North America, Russia)	(Kämpf et al. 2016)
		836 (235 studies)	meta-analysis (paired sites)	n. a.	0 – 100+	n. a.	Mostly south and north tem-perate zone	(Li et al. 2018)
		6	paired sites, repeated sampling	2010 – 2017	0 – 30	Eutric Luvisol/Cam bisol	Northern Germany	(Loges et al. 2018)
		12	paired sites, repeated sampling	1949 – 2009	0 – 20	Luvisol	Zurich, Switzer-land	(Oberholzer et al. 2014)
		6	paired sites	n. a.	0 – 30	Vertic Cambisol	Calabria, Italy	(Papini et al. 2011)
		24	paired sites, chronose-quence	n. a.	0 – 80	Varying	Europe	(Poeplau and Don 2013)
		322 (95 studies)	meta-analysis	n. a.	0 – 30 (±6)	n. a.	Tem-perate	(Poeplau et al. 2011)

			(mostly paired sites)				zone, World-wide	
		2	paired sites, repeated sampling	1999 – 2010	0 – 20	Eutric Cambisol	Czech Moravian Upland	(Pospisilova et al. 2011)
		3	paired sites	November 1995	0 – 20	Fine-silty loam	Sydney, Australia	(Six et al. 1998)
		313	chronose-quence	1995 – 2001	0 – 10	n. a.	New Zealand	(Sparling and Schipper 2004)
		11	chronose-quence	before 1995, 1995 – 2000	top horizon	Sandy (predomi-nant), silty	North-west Germany	(Springob et al. 2001)2
		398 (81 studies)	meta-analysis	n. a.	0 – 100+	n. a.	World-wide	(Tang et al. 2019)
		4	paired sites	October 2002 – October 2004	0 – 10	Sandy loam	Melle, Belgium	(van Eekeren et al. 2008)
		717	chronose-quence	2000 and 2004	top horizon	Cambisols (mostly), Leptosols, Regosols, Stagnosols, Albeluvisols, Planosols, Gleysols	Bavaria, Germany	(Wiesmeier et al. 2012)
		717	chronose-quence	Collected after 1990, main part between 2000 and 2004	0 – 100+	Cambisols (mostly), Leptosols, Regosols, Stagnosols, Albeluvisols, Planosols, Gleysols	Bavaria, Germany	(Wiesmeier et al. 2013)

		6	chronose-quence	July 2012	0 – 10	Calcic-Orthic Aridisols (sandy loam)	Inner Mongolia, Northern China	(Zhang et al. 2013)
Stable OM	GL > AL	4	paired sites	March 2015	6 – 15 cm	Chromic Luvisol (silty clay loam)	Rotham-stedt, Harpen-den, UK	(Jensen et al. 2020)
		8	paired sites	2012	0 – 30	Cambisols and Fluvisols	Klamputė & Dembava, Central Lithuania	(Liaudanskiene et al. 2013)
		6	paired sites	n. a.	0 – 30	Sandy soils	Hoshiar-pur, North-West India	(Saha et al. 2011)
		5	paired sites	May & October 2012	0 – 30	Calcic Chernozem	Novi Sad, Serbia	(Seremesic et al. 2020)
Light fraction OM	GL > AL	154 (28 Studies)	meta-analysis	n. a.	n. a.	n. a.	n. a.	(Gosling et al. 2013)
		5	paired sites	May & October 2012	0 – 30	Calcic Chernozem	Novi Sad, Serbia	(Seremesic et al. 2020)

[1] Experiment only focused on short- and medium-term effects (2 year-long experiment).

[2] For AL in the Ap horizon, which was close to 30 cm. For GL in the A horizon, which was on average 28.5 cm.

Processes mediating changes in carbon stocks

Understanding the processes that cause changes in the SOC turnover allows for a long-term implementation of management practices that foster SOC sequestration. The key factors influencing the long-term storage of C in the terrestrial pool are: 1) the available specific surface of mineral particles, 2) input and output rates of C, 3) biochemical decomposability of SOC, especially labile

fractions (e.g. light-fraction or particulate fraction) of organic C and 4) physical protection of aggregates (Post and Kwon 2000).

Specific surface of mineral particles

The available specific surface of mineral particles in soils is a limiting factor for SOC storage (Don et al. 2009; Rodríguez-Albarracín et al. 2023) due to a significant and positive correlation between SOC, the silt and clay content (Tang et al. 2019). Thus, on sites with less mineral surface, such as sandy soils, the differences in SOC accumulation caused by management may be smaller than on sites with a a higher available specific mineral surface (Cai et al. 2016). Accordingly, John et al. (2005) presented evidence indicating that 86 to 91 % of SOC was associated with the mineral fraction of silt and clay in both grassland and arable soils, and that approximately 85 % of the differences in SOC concentration between arable and grassland are explained by changes in mineral associated SOC pools.

Input and output rates of SOC

Given sufficient available mineral surface, a positive SOC balance is achieved when C inputs from belowground biomass production, retention of litter (including harvest residues), and organic fertilisers application exceed losses from harvest, emissions, leaching or erosion. Perennial grassland vegetation is characterised by a dense root network, continuous belowground biomass production and higher root C contents than arable land; whereas arable land often has a sparser root system due to its annual crop rotation and regular tillage, resulting in reduced belowground SOC allocation (Haynes and Francis 1990; Rees et al. 2005; DuPont et al. 2010; Wiesmeier et al. 2013; DuPont et al. 2014; Attard et al. 2016; Gregory et al. 2016; Li et al. 2018; Pausch and Kuzyakov 2018; Yang et al. 2023). Inputs of SOC from root turnover are partially composed of persistent compounds, making them a sustainable and long-lasting source of C inputs (Soussana et al. 2004; Yang et al. 2023). As an effect of vegetation regrowth and increased productivity, increasing the mowing frequency on grasslands additionally promotes SOC sequestration due to repeated root regrowth and belowground C inputs (Adewopo et al. 2014; Nüsse et al. 2018; Poeplau 2021). Arable land is at higher risk of C losses, especially if all crop residues are removed (Chapman et al. 2013; Wiesmeier et al. 2013; Liao et al. 2015; Panettieri et al. 2017). Grasslands also typically receive higher application rates of C-rich organic fertilisers, while arable land typically receives C-deficient mineral fertilisers (McLauchlan 2006; Maltas et al. 2018; Jacobs et al. 2020). While SOC sequestration benefits from inputs of N by increased plant growth and can be limited if N is deficient (Dilly et al. 2003; McGechan et al. 2005; Rees et al. 2005; McLauchlan 2006; Batlle-Aguilar et al. 2011; Poeplau et al. 2018), on European soils N inputs are often higher than the requirements (Freibauer et al. 2004). The substitution of mineral fertilisers with organic fertilisers is therefore a viable method to meet the N demand and increase the C inputs on

arable land (Smith et al. 2005; McLauchlan 2006; Bai et al. 2023) and at least compensate the high C output rates (Taube and Herrmann 2009; Steinmann et al. 2016). While the losses in the form of CO_2 emissions are higher on grassland due to higher microbial activity, the higher inputs and returns result in an overall positive net SOC balance (Wang et al. 2015; Attard et al. 2016; Jozefowska et al. 2016).

Biochemical decomposability of SOC

The chemical decomposition of SOC is the result of biological metabolism by the activity of soil biota, which is controlled by aeration, water availability and temperature (Chen et al. 2018; Tang et al. 2019). The most important factor directly affecting SOC decomposition and sequestration is soil moisture (Wiesmeier et al. 2013; Li et al. 2018; Fu et al. 2023). The texture of a soil thereby determines the availability and retention of water and how changing moisture contents affect decomposition rates (Moyano et al. 2011; Tang et al. 2017; Pallandt et al. 2022). On coarse soils, the turnover and SOC decomposition increase with increasing soil moisture due to a good drainage potential and aeration, while fine soils become anaerobic above a certain soil moisture content, which then reduces the turnover and results in SOC accumulation (Bormann and Klaassen 2008; Tang et al. 2017; Singh et al. 2019; Patel et al. 2021; Pallandt et al. 2022). Differences in texture between grassland and arable sites may therefore be partially responsible for different potentials for biological decomposability of SOC (Fu et al. 2023). Regardless, the transformation of fresh, labile OM fractions into more stable fractions is particularly fostered under undisturbed conditions (Liaudanskiene et al. 2013). Typically regular tillage activities under arable land increase aeration and mineralisation, and reduce the stabilisation of labile SOC in minerals (Li et al. 2018; Jensen et al. 2020; Seremesic et al. 2020). A higher proportion of stable SOC is thus usually found in grassland soils or arable soils under no or conservation tillage (Saha et al. 2011; Liaudanskiene et al. 2013; Jensen et al. 2020; Seremesic et al. 2020).

Stabilisation in soil aggregates

Since the majority of SOC is stored in macroaggregates, the aggregate stability is directly related to a site's potential to sequester SOC (Post and Kwon 2000; Linsler et al. 2015; Scartazza et al. 2023). Soil aggregates and SOC form a reciprocal relationship, since on the one hand SOC stabilises aggregates due to its binding properties (Abbasi and Rasool 2005; Abbasi et al. 2007; Linsler et al. 2015), while on the other hand aggregates protect therein stored SOC from decomposition (Six et al. 1998; Post and Kwon 2000; John et al. 2005; Yamashita et al. 2006). Tillage has been shown to be a major detrimental factor to SOC by eroding the SOC-rich topsoil and causing aggregate-breakdown (Six et al. 1998; Abbasi and Rasool 2005; John et al. 2005; Wiesmeier et al. 2013). As a consequence, the intra-aggregate surfaces containing otherwise physically protected SOC are exposed to decomposers (Post and Kwon 2000; John et al. 2005; Wiesmeier et al. 2013). In contrast, SOC sequestration in undisturbed soils is

enhanced by the formation of intra-aggregate particulate SOC that is incorporated into stable microaggregates and also stabilises macroaggregates (Six et al. 1998; John et al. 2005; Yamashita et al. 2006). On grassland soils a high abundance of perennial roots and fungal hyphae further stabilises aggregates (Low 1972; Haynes and Francis 1990; Kodesova et al. 2011; Spurgeon et al. 2013; Zhang et al. 2013; DuPont et al. 2014). Promoting SOC sequestration in arable soils can therefore be achieved by physically protecting aggregates, e.g. by establishing a permanent vegetation cover, implementing crop-pasture rotations, by planting catch crops, or by the abandonment or reduction of tillage (Haynes and Francis 1990; McLauchlan 2006; Valkama et al. 2020). However, the intra-aggregate protection of SOC may be limited by management practices that result in soil compaction and hinder root growth even under permanent vegetation (Low 1972; Ball et al. 1997), e.g. by the use of heavy machinery, uncontrolled and intensive grazing, and field traffic and harvesting during conditions when the soil is particularly susceptible to compaction (Freibauer et al. 2004; Papini et al. 2011; Chen et al. 2017).

Vertical distribution and incorporation of SOC into the subsoil

The majority of sequestration effects occurs in the topsoil (Schlesinger et al. 2000; Chapman et al. 2013; Poeplau et al. 2018; Skadell et al. 2023). However, the long-term SOC storage is fostered by the incorporation of SOC into deeper layers, where slower turnover processes occur due to lower oxygen concentrations and fewer disturbances, thus providing stable conditions for long-term storage (Don et al. 2009; Poeplau and Don 2013; Wiesmeier et al. 2013; Tang et al. 2017; Pallandt et al. 2022). Due to fundamental differences in management, land cover and therefrom resulting translocation and mixing processes, arable and grassland exhibit different vertical SOC distributions across the soil profile and several studies have demonstrated the need to consider SOC changes of subsoil horizons (Conant et al. 2001; Don et al. 2009; Leifeld et al. 2011; Papini et al. 2011; Wiesmeier et al. 2013; Gosling et al. 2017; Skadell et al. 2023). The key processes affecting the vertical translocation and distribution of SOC across the profile are 1) tillage and 2) bioturbation, while translocation and losses as dissolved organic C are generally considered as minor in both arable and grassland systems (Chantigny 2003; Don et al. 2009; Jozefowska et al. 2016).

Tillage

Tilling of arable land primarily has a homogenising effect on SOC contents in the tillage layer (typically the top 30 cm), whereas grasslands an non-tilled arable land show a gradual decline in SOC with increasing depth, with approximately one third of the SOC stored in the top 10 cm (Don et al. 2009; Papini et al. 2011; Attard et al. 2016; Gregory et al. 2016; Panettieri et al. 2017). Accordingly, the conversion of grassland to tilled arable land typically results in SOC changes in the tillage layer, while

lower layers are usually unaffected (Hermle et al. 2008; Tang et al. 2019). A less common management practice that strongly affects the vertical distribution and can promote SOC incorporation into deeper layers is full inversion or deep tillage. With the mechanical inversion of the profile the SOC-rich topsoil is buried beneath the usual tillage layer and subsoil substrate with low SOC content is brought to the surface, where sequestration can resume (Alcántara et al. 2016; Schiedung et al. 2019; Feng et al. 2020; Lawrence-Smith et al. 2021; Madigan et al. 2022).

Bioturbation

Bioturbation is the process of mixing of soil layers by burrowing activities of soil biota, especially deep-burrowing earthworms (Jozefowska et al. 2016). The permanent, undisturbed vegetation of grassland provides favourable conditions for soil fauna, thereby promoting the transport of SOC into deeper layers by bioturbation (Low 1972; van Lanen et al. 1992; Conant et al. 2001; van Eekeren et al. 2008; Don et al. 2009; Nieminen et al. 2011; Spurgeon et al. 2013; Jozefowska et al. 2016; Arai et al. 2018). Particularly grazed and extensive grasslands also promote a high plant species diversity (Czerwinski et al. 2018; Salek et al. 2018; Kapas et al. 2020; Pardo et al. 2020), which in turn promotes the abundance and activity of soil biota (Lange et al. 2014; Strecker et al. 2016). In contrast, regular tillage and seedbed preparation on arable land reduce earthworm abundance due to physical disturbance, unless under no-tillage management (van Lanen et al. 1992; Nieminen et al. 2011; Jozefowska et al. 2016; Arai et al. 2018).

Response of land use change on C balance and SOC stocks

Former land use can affect the soil properties up to a century after conversion (McLauchlan 2006; Forey and Dutoit 2012) and the effects of changes in management on SOC are typically not immediately apparent (Jones and Donnelly 2004). This lag is particularly mediated by the nutrient availability, land use intensity and tillage practices (Conant et al. 2001; DuPont et al. 2010; Attard et al. 2016; Horrocks et al. 2016; Gosling et al. 2017). On the one hand, low N availability (Gosling et al. 2017) or too extensive management (Horrocks et al. 2016) are potential causes of slowed SOC sequestration after conversion from arable to grassland, while the absence of tillage during arable use after conversion from grassland may also result in no changes in the short term (DuPont et al. 2010). On the other hand, high sequestration rates on newly established grassland may be a result of the depleted SOC stocks from previous arable cultivation (Conant et al. 2001) and high SOC levels on arable land may also result from residual roots from previous grassland vegetation (DuPont et al. 2014). Thus, early changes in SOC may not be representative of recent management and may be negligible when compared to the long-term use (Yamashita et al. 2006). Instead, currently observed changes in SOC can be attributed to

land use change over the past 20 to 30 years (Janssens et al. 2003). Therefore, in order to understand long-term changes in SOC contents, first the dynamic equilibrium of C, i.e. the time until SOC changes come to a halt after a land use change, needs to be addressed. During the time between land use change until an equilibrium is reached, a soil can act as either a sink or source of C (Post and Kwon 2000; Guo and Gifford 2002; Pospisilova et al. 2011). Then, the absolute effect of land use on SOC can be discussed, including the effect of different management strategies.

Dynamic equilibrium

Equilibrium of SOC is reached when the SOC fraction that is lost is equal to the newly formed SOC (Pospisilova et al. 2011). After a disturbance or land use change, a new equilibrium can be reached in response to new conditions in the soil and management practices (Post and Kwon 2000; Guo and Gifford 2002). The SOC sequestration rates until a new equilibrium is reached are not linear and begin to slow down as they approach saturation (Freibauer et al. 2004; Kämpf et al. 2016). Most of SOC is sequestered during the first 20 (Smith 2004a, 2004b; Kämpf et al. 2016) to 40 years (Conant et al. 2001). However, the exact time required to reach a new equilibrium is highly variable. In general, an equilibrium is reached faster following conversion of grassland to arable land than vice versa, because the establishment of a perennial vegetation, recovery of the rhizosphere and subsequent SOC sequestration are slower processes than disturbance by tillage and decomposition of SOC (Poeplau et al. 2011; Attard et al. 2016; Arai et al. 2018). Though depending on the exact balance of C inputs to losses, usually an equilibrium after conversion of grassland to arable land can be reached in as little as 17 (Poeplau et al. 2011) to 20 years (Tang et al. 2019), while after conversion of arable to grassland, it can be reached as long as after 100 (Freibauer et al. 2004) to even 120 years (Poeplau et al. 2011).

SOC contents at equilibrium

While multiple factors influence the potential for sequestration, such as climate, soil texture and most prominently management practices, under otherwise similar site conditions and conventional management SOC contents typically decrease after conversion of grassland to arable land until they reach equilibrium (Springob et al. 2001). A 60-year long fertilisation experiment on grassland converted to arable land showed an average relative loss of 29 % of SOC in the top 20 cm, regardless of the fertiliser type and application rate, with evidence of ongoing SOC decline (Oberholzer et al. 2014). Losses of SOC in the topsoil of arable land after conversion from grassland have been reported to be as high as relative 38 (Başaran et al. 2008) and 59 % (Guo and Gifford 2002). Conversely, the SOC content increases when arable land is converted to grassland, as demonstrated in several studies (Six et al. 1998; Franzluebbers et al. 2000; Knops and Tilman 2000; Freibauer et al. 2004; Sparling and Schipper 2004; Abbasi and Rasool 2005; Goidts and van Wesemael 2007; Hermle et al. 2008; Papini et

al. 2011; Pospisilova et al. 2011; Wiesmeier et al. 2012; Gajic 2013; Poeplau and Don 2013; Wiesmeier et al. 2013; Zhang et al. 2013; DuPont et al. 2014; Wang et al. 2015; Baddeley et al. 2017; Horakova et al. 2017; Loges et al. 2018). A review of 49 studies reported annual SOC increases of > 3 to 5 % after conversion of cropland to grassland (Conant et al. 2001). According to a meta-analysis by Guo and Gifford (2002), under otherwise similar site conditions the increase in SOC after conversion from arable land to grassland was as high as 19 %. As a result of long-term accumulation, SOC stocks in grassland can be two (Springob et al. 2001; Gregory et al. 2016) to three times higher (van Eekeren et al. 2008) than in arable land.

2.3. Soil nitrogen

Soil N is an essential plant nutrient that directly affects plant productivity (Gosling et al. 2017) and limits plant growth if lacking, even under optimal water availability (Haynes and Francis 1990; Bouwman et al. 2009). The use of mineral and organic N fertilisers to increase yields is thereby a common practice in agriculture. In most agricultural systems, N input is dominated by fertiliser application and returns by animal excreta (Bouwman et al. 2009; Skiba et al. 2009; Luo et al. 2022). The major causes of losses are plant uptake, removal by harvest, grazing, leaching and, to some extent, gaseous emissions (Haynes and Francis 1990; Di and Cameron 2002; Bouwman et al. 2009; Bouwman et al. 2013; Congreves et al. 2016; Scheer et al. 2020). Annual fertiliser application rates should be determined according to the annual plant uptake and residual N contents in the soil. Otherwise, N surpluses pollute water and land ecosystems, affect air and soil quality, climate, as well as biodiversity and human health (Haynes and Francis 1990; Di and Cameron 2002; McLauchlan 2006; Conijn et al. 2018; Scheer et al. 2020). Through N inputs and outputs and its ability to alter turnover conditions in the soil, land use has a profound impact on the ability of soils to store N, provide available N for plants, the N leaching potential, and the gaseous N emissions (Robertson et al. 1994; Di and Cameron 2002; Abbasi et al. 2007; Skiba et al. 2009; Dupas et al. 2015; Horrocks et al. 2016; Buchen et al. 2017). Table 2 provides a summary of the effect of land use on total N, available N, N leaching and gaseous N emissions.

Table 2: Summary of results measuring the effect of land management on nitrogen (N) dynamics, in particular total N, available N, N leaching, and nitrous oxide (N_2) emissions. The comparison of a given property was simplified into categories as being larger on grassland (GL > AL), larger on arable land (GL < AL) or not significantly different (GL = AL).

Property	Relation	Number of sites	Sampling strategy	Sampling time	Depth [cm]	Soil type / texture class	Study region	Reference
Total N	GL = AL	2	paired sites, repeated sampling	June 2006 – June 2007	0 – 100	Cumulic Phaeozem	Ottawa County, Kansas, USA	(DuPont et al. 2010)
		8	paired sites, chronose-quence	March and July 2010 and 2011	0 – 10	Silty loam, Brown Earth	South Eastern Scotland	(Horrocks et al. 2016)
	GL > AL	2	paired sites	July 2004 – May 2005	0 – 30	n. a.	Northeast Pakistan	(Abbasi et al. 2007)
		1	repeated sampling	1991 – 2012	0 – 15	Dystric Cambisol	Aberdeen, NE Scotland	(Baddeley et al. 2017)
		2	paired sites, repeated sampling	October 2008	0 – 75	Chromic Luvisol (silty clay loam)	Rotham-stedt, Harpen-den, UK	(Gregory et al. 2016)
		12	paired sites	n. a.	0 – 30	Gleyic/ Haplic Leptosols, Haplic/ Stagnic Cambisols	Outer Western Carpa-thians, Poland	(Jozefowska et al. 2016)
		22	chronose-quence, repeated sampling	1945 – 1970	n. a.	Varying	East central England	(Low 1972)
		2	paired sites	November 1984 – May 1986	0 – 28	Clay	South-East Queens-	(Robertson et al. 1994)

							land, Australia	
		313	chronose-quence	1995 – 2001	0 – 10	n. a.	New Zealand	(Sparling and Schipper 2004)
		160	paired sites, chronose-quence	March & June 2015	0 – 15	Lithology: Karst dolomite and limestone, non-karst clasolite	South-west China	(Wang et al. 2018)
		717	chronose-quence	2000 and 2004	top horizon	Cambisols (mostly), Leptosols, Regosols, Stagnosols, Albeluvisols, Planosols, Gleysols	Bavaria, Germany	(Wiesmeier et al. 2013) [3]
		6	chronose-quence	July 2012	0 – 10	Calcic-Orthic Aridisols (sandy loam)	Inner Mongolia, Northern China	(Zhang et al. 2013)
Available N	GL = AL	8	paired sites, chronose-quence	March and July 2010 and 2011	0 – 10	Silty loam, Brown Earth	South Eastern Scotland	(Horrocks et al. 2016)
		n.a.	review (chrono-sequen-ce, rep-eated sampling)	n.a.	n.a.	n.a.	World-wide	(McLauchlan 2006)
		398 (81 studies)	meta-analysis	n. a.	0 – 100+	n. a.	World-wide	(Tang et al. 2019)

N leaching	GL < AL	4	paired sites, repeated sampling	April 2014 – May 2015	0 – 10	Histic Gleysol, Plaggic Anthrosol	Lower Saxony, Germany	(Buchen et al. 2017)
		n. a.	review	n. a.	n. a.	Varying	n. a.	(Di and Cameron 2002)
		2	repeated sampling	October 2010 – September 2015	0 – 15	Silt, loam	Cork, Ireland; Brittany, France	(Dupas et al. 2017)
		250	meta-analysis	1991 - 2009	n. a.	Sandy soils	Nether-lands	(Fraters et al. 2015)
		6	paired sites	April 2005 – June 2012	105	Cambisol	Lusignan, France	(Kunrath et al. 2015)
		31	paired sites, repeated sampling	1986 – 2005	n. a.	Mostly Cambisol, Planosol, Gleysol	Cesky Krumlov district, Czech Republic	(Kvítek et al. 2009)
		62	paired sites, repeated sampling	2003 – 2006	0 – 30	Eutric Fluvisol	Jena, Germany	(Leimer et al. 2015)
		82	paired sites, repeated sampling	2003 – 2007	0 – 15	Eutric Fluvisol	Jena, Germany	(Oelmann et al. 2011)
		2	paired sites	November 1984 – May 1986	0 – 28	Clay	South-East Queens-land, Australia	(Robertson et al. 1994)

N_2O emissions	GL < AL	2	paired sites, repeated sampling	June 2002 – September 2004	0 – 20	Alfisol	South-west Michigan, USA	(Grandy and Robertson 2006)
		6	paired sites	2007 – 2008	0 – 10	Silty clay loam	Québec City, Canada	(MacDonald et al. 2011)
		4	paired sites	2010 - 2011	0 – 10	Eutric Luvisol	Kiel, Germany	(Reinsch et al. 2018b)
	GL = AL	6	paired sites, repeated sampling	May – July 2014	0 – 30	Histic Gleysol, Plaggic Anthrosol	Lower Saxony, Germany	(Buchen et al. 2016)
	GL > AL	6	paired sites	April 1997 – February 1998	0 – 15	Loamy sand, sand, silty loam, sandy loam	Belgium	(Goossens et al. 2001)

[3] *N stocks were similar, yet N concentrations were higher under grassland than under arable land.*

Land use related changes in nitrogen contents

Changes in the soil N stocks must be considered in terms of both 1) total soil N, which consists largely of organically bound N, i.e. immobilised and stored in SOM and microbial biomass, and 2) exclusively mineral N (Haynes and Francis 1990; Barrett and Burke 2000; Cookson et al. 2007). Following SOM mineralisation, mineral N forms, primarily nitrate (NO_3^-) and ammonium (NH_4^+), are released into soil solution and are available for plant uptake, although they may also be subject to various mechanisms of losses (Haynes and Francis 1990; Di and Cameron 2002; Horrocks et al. 2016; Buchen et al. 2017).

Total N stocks

Grasslands typically have a higher total N storage capacity and a higher potential to be N sinks than arable lands (Franzluebbers et al. 2000; Skiba et al. 2009) due to their higher content of SOC, which has a significant linear relationship with total N (Barrett and Burke 2000; Abbasi et al. 2007). The effects of land management on total N contents in soils also prevail over site-specific effects, such as climate, topology and lithology (Garten and Ashwood 2002; Sparling and Schipper 2004; Jozefowska et al. 2016; Wang et al. 2018). In a study by Robertson et al. (1994), higher total N contents than in arable land were reported for unfertilised pastures due to higher SOC contents. Correspondingly, Gregory et al. (2016) reported losses of 61 % in total N in the upper 15 cm and of 22 % in the 30 to 60 cm layer 59

years after conversion of grassland to arable land. Consequently, it is possible to increase the total N stocks on the long term by converting arable to grassland or introducing management practices that foster SOC sequestration (Zhang et al. 2013; Baddeley et al. 2017). Particularly the inclusion of legumes into grassland mixtures and catch crops on arable land promotes N accumulation (Knops and Tilman 2000; Eriksen et al. 2008; Oelmann et al. 2011; Horrocks et al. 2016). However, the N accumulation slows down with increasing N contents, as it approaches saturation (Knops and Tilman 2000).

Available N

Compared to total N, available N is more closely related to short-term management effects, especially N fertiliser inputs, and shows pronounced seasonal variations (Di and Cameron 2002; Horrocks et al. 2016). The mineralisation, removal, and risk of losses of available N is higher in tilled soils grown with annual crops, while losses are lower under an undisturbed permanent vegetation due to larger biomass returns and a continuous N uptake and cycling (Attard et al. 2016). Accordingly, the amount of available N did not change significantly after conversion of unfertilised grassland to fertilised cropland in a study by Tang et al. (2019) due to higher losses under arable cultivation and higher N retention under grassland. Vice versa, elevated available N contents as an effect of high fertiliser application during prior arable cultivation decrease slowly after conversion to grassland due to lower losses and higher returns (McLauchlan 2006; Attard et al. 2016; Horrocks et al. 2016). Available N was therefore also not significantly different in four pairs of unfertilised grassland representing different stages in succession (three to nine years) after conversion from fertilised arable land in a study by Horrocks et al. (2016). Thus, a viable method to at least partly counteract the higher losses of available N under arable cultivation may be rotational farming including ley phases: in comparison with a permanent arable variant the alteration of three years of ley use with three years of arable rotations resulted in increased soil fertility, higher yields and reduced N fertiliser requirement over a time span of 31 years (Nevens and Reheul 2003). However, the intervals of rotational farming require enough time for the manifestation of effects related to vegetation cover, as evident by the survey of Attard et al. (2016), where changes in available soil N related to conversion of grassland to arable land and vice versa became apparent after 36 months due to the time needed to establish a new root system.

Nitrogen leaching potential

Leaching of N is the process of mobilisation of soluble N and subsequent seepage into groundwater, where it poses a threat to health and the environment. The main part of the leachable N fraction comprises NO_3^-, while losses as dissolved organic nitrogen (DON) are second in importance (Di and Cameron 2002; Krol et al. 2016). Leaching as NH_4^+ can be considered negligible due to its high affinity

to be adsorbed by the surface of clay minerals and typically low NH_4^+ concentrations due to a rapid conversion to NO_3^- (Di and Cameron 2002; Attard et al. 2016; Krol et al. 2016). The management-related factors that control N leaching are 1) the balance between N input (e.g. from fertilisers or animal excreta) and crop uptake, 2) soil water availability and permeability, 3) N immobilisation (i.e. by microbial biomass or adsorption by SOM) and 4) plant cover and diversity (Rodrigo et al. 1997; Di and Cameron 2002; MacDonald et al. 2011; Oelmann et al. 2011; Dupas et al. 2015).

N input and uptake

The balance between N input and plant uptake determines the N surplus in the soil and thus has a direct impact on the risk of leaching. Compared to arable land, grasslands have a higher and more constant N uptake from the soil due to a longer growing season and perennial vegetation (Moreau et al. 2012; McDowell et al. 2014; Dupas et al. 2017). Several studies therefore conclude a lower N leaching risk under grassland than arable land (Di and Cameron 2002; Segato et al. 2009; Kunrath et al. 2015), even if N fertiliser application rates are higher on grassland (Kunrath et al. 2015). On arable land, the leaching risk is particularly high during the bare periods between harvest and crop emergence due to lacking uptake (Sapkota et al. 2012; Fraters et al. 2015). Reducing this leaching risk on arable land can be achieved by reducing the total amount of fertiliser applied, avoiding fertiliser application during periods with low expected uptake, planting catch crops to maintain a constant N uptake or by an implementation of fodder grass within a crop rotation (Di and Cameron 2002; Kayser et al. 2004; Kunrath et al. 2015; Buchen et al. 2017; Zhao J et al. 2020).

Soil water availability and permeability

Leaching of N depends not only on the N availability, but also on the amount of water available for infiltration and percolation (Cookson et al. 2000; Attard et al. 2010; MacDonald et al. 2011; Simonin et al. 2015; Yan et al. 2015; Smit et al. 2021). Differences in vegetation cover (perennial vs. annual) explain the large differences in leaching losses between cropland and grassland (Fraters et al. 2015; Zhao J et al. 2020), since vegetation cover affects evaporation, thereby minimising the leaching risk as soil moisture decreases (Attard et al. 2016). In the absence of catch crops, the risk of N leaching is expected to be highest on arable land in autumn and winter when the soil moisture is highest, especially during the bare phases after tillage and before crop emergence, and reaches a minimum in summer, when water uptake increases and the soil moisture decreases (Shepherd et al. 2001; Kayser et al. 2004; Cookson et al. 2007; Eriksen et al. 2008; Kahle et al. 2018). In addition, tillage has long-term negative effects on soil structure and temporarily increases infiltration (Six et al. 1998; Bormann and Klaassen 2008; Gajic 2013), which may contribute to the higher leaching potential of arable land if practiced regularly (Robertson et al. 1994; Kvítek et al. 2009; Attard et al. 2016).

N immobilisation

A reduced risk of N leaching can also be achieved by increasing N immobilisation, e.g. by promoting soil microbial activity and high contents of SOC that adsorb soil N (Robertson et al. 1994; Barrett and Burke 2000; Cookson et al. 2007; Luxhøi et al. 2008; Segato et al. 2009; Attard et al. 2016). In grassland soils, the competition for NH_4^+ between microbiota and nitrifying bacteria is higher than in arable soils, so that a greater proportion of N is consumed and temporarily stored in microbial biomass as well as in SOM, whereas arable soils have an increased activity of nitrifying bacteria, which promotes nitrification and NO_3^- availability (Grandy and Robertson 2006; Attard et al. 2016). Therefore, the retention of crop residues on arable land is a viable method to reduce soil NO_3^- concentrations in the post-harvest period through the assimilation of N into SOM (Barrett and Burke 2000; Luxhøi et al. 2008; Köbke et al. 2018).

Plant diversity

Evidence has been presented for a positive correlation between plant diversity and plant available N (Mulder et al. 2002; Oelmann et al. 2011). A high biodiversity of grassland flora effectively reduces the need for fertilisation through a more closed N cycle and constant uptake, thus reducing the risk of N leaching (Leimer et al. 2015). In line with this, a meta-analysis confirmed a significant reduction of N leaching in grassland mixtures compared to monocultures (Wang et al. 2020).

Gaseous nitrogen emissions

Gaseous N emissions occur in the form of diatomic nitrogen (N_2), nitrous oxide (N_2O), nitric oxide (NO) and ammonia (NH_3) (Di and Cameron 2002; Bouwman et al. 2013). Compared to other pathways of N losses (particularly plant uptake and leaching), gaseous emissions account for smaller quantities and therefore play a secondary role in the overall N balance of agricultural soils (Bouwman et al. 2013; Congreves et al. 2016). However, N_2O in particular is a GHG of concern because it has the 265-fold global warming potential of CO_2 (Mori and Hojito 2007; IPCC 2015; Gosling et al. 2017) and accounts for the majority of agricultural GHG emissions (Herr et al. 2019). Key drivers of N_2O emissions are N availability, respiration activity, aeration, and soil moisture (Buchen et al. 2017). Though primarily a product of anaerobic denitrification, N_2O is also a by-product of nitrification under aerobic conditions (Mori and Hojito 2007; Ji et al. 2015; Buchen et al. 2017; Scheer et al. 2020). When there is sufficient available N, the highest N_2O emissions typically occur with elevated soil moisture, usually in spring or after rewetting of the soil (Herr et al. 2019; Luo et al. 2022). Emissions of N_2O are consequently greatly affected by management practices, particularly 1) fertiliser application and 2) tillage (Skiba et al. 2009; Congreves et al. 2016; Fan et al. 2023).

N Fertilisation

The input of readily available N from fertilisers, whether in mineral or organic form, is a direct cause of N_2O emissions in both arable and grassland by exceeding soil denitrification capacity and plant uptake (Mori and Hojito 2007; MacDonald et al. 2011; Buchen et al. 2017; Luo et al. 2022; Fan et al. 2023). Emissions of N typically peak shortly after fertilisation events (Herr et al. 2019). This effect prevails across different climates and pedogenic properties, as shown by a comparison of arable and grassland sites, where N_2O emissions increased with N input: the highest emissions (12.47 ± 18.09 kg N_2O-N ha^{-1}) were detected on an intensive grassland receiving 326 kg N ha^{-1} input, while arable sites with lower N fertiliser inputs had lower N_2O emissions and the lowest emissions (0.16 ± 0.29 kg N_2O-N ha^{-1}) occurred on an unfertilised grassland site (Skiba et al. 2009). Similar results were presented by Goossens et al. (2001), who also found that the highest N_2O emissions occurred on an intensively managed grassland (13.7 kg N_2O-N ha^{-1}, with a N input of 506 kg N ha^{-1}) and on arable land with fodder grass (23.7 kg N_2O-N ha^{-1}, with a N input of 318 kg N ha^{-1}), while emissions were correspondingly lower on arable and grassland sites with lower fertilisation.

Aeration, tillage and land cover

Tillage practices are an indirect cause of increased gas fluxes by breaking down aggregates, exposing previously protected SOM, loosening the soil, and increasing aeration and mineralisation (Haynes and Francis 1990; Grandy and Robertson 2006; Attard et al. 2016). Increased mineralisation of SOM results in higher N_2O fluxes due to elevated microbial availability of NO_3^- for denitrification (Necpálová et al. 2013; Kandel et al. 2018). Conversion of grassland to arable land therefore generally increases GHG emissions due to typically regular tillage under arable cultivation (MacDonald et al. 2011; Kandel et al. 2018). Reported N_2O fluxes were up to 7.7 times higher in the first year after conversion of grassland to arable land, and up to 6.7 times higher two years after conversion (Grandy and Robertson 2006). However, as shown by Reinsch et al. (2018b), the effect of tillage varies with time, as the highest N_2O emissions were reported on grassland tilled and reseeded in autumn (14.08 ±5.72 kg N_2O-N ha^{-1}), followed by grassland converted to a maize field in spring (6.08 ± 0.77 kg N_2O-N ha^{-1}) and grassland tilled and reseeded in spring (3.73 ± 0.33 kg N_2O-N ha^{-1}), while emissions on intact permanent grassland (i.e. no tillage) were the lowest (0.84 ± 0.29 kg N_2O-N ha^{-1}). The increased N_2O emissions were associated with freeze-thaw cycles, which were particularly pronounced on soils tilled in autumn, which in turn promoted water infiltration (Reinsch et al. 2018b). MacDonald et al. (2011) further demonstrate that the differences in N emissions are particularly dependent on land cover and tillage type: the cumulative emissions of two undisturbed control grassland sites, one unfertilised (0.07 g N m^{-2}) and one organically fertilised with 369 kg N ha^{-1} y^{-1} (0.09 g N m^{-2}), were still lower than those of

both a) subplots fertilised with the same amount of N and tilled in spring (2.0 g N m^{-2}) or autumn (1.7 g N m^{-2}), and even b) those of unfertilised subplots tilled in spring (1.2 g N m^{-2}) or autumn (0.9 g N m^{-2}). As an exception, one study showed no significant differences in N_2O emissions 44 days after grassland was converted to arable land, but the authors stress that the time span was insufficient to detect significant changes (Buchen et al. 2016).

2.4. Soil C/N ratio

The C/N ratio is the mass ratio of organic C to N in the soil and reflects the quality of SOM and the mineralisation rate (Knops and Tilman 2000). An increasing C/N ratio indicates that N mineralisation exceeds C mineralisation, which may lead to N limiting conditions and reduced soil fertility (Abbasi et al. 2007). Conversely, a decreasing C/N ratio implies an improvement in SOM quality and soil fertility via N accumulation (Baddeley et al. 2017). No changes occurring in the C/N ratio imply very similar dynamics in C and N mineralisation and immobilisation (Necpálová et al. 2013). Changes in the C/N ratio occur largely due to land management, e.g. changes in the vegetation cover, the plant N uptake and inputs by residues, animal excreta and fertilisers (Franzluebbers et al. 2000; Dilly et al. 2003; Abbasi et al. 2007), and are therefore mainly limited to the topsoil. The C/N ratios of agricultural topsoils typically range between 7:1 to 20:1, with typically narrower ratios under grassland than under arable land under otherwise similar site conditions (Dilly et al. 2003; Sparling and Schipper 2004; Abbasi et al. 2007). Higher C/N ratios in arable land (10:1) than in grassland (7:1) were attributed to a higher N mineralisation, greater N decline as an effect of higher uptake and lower return under arable cultivation than under grassland use (Dilly et al. 2003; Abbasi et al. 2007). Conversely, Baddeley et al. (2017) showed that abandonment and reconversion of arable land to grassland resulted in a narrowing C/N ratio. The narrowing of the C/N ratio is particularly promoted by the application of organic fertilisers with a narrow C/N ratio (Franzluebbers et al. 2000). Table 3 summarises the reports comparing the effects of arable and grassland use on the soil C/N ratio.

Table 3: Summary of results measuring effects of land management on the ratio of carbon to nitrogen (C/N-ratio) in the soil. The comparison was simplified into categories as being larger on grassland (GL > AL), larger on arable land (GL < AL) or not significantly different (GL = AL).

Property	Relation	Number of sites	Sampling strategy	Sampling time	Depth [cm]	Soil type / texture class	Study region	Reference
C/N-ratio	GL < AL	2	paired sites	July 2004 – May 2005	0 – 30	n. a.	Northeast Pakistan	(Abbasi et al. 2007)
		1	repeated sampling	1991 – 2012	0 – 15	Dystric Cambisol	Aberdeen, NE Scotland	(Baddeley et al. 2017)
	GL = AL	2	chronose-quence	May 1997	0 – 20	Sandy loam, loam, clay loam	Watkins-ville, Georgia, USA	(Franzluebbers et al. 2000)

2.5. Soil phosphorus

Phosphorus (P) is an essential, often limiting, macronutrient (Bouwman et al. 2009; Conijn et al. 2018). Its natural supply largely depends on weathering of the geological parent material, although this accounts for relatively small amounts (Smil 2000; McLauchlan 2006). The main P pathways in agricultural systems are 1) uptake by harvest or grazing, 2) inputs by fertilisation and animal excreta and 3) losses by erosion or leaching (Sparling and Schipper 2004; Ekholm et al. 2005; McLauchlan 2006; Bouwman et al. 2013; Dupas et al. 2017; Conijn et al. 2018). Table 4 summarises the effects of land use on soil available and total P contents.

Table 4: Summary of results measuring effects of land management on soil phosphorus (P) contents, in particular available and total P. The comparison of a given property was simplified into categories as being larger on grassland (GL > AL), larger on arable land (GL < AL) or not significantly different (GL = AL).

Property	Relation	Number of sites	Sampling strategy	Sampling time	Depth [cm]	Soil type / texture class	Study region	Reference
Available P	GL < AL	6	chronose-quence	2008 – 2009	0 – 30	Silt, clay, clay loam, silty loam	Welles-bourne, England	(Gosling et al. 2017)

	GL = AL	8	paired sites, chronose-quence	March and July 2010 and 2011	0 – 10	Silty loam, Brown Earth	South Eastern Scotland	(Horrocks et al. 2016)
	GL > AL	2	paired sites	n. a.	0 – 15	n. a.	Northeast Pakistan	(Abbasi and Rasool 2005)
		2	paired sites	July 2004 – May 2005	0 – 30	n. a.	Northeast Pakistan	(Abbasi et al. 2007)
Total P	GL < AL	6	chronose-quence	2008 – 2009	0 – 30	Silt, clay, clay loam, silty loam	Welles-bourne, England	(Gosling et al. 2017)
		160	paired sites, chronose-quence	March & June 2015	0 – 15	Lithology: Karst dolomite and limestone, non-karst clasolite	South-west China	(Wang et al. 2018)
	GL = AL	2	paired sites, repeated sampling	June 2006 – June 2007	0 –100	Cumulic Phaeozem	Ottawa County, Kansas, USA	(DuPont et al. 2010)
		64	paired sites, chronose-quence	March & April 2007	0 – 20	Varying	England	(Hathaway-Jenkins et al. 2011)
		8	paired sites, chronose quence	March and July 2010 and 2011	0 – 10	Silty loam, Brown Earth	South Eastern Scotland	(Horrocks et al. 2016)

Uptake by harvest or grazing

The P uptake in form of grazing and harvest typically exceeds the natural supply in agricultural soils (Smil 2000; McLauchlan 2006; Bouwman et al. 2009; Ohm et al. 2015). However, grasslands usually have a more closed P cycle than arable land, because a smaller proportion of P is removed from grassland soils by harvest and grazing, whereas the plant uptake and removal on arable land is often

higher than the return from crop residues (Haynes and Francis 1990; Abbasi et al. 2007; Bouwman et al. 2009; Ohm et al. 2015).

Fertilisation and animal excreta

Overall, arable lands typically receive higher P fertiliser input rates than grassland (Bouwman et al. 2009). Furthermore, on grasslands the majority of the P requirement is met by organic fertilisers, whereas arable soils are primarily fertilised with mineral P, although those in cattle-rich regions may receive high organic fertiliser inputs as well (McLauchlan 2006; Ohm et al. 2015; Sattari et al. 2016). Because the organic fertiliser inputs are often determined by the N requirements, fertiliser P inputs in agricultural systems may exceed P requirements (Ekholm et al. 2005; Möller 2009), which poses a risk of overfertilisation. Elevated P contents in the topsoils of arable land have been reported to be caused by high fertiliser application rates (Wang et al. 2018), whereas grasslands are typically at lower risk of P over-fertilisation due to a more closed nutrient cycle and better P retention capacity (Haynes and Francis 1990; McLauchlan 2006; Bouwman et al. 2009; Ohm et al. 2015). In particular, farming practices have a major impact on the need for additional P fertilisation (Sattari et al. 2016): on pastures the returns in the form of livestock excreta reduce the need for P fertilisation, whereas arable land and meadows, in the absence of grazing, rely on additional fertilisation due to the lack of returns from animal excreta (Ekholm et al. 2005; Ohm et al. 2015; Ohm et al. 2017; Conijn et al. 2018). Animal excreta play a central role in the nutrient cycling of grassland, since only a small fraction of nutrients is retained in animal products, such as meat, milk and eggs, while the majority of nutrients is excreted (Liu et al. 2017). Substantiating this, similar P contents were found when comparing P-fertilised arable land with unfertilised but grazed grassland (Hathaway-Jenkins et al. 2011). Conversely, declining P contents have been reported under annually mown grassland in the absence of fertiliser or manure inputs, when compared to P-fertilised arable land (Gosling et al. 2017). After cessation of P fertilisation on arable land and conversion to grassland the soil P contents typically decline at a slow rate due to the aforementioned lower requirements for additional P inputs under grassland vegetation (Ekholm et al. 2005; McLauchlan 2006; Gosling et al. 2017; Ohm et al. 2017). Elevated P contents of arable land may therefore not change significantly in the short term after conversion to grassland (Horrocks et al. 2016) and might persist for up to 50 years (Fagan et al. 2008), although legacy effects of previous P fertilisation lasting from centuries to millennia have also been documented in a review (McLauchlan 2006).

Losses by leaching and erosion

When P inputs surpass the requirements, the resulting losses through either leaching or erosion are at risk of causing off-site damages, such as water eutrophication and loss of biodiversity (Smil 2000; Ekholm et al. 2005; Liu et al. 2017). Although P availability was reported to be higher in grassland than in arable land, this did not result in increased P loads in groundwater (Dupas et al. 2017), due to a positive correlation between available P and SOC (Abbasi and Rasool 2005; Abbasi et al. 2007). Nevertheless, P losses by leaching are generally considered to be low due to its overall low solubility (Smil 2000). Instead, major P losses are a result of tillage-induced erosion due to the strong binding of P in soil aggregates (Ekholm et al. 2005; McLauchlan 2006; Dupas et al. 2017). This risk is particularly high on arable land, where tillage and therewith associated topsoil erosion are common (Schlesinger et al. 2000; Freibauer et al. 2004; Abbasi and Rasool 2005; Ekholm et al. 2005). Tillage has been associated with more than 75 % of increased topsoil erosion under arable land, resulting in a significant risk of P loss and off-site damage (Smil 2000).

2.6. Other nutrients

Besides C, N and P, other nutrients, especially potassium (K), and micronutrients are also important for plant growth and soil quality. Grasslands were reported to have higher available K contents than arable land due to a positive correlation of SOC with K (Abbasi and Rasool 2005). These findings were confirmed even after accounting for seasonal variations due to management and weather (Abbasi et al. 2007). However, the application of manure or mineral fertiliser on arable land can offset this effect, resulting in no differences in soil K (Hathaway-Jenkins et al. 2011). For micronutrients, i.e. zinc (Zn), copper (Cu), manganese (Mn) and iron (Fe), while one study found no deficit in either arable or grassland soils, the levels were two to five times higher in grassland due to the higher SOC content and its nutrient binding properties (Abbasi and Rasool 2005). A summary of available studies on soil K and other micro- and macronutrients in arable and grassland is given in Table 5.

Table 5: Summary of results measuring effects of land management on contents of available and total potassium (K), micro- and macronutrients in the soil. The comparison of a given property was simplified into categories as being larger on grassland (GL > AL), larger on arable land (GL < AL) or not significantly different (GL = AL).

Property	Relation	Number of sites	Sampling strategy	Sampling time	Depth [cm]	Soil type / texture class	Study region	Reference
Available K	GL > AL	2	paired sites	n. a.	0 – 15	n. a.	Northeast Pakistan	(Abbasi and Rasool 2005)

		2	paired sites	July 2004 – May 2005	0 – 30	n. a.	Northeast Pakistan	(Abbasi et al. 2007)
	GL = AL	6	paired sites	n. a.	0 – 30	Vertic Cambisol	Calabria, Italy	(Papini et al. 2011)
Total K	GL = AL	64	paired sites, chronose-quence	March & April 2007	0 – 20	Varying	England	(Hathaway-Jenkins et al. 2011)
Micro- and macronutri ents	GL > AL	2	paired sites	n. a.	0 – 15	n. a.	Northeast Pakistan	(Abbasi and Rasool 2005)

2.7. pH value

The soil pH value indicates the acidity of a soil solution, which directly affects the cation exchange capacity of a soil (Başaran et al. 2008) and thus the mobility of nutrients and toxic elements (Cao et al. 2001). Due to differences in vegetation composition, optimal pH values for arable and grassland differ, with target values being about 0.5 pH units lower in grassland than in arable land (Goulding 2016; Müller et al. 2022). The pH value of a soil is largely influenced by 1) the mineralogical composition of the parent material, 2) management practices (liming and fertilisation) and 3) the vegetation cover and SOC contents (Müller et al. 2022). Table 6 provides a summary of studies comparing the effects of land use on the soil pH.

Table 6: Summary of results measuring effects of land management on soil pH-levels. The comparison was simplified into categories as being larger on grassland (GL > AL), larger on arable land (GL < AL) or not significantly different (GL = AL).

Property	**Relation**	**Number of sites**	**Sampling strategy**	**Sampling time**	**Depth [cm]**	**Soil type / texture class**	**Study region**	**Reference**
pH-level	GL < AL	2	paired sites	n. a.	0 – 15	n. a.	Northeast Pakistan	(Abbasi and Rasool 2005)
		2	paired sites	July 2004 – May 2005	0 – 30	n. a.	Northeast Pakistan	(Abbasi et al. 2007)
		1	repeated sampling	n. a.	0 – 20	Clay loam	Indagai Mountain Pass,	(Başaran et al. 2008)

							Cankiri, Turkey	
		4	paired sites	August 1999 – June 2001	0 – 10	Typic Hapludoll (coarse-loamy)	Córdoba, Argentina	(Bedano et al. 2006)
		3009	meta-analysis	2011 – 2018	0 – 50	Varying	Germany	(Müller et al. 2022)
		6	paired sites	n. a.	0 – 30	Vertic Cambisol	Calabria, Italy	(Papini et al. 2011)
		313	chronose-quence	1995 – 2001	0 – 10	n. a.	New Zealand	(Sparling and Schipper 2004)
		4	paired sites	October 2002 – October 2004	0 – 10	Sandy loam	Melle, Belgium	(van Eekeren et al. 2008)
		717	chronose-quence	2000 and 2004	top horizon	Cambisols (mostly), Leptosols, Regosols, Stagnosols, Albeluvisols, Planosols, Gleysols	Bavaria, Germany	(Wiesmeier et al. 2012)
	GL = AL	4	paired sites	May 2004	0 – 50	Stagnic Vertisol, Arenosol	Thuringia, Germany	(Don et al. 2009)
		6	chronose-quence	2008 – 2009	0 – 30	Silt, clay, clay loam, silty loam	Welles-bourne, England	(Gosling et al. 2017)

Mineralogical composition

The carbonate content of the parent material largely determines the geogenic base level of the soil, its ability to buffer pH changes and resist acidification (Don et al. 2009; Papini et al. 2011; Müller et al. 2022). Furthermore, there is a strong relationship between clay minerals and pH, as clay-rich soils are able to bind more cations, thus buffering pH changes and having higher pH values than soils with lower

clay content (Başaran et al. 2008). Due to higher production demands and economic pressures, fertile clay soils in the temperate zone are predominantly used for arable cultivation, while grasslands are largely located on less productive soils (e.g. Hodgson et al. 2005; Taube et al. 2014; Liu et al. 2017; Griffiths et al. 2020; Smit et al. 2021; Müller et al. 2022). This bias towards arable soils being associated with higher clay content is reflected by the pH level. Accordingly, Başaran et al. (2008) reported significantly lower pH levels in grassland soils (6.9 ± 0.05) than in arable soils (7.2 ± 0.03), as the soil pH was spatially correlated with the soil clay content and arable soils were mostly found on clay-rich soils. Müller et al. (2022) also partly attributed lower pH-levels in grassland to their distribution, as they are predominantly located on sites with acidic parent material.

Fertiliser application and liming

The application of acidifying fertilisers, e.g. sulphur-fertilisers, ammonium-based fertilisers or urea, is a key contributor to decreasing soil pH in both arable and grassland (Bolan et al. 2003; Goulding 2016; Holland et al. 2018; Tibbett et al. 2019). A meta-analysis by Chen et al. (2018) found that the application of acidifying N fertilisers resulted in a significant decrease in soil pH by an average of 0.10 units compared to control treatments. Fertiliser-induced soil acidification is particularly detrimental to nutrient availability on already acidic soils and therefore requires countermeasures (Chambers and Garwood 1998; Goulding 2016; Müller et al. 2022). The pH buffering and -raising effect of base cations, e.g. by liming, can be used for agricultural practices on acidic soils to counteract acidification and achieve conditions favourable for agricultural use (Sparling and Schipper 2004; Goulding 2016; Horakova et al. 2017; Müller et al. 2022). Sparling and Schipper (2004) therefore reported elevated pH values in arable land (6.17 ± 0.09) and grassland (5.74 ± 0.03) as a result of long-term lime application in comparison to sites without lime application. An analysis of soil inventory data by Müller et al. (2022) also reported liming as an important factor explaining soil pH. Liming was preferentially applied to arable land, probably due to the greater economic importance of arable land over grassland at the farm level: 65.4% of arable land and 22.8% of grassland received lime application, although a larger proportion of grassland soils (52%) than arable soils (41%) had pH values below the recommended optimum (Müller et al. 2022).

Plant cover

Soil pH is highly dependent on the plant cover, which has an acidifying effect on the soil pH via organo- and humic acids excreted by roots (Abbasi and Rasool 2005; Abbasi et al. 2007; Başaran et al. 2008). Thus, several studies confirm lower pH-levels in grassland soils than in arable soils due to the acidifying effect of a perennial vegetation and the negative correlation between SOC and pH (Sparling and

Schipper 2004; Abbasi and Rasool 2005; John et al. 2005; Bedano et al. 2006; Abbasi et al. 2007; van Eekeren et al. 2008; Papini et al. 2011; Wiesmeier et al. 2012). The lower soil pH of a former grassland or a crop rotation with grass can persist for two (Gosling et al. 2017) to three years or longer (van Eekeren et al. 2008) after conversion to arable land. However, for similar SOC contents and otherwise similar site conditions, differences in pH-levels may be small, as shown by a report by Don et al. (2009), where the surface soil pH (top 20 cm) did not differ significantly between two pairs of arable and grassland sites, because the SOC content was limited by the available mineral surface required for SOC sequestration.

2.8. Cation exchange capacity

Cation exchange capacity (CEC) is an indicator of the soil's ability to store and deliver nutrients as well as retain contaminants, and is an essential parameter for monitoring chemical soil quality (Mattila and Rajala 2022). The CEC is positively correlated with SOC and clay contents (Abbasi et al. 2007) due to their cation binding capacity (Beldin et al. 2007; Sidi et al. 2015). Land-use related changes in SOC thereby outweigh the effects of the clay content on the CEC (Abbasi et al. 2007). Thus, perennial vegetation with higher SOC inputs typically results in a higher CEC and better nutrient availability under grassland (Haynes and Francis 1990; Franzluebbers et al. 2000; van Eekeren et al. 2008; Gajic 2013). Abbasi et al. (2007) showed higher CEC in grassland soils (16.98 cmol kg^{-1}) than in arable soils (14.01 cmol kg^{-1}) when averaging measurements over depth (0 – 30 cm) and time (bi-monthly measurements over two years). Small differences in clay and SOC contents between arable and grassland may in turn result in non-significant differences in the CEC (Papini et al. 2011). Lower SOC content and CEC under arable land can be partially compensated by organic fertiliser application, although a low clay content may limit SOC accumulation and thus have no effect on the CEC (Don et al. 2009). Available studies comparing the effects of arable use and grassland use on the CEC are summarised in Table 7.

2.9. Electrical conductivity

Electrical conductivity (EC) is a measure of the salt ions in a soil solution. Because the EC of a soil depends on the amount of moisture, it is primarily correlated with the soil texture and increases with increasing clay contents (Grisso et al. 2009; Moral et al. 2010). In turn, management practices, especially fertilisation, irrigation and tillage, have a direct effect on salt ion concentrations. Changes in EC may therefore indicate changes in nutrient availability. Reports on differences in EC between arable land and grassland sites are scarce, thus the effect of land management on the EC is elusive and requires more attention. Only one study provided a direct comparison of EC in arable and grassland soils: Abbasi and Rasool (2005) found a significantly higher EC in arable land (0.225 dS m^{-1}) than in

grassland (0.217 dS m^{-1}), although the authors did not attempt to explain the underlying processes or the relatively small magnitude of the difference. Further details on the direct comparison of the EC in arable and grassland soils are summarised in Table 7.

Table 7: Summary of results measuring effects of land management on the cation exchange capacity (CEC) and electrical conductivity (EC). The comparison of a given property was simplified into categories as being larger on grassland (GL > AL), larger on arable land (GL < AL) or not significantly different (GL = AL).

Property	Relation	Number of sites	Sampling strategy	Sampling time	Depth [cm]	Soil type / texture class	Study region	Reference
CEC	GL = AL	4	paired sites	May 2004	0 – 50	Stagnic Vertisol, Arenosol	Thuringia, Germany	(Don et al. 2009)
		6	paired sites	n. a.	0 – 30	Vertic Cambisol	Calabria, Italy	(Papini et al. 2011) [4]
	GL > AL	2	paired sites	n. a.	0 – 15	n. a.	Northeast Pakistan	(Abbasi and Rasool 2005)
EC	GL < AL	2	paired sites	n. a.	0 – 15	n. a.	Northeast Pakistan	(Abbasi and Rasool 2005)

[4] *similar values for arable and grassland reported on two sites, while higher values on grassland reported on one site.*

2.10. Summary and conclusion

This review summarises and discusses the current knowledge on the effects of grassland and arable land use in the temperate zone on chemical soil properties, considering the provision of ecosystem services (e.g. SOC sequestration, mitigation of N leaching and gaseous N emissions, nutrient provision, regulation of soil pH). In addition to the effects related to long-term arable and grassland use, this paper also addresses the effects of short-term changes. Arable and grassland use requires sustainable management practices in order to maintain soil quality as well as nutrient availability to ensure a resilience to climate change, which will remain a challenge for future decades. Nevertheless, this is often a balancing act between several viable options and conflicting interests, as there is no single "silver bullet" solution for achieving both maximal sustainability and productivity on one site. A preferred land use, whether arable land or grassland, should be strictly adapted to the climatic and pedogenetic conditions in situ, taking into account the advantages and disadvantages of the sites

under both uses, while opting for reduced management intensity to compensate for negative management-related effects.

Arable land and grassland provide different sets of ecosystem services, with arable land focusing on provisioning services and grassland balancing between provisioning and regulating services. Arable land is essential for the production of food and energy crops, but the negative effects of management on soil quality, high risk of GHG emissions, nutrient losses and associated off-site damage put it in a tight spot between economic and environmental demands, with the former usually taking precedence. In contrast, grassland is more beneficial in terms of soil quality, maintaining biodiversity, SOC sequestration and nutrient loss reduction, although these benefits are often overshadowed by the questionable profitability.

Unless under no or conservation tillage, regular tillage and the associated increased aggregate breakdown, aeration and subsequent biochemical decomposition are central causes of declining SOC under arable land, which also affects long-term fertility. In contrast, perennial and undisturbed vegetation under grassland promotes SOC sequestration. The frequent removal of crop residues on arable land also contributes to lower C returns, whereas on grassland a greater proportion of C is returned to the soil via residues and animal manure. Grasslands also typically receive larger amounts of organic fertilisers with high C contents whereas mineral fertilisers with low C content are common on arable land. Nevertheless, higher SOC sequestration under grassland can be counteracted by emissions from grazing cattle, especially at high stocking densities.

The application of mineral fertilisers under arable land leads to both higher NO_3^- leaching and gaseous N_2O losses. The risk of leaching is particularly high immediately after fertilisation and tillage, when the soil is loosened, and during bare periods without vegetation cover, due to lack of uptake and increased moisture typical of arable land in autumn and winter. On the other hand, grassland vegetation, which is particularly rich in biodiversity, limits leaching through constant uptake and high contents of SOC, which adsorbs NO_3^-. Furthermore, regular tillage of arable land prevents a complete reduction of NO_3^-, with N_2O being emitted as a by-product in large quantities. The typically higher mineral N inputs and lower SOC contents under arable land further contributes to higher N_2O emissions by increasing the availability of NO_3^- for denitrification. In particular, thawing of previously frozen soils and increasing soil moisture have been associated with increased N_2O emissions. This effect is particularly pronounced on soils that are kept bare in winter due to lack of water uptake, while water fluctuations and the corresponding moisture-driven N_2O emissions are lower under perennial vegetation.

Another concern in agricultural soils is the availability and accumulation of P, which is sometimes a limiting nutrient. In particular, arable land has a larger P removal by harvest. In contrast, on grazed grassland, a greater proportion of P is returned to the soil with animal excreta, and the higher SOC contents further promote P storage and reduce losses. Regular tillage practices and bare periods on arable land also contribute to increased P losses through topsoil erosion. The higher total P losses on arable land motivate a higher need for additional P fertilisation. The required P is often supplied in combination with N, especially when applied as organic fertiliser. Since the amount of applied fertiliser is often determined by the N requirement and the P requirement is lower than the provided input, this results in a substantial risk of P overfertilisation. Especially when eroded into surface water high P contents are a cause for eutrophication. Elevated P contents under grassland are slow to decline due to lower losses and have been reported to persist for decades to centuries, sometimes even preventing the establishment of biodiverse vegetation. For K and other micronutrients, no deficit was found under either arable or grassland, but grasslands were reported to have higher contents, despite lower fertiliser inputs.

Several effects lead to lower pH values in grasslands than in arable soils: on the geogenic side, grasslands are more often located on acidic parent material, whereas arable cultivation is preferred on clay soils with higher pH values and better pH buffering capacity; on the biogenic side, higher SOC contents under permanent grassland vegetation reduce the soil pH; on the management side, due to economic incentives, arable soils are more often treated with lime to counteract soil acidification than grasslands. While a lower pH generally favours nutrient availability under grassland, which was also supported by a higher CEC, severe acidification can be detrimental, increasing nutrient losses and mobility of toxic elements.

2.11. Recommendations

Chemical soil properties can be largely improved by fostering SOC sequestration. High SOC contents improve nutrient availability and also limit N leaching, nutrient losses through erosion and gaseous N emissions. Most management practices that improve chemical soil properties are aimed at “closing” the nutrient cycle, i.e. reducing outputs and increasing returns. Based on the findings presented in this review, we recommend that the following management practices should be implemented, if possible:

- Increase OM inputs on arable land by retaining crop residues, alternating grassland phases within an arable rotation and substituting mineral fertilisers with organic fertilisers;
- Determine organic fertiliser inputs based on N and P requirement to avoid N and P overfertilisation and supply the remaining requirement for N and P through mineral fertilisers;

- Include legumes in crop rotations and grassland mixtures to increase biogenic N fixation and reduce N fertiliser requirements;
- Improve nutrient cycling by fostering a diverse vegetation and avoiding monocultures;
- Reduce erosion by avoiding bare periods on arable land by planting catch crops and implementing reduced and conservation tillage;
- Reduce leaching by avoiding tillage during the wet season from autumn to winter, especially when soil mineral N is high;
- Prefer grazing over mowing on grassland, but also avoid overstocking;
- Monitoring soil pH, CEC and EC, especially if acidifying fertilisers are applied, and liming the soil if necessary to counteract acidification.

Contributor details

Matthias Filipiak: Methodology, Investigation, Data Curation, Writing – Original Draft, Writing – Review and Editing, Visualisation. Katrin Kuka: Conceptualisation, Writing – Review and Editing, Supervision.

Disclosure statement

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3. Simulation-based assessment of the soil organic carbon sequestration in grasslands in relation to management and climate change scenarios

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Simulation-based assessment of the soil organic carbon sequestration in grasslands in relation to management and climate change scenarios

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ABSTRACT

Soil organic carbon (SOC) is crucial for the quality and productivity of terrestrial ecosystems and its sequestration plays an important role in mitigating climate change. Understanding the effects of agricultural management under future climate on the SOC balance helps decision making in environmental policies. Thereby, grasslands will play a key role, since future climate change may prolong the vegetation period.

We used 24 representative grassland sites in Germany to assess the SOC balance obtained from the CANDY model in relation to ten management regimes, 18 future climate change scenarios and different soil types. Simulations were conducted over a period of 110 years.

For most of the selected grassland sites an increase in both air temperature and precipitation was observed in the future climate. The effect of management on the SOC balance largely exceeded the effect of soil type and climate. An increasing management intensity (i.e. three to five cuts) generally increased the SOC balance, while extensive management (i.e. two or fewer cuts) lead to SOC losses. The seasonal variation of precipitation was the most important climate metric, with increased SOC sequestration rates being observed with increasing growing season precipitation. Clay soils had the potential for both highest gains and highest losses depending on management and precipitation. Given an overall lower SOC storage potential in sands and loams, the SOC balance in those soil types varied the least in response to climate change.

Abbreviations: AAM, annual average mean (model variant); AIC, Akaike information criterion; BAT, biologic active time; BD, bulk density; BÜK200, national soil map (Bodenübersichtskarte) in scale of 1:200.000; C, carbon; CANDY, Carbon And Nitrogen Dynamics; CDC, climate data centre; CM, candidate model; CU, cattle units; DM, dry matter; DüV, German fertilization ordinance (Düngeverordnung); DWD, German meteorological service (Deutscher Wetterdienst); FC, field capacity; FM, fresh matter; GCEF, global change experimental facility; GHG, greenhouse gas; GiFACE, Free Air CO_2 Enrichment experiment in Giessen; GM, global model; Hc, heat capacity; IPCC, Intergovernmental Panel on Climate Change; IVM, inter-annual variance of means (model variant); KA5, German soil classification guideline; Ks, saturated hydraulic conductivity; MIV, mean intra-annual variability (model variant); N, nitrogen; PD, particle density; PTF, pedotransferfunction; PWP, permanent wilting point; RCP, representative climate pathways; RMSE, root mean square error; RRMSE, relative RMSE; R^2, coefficient of determination; SAC, spatial auto-correlation; SOC, soil organic carbon; SP, scenario and projection (model variant); SSP, shared socioeconomic pathways.

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We conclude that fostering SOC sequestration is possible in grassland soils by increasing management intensity, which involves increased fertilizer input and field traffic. This however may stand in conflict with other policy aims, such as preserving biodiversity. Multicriterial assessments are required to estimate the nett greenhouse gas balance and other aspects associated with these management practices at a farm scale.

1. Introduction

In a resolution from March 24, 2021, Germany's Federal Constitutional Court established a new goal of reducing agricultural greenhouse gas (GHG) emissions by 34% by the year 2030, in comparison to the reference year 1990, and achieving climate neutrality by 2045 [1]. Sequestration of soil organic carbon (SOC), i.e. the removal of carbon dioxide (CO_2) from the atmosphere and increasing the C pool size in soil and vegetation, is a central strategy for the mitigation of GHG emissions [2]. Recent studies show the importance of identifying potential synergies between land-based adaptation and mitigation strategies, linking issues of SOC sequestration with emissions of GHGs and long-term sustainability of production systems within coherent climate policy frameworks [2–6]. The soil C cycle is furthermore central to the functioning of terrestrial ecosystems [4,7–11] and high SOC contents are also beneficial to soil functions relevant for agriculture, such as the regulation of nutrient-, water- and temperature flows and the soil structure [12–16]. Due to their year-round plant cover, high biodiversity and usually undisturbed soil, grassland areas are important ecosystems for SOC sequestration [10,17–23].

For steady state soil systems, the equilibrium between C input and release results in a characteristic SOC content depending on local environmental conditions [24], e.g. management, climate and soil properties [15,25,26]. There is evidence suggesting a prevalence of management-related effects (i.e. cutting frequency, grazing intensity, fertilization) on SOC sequestration over that of soil type and climate [15,19,27,28]. Therefore, it is both possible and necessary to adapt management practices, e.g. by increasing the cutting frequency, grazing regimes or fertilizer input, to regional climatic conditions expected in the future decades in order to increase the sequestration potential of grassland soils [29–32]. Issuing of CO_2 certificates can provide the necessary incentive for farmers to implement such management practices [33]. However, the magnitude of the individual effects varies substantially across soil types and climates. It is therefore crucial to identify soil characteristics and management practices that display a high potential for long-term SOC sequestration under a broad spectrum of climatic conditions [3,31].

The analysis of long-term SOC sequestration requires prognostic approaches. Hence modelling the terrestrial C cycle has become a major research topic, which allows identifying key drivers for SOC sequestration and thereby making management recommendations based on present and expected future conditions. Process oriented ecosystems models that account for the major processes and interactions between various components of the ecosystems are the best available tools to increase our understanding and to make descriptions of impacts of climate change and its variability [3]. Models like Roth-C [34], DNDC [35], MONICA [36], DAYCENT [37], PaSim [38], ecosys [39], ECOSSE [40,41] and CANDY [42] have been developed in order to simulate long-term SOC turnover and its stabilization in the soil. Their application allows a scenario-based evaluation of management practices and their impact on terrestrial C cycles as well as the identification of preferential sites with high potential of long-term SOC storage. Several studies already conducted precise assessments of soil productivity, SOC dynamics and stocks on a grid-level up to a resolution of 1 km on regional, national and even continental scales, thereby substantially contributing to the assessment of the global sequestration potential (e.g. Refs. [3, 43–47]). Nevertheless, these studies rarely investigate changes under future climate and those that do (i.e. [3,43,45,46]), consider only few climate scenarios, thereby limiting the robustness of the results to the uncertainty of future climate change, and furthermore lack multiple management variants considered per simulation unit, thereby not providing any suggestions to management alternatives. Other simulation studies dedicated to specifically investigating the interactions of climate effects across multiple sites and management practices are limited either by a narrow gradient of management regimes considered, lack of future climate scenarios, too few sites considered or lack of replication of different management regimes under varying pedo-climatic conditions (e.g. Refs. [48–51]).

This study is motivated by the necessity to amplify the SOC sequestration in agriculture in order to meet the goals imposed by global and national policies [1,2] and provides a foundation for political decision-makers for management practices that are most suited to these goals under the predicted future climate change. The focus of this study is on the investigation of the SOC balance in grassland soils depending on soil characteristics, management intensity and climate change over a long period using the CANDY model. Validations of CANDY-based simulations of several ecosystem functions, including SOC sequestration [52,53], water quality [54] and lateral as well as vertical nutrient transport [55] were successfully conducted on various international long-term field experiments and study sites. In order to increase the national coverage and to account for uncertainties in the predicted SOC sequestration that arise from future climate scenarios [56], simulations were conducted across 24 representative grassland sites in Germany by applying 18 site-specific representative climate pathways (RCP) scenarios [57] and ten management regimes over a period of 80 years (2020–2099), including a baseline simulation of 30 years under ambient climate (1990–2019), which allows for a steady-state initialization of the SOM pools under conditions representative of each site.

The simulations aimed at testing the following hypotheses:

1. The impact of climate change on the SOC balance depends on the soil type.
2. Increasing management intensity can enhance the SOC sequestration rate.
3. The effect of management on the SOC balance prevails regardless of soil types and climate.

In the following sections, we first introduce the CANDY model, its core concepts and the model initialization process. We then describe the data used for parametrization, validation and scenario simulations and present our approach of the selection of future scenario sites, derivation of representative management regimes and future climate scenarios. Then, the initialization of the future scenario simulations is described, followed by a section outlining the statistical evaluation based on the multimodel inference approach [58]. We present the results of the model parametrization, validation and the change of the topsoil SOC contents in the future scenario period (2020 to 2099) in dependence of site properties (soil type, management, climate). We finally identify essential drivers that promote SOC sequestration as well as site conditions with the highest potential for sequestration and site conditions with the highest risk of SOC losses under future climate change.

2. Material and methods

The general work process encompassed the re-parametrization of the CANDY model with field experiment data, validation with long-term soil survey sites, derivation of management regimes, selection of scenario sites based on a variety of digital map data and subsequent simulation and evaluation of future SOC balance (Fig. 1).

2.1. CANDY model

CANDY [42,52] is a deterministic, one-dimensional mathematical process model that can simulate the C- and N-dynamics, soil temperature and soil water dynamics in a soil profile of the unsaturated zone of arable land and grassland up to a daily resolution. In CANDY, the concept of biological active time (BAT) is implemented to determine the site-specific turnover conditions in the soil, which reflect on the impact of temperature, soil moisture and aeration of the soil on the SOC turnover [73]. Therefore, this model is particularly useful to evaluate the effect of climate scenarios on SOC turnover. The soil profile to be simulated is divided into homogeneous horizons up to a depth of 200 cm. Simulations are conducted based on a series of fixed generalized parameters (e.g. plant development characteristics, nutrient turnover- and transport rates) as well as site-specific drivers, i.e. soil properties, meteorological data and agricultural management data [74]. For grassland simulations using CANDY the following input data are required: 1) Annual land use information (sowing date; beginning and end of grazing, number of cattle units; type, date and amount of fertilizers applied; cutting date and yield), 2) daily climate data (air temperature; precipitation; global radiation or sunshine duration) and 3) physico-chemical soil properties of each homogenous soil horizon (soil texture; bulk (BD) and particle density (PD); rock content; water content at permanent wilting point (PWP) and field capacity (FC); saturated hydraulic conductivity (Ks); specific heat capacity (Hc)).

The sowing event specifies the cultivar and land use of any given simulation unit. Due to the continuous nature of the simulated permanent grassland, the sowing event is solely required for the site initialization.

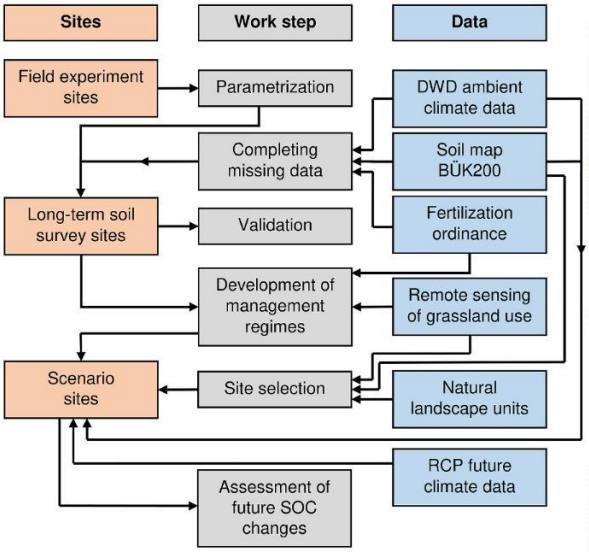

Fig. 1. Simplified flowchart of all work involved in the assessment of the sequestration potential. The work steps are ordered sequentially from top to bottom. Arrows indicate the direction in which results or data were used for later work steps or sites. Field experiments served the parametrization, which was validated based on long-term soil survey sites. Data gaps in validation sites were filled using the BÜK200 soil map [dataset] [59], the DWD Climate Data Centre [60–62] and the German fertilization ordinance [63]. Management regimes were developed based on the fertilization ordinance, remote sensing data [dataset] [64–66] and data from the validation sites. Scenario sites were selected based on the remote sensing data, the BÜK200 and natural landscape units [67]. Input data for scenario sites was acquired from the BÜK200 [59], DWD Climate Data Centre [60–62] and RCP scenarios [57,68–71,72]. Future SOC changes were assessed based on the scenario sites.

The physico-chemical soil properties for each site were assumed static, since they were derived from generalized map data. Internal pedotransferfunctions in the CANDY model (PTFs, [42,52,75]) allow the estimation of missing soil properties (e.g. PWP, FC, Ks) based on easily available properties (e.g. texture, BD). The SOM pools within CANDY are by default initialized in a steady state. The calculation of the inert C content was always done with the equation "particle surface" according to Refs. [74,76], which was identified as a reliable method to initialize the long-term-stabilized pool of the model (see also [77,78]).

The initial value of the SOC content and its division into the convertible pool and the inert pool at the start of the modelling have a decisive impact on the simulation results especially in short periods of few years [78,79]. This requires a high precision of the initial value, which does not always apply to measured field data due to systematic or methodological errors of SOC measurements. In addition, a measured SOC value is not always available when starting the simulation. Hence, to initialize the model, the CANDY Optimizer Tool in the version 20.15.2.2 [80] was used for the parametrization and validation. An optimized starting value was determined by a stepwise modification by means of several simulation runs (typically up to 50) until the root mean square error (RMSE) was minimized. This procedure allows to reduce the statistical error of simulations and bring the start of the simulation forward when necessary [81]. Combined with the steady-state initialization and partition of the C pools as outlined above, this procedure allows to begin simulations with the period of interest without the requirement to account for past management (see also [77,78]). Due to its prior focus on arable land, selected parameters within the CANDY model (i.e. the transpiration coefficient, the duration of the vegetation period and the C/N-ratio in fresh organic matter) were re-parametrized to increase its accuracy and robustness on grassland sites.

2.2. Data availability

2.2.1. Field experiments

For the parametrization, datasets including the management history, climate and pedogenic properties from three field experiments were used: 1) the Global Change Experimental Facility in Bad Lauchstädt (GCEF, 20 plots with 4 years of SOC measurement data each, [82]), 2) the Free Air CO_2 Enrichment experiment in Giessen (GiFACE, 3 plots with 21 years of data each, [83]) and 3) the Jena Experiment (5 plots with 7 years of data each, published in the PANGAEA network [dataset] [84–89]). These data are particularly suitable for parametrization since they were measured under controlled management conditions (i.e. similar dates of operations, similar fertilizer input under all replicated plots) that remain largely constant across the entire duration of the field experiment and represent a high level of detail.

2.2.2. Long-term survey sites

The validation was carried out with data from long-term survey grassland sites from Lower Saxony (14 sites with a total of 256 years of SOC measurement data), Brandenburg (3 sites, 126 years of data) and Baden-Württemberg (6 sites, 137 years of data). The sites are spread across a variety of soil types and climatic regions in Germany. These data were also used for the derivation of management regimes for the scenario simulations, alongside the datasets of long-term survey sites from Thuringia (4 sites, 94 years of data), Bavaria (23 sites, 610 years of data), Saxony (2 sites, 19 years of data) and Hesse (7 sites, 111 years of data).

2.2.3. Map data

A broad variety of digital geographic map material served as basis for the selection of scenario sites, derivation of soil properties and management regimes, using Arc GIS 10.3 [90]. The map data include 1) the open-access national soil map BÜK200 [dataset] [59], 2) a digitalized climate landscape map [67] and 3) remote sensing data on grassland use, which is also available online [dataset] [64–66]. The [dataset] BÜK200 contains 1) site-specific information in a scale of 1:200.000 on soil system unit, soil region, the soil type including physical and chemical soil properties according to the German classification system provided by the KA5 soil mapping guideline [91] and 2) land use information. CANDY simulations based on soil maps were proven viable in previous studies [79]. The climate landscape map provides a classification of 506 natural landscape units within Germany in a scale of 1:200.000. The remote sensing data comprises raster data across Germany on 1) grassland use in the year 2016, 2) the annual number of cuts for 2017 until 2019 and 3) cutting dates for 2018 and 2019 in a spatial resolution of 30 m [dataset] [64–66]. These digital map data provide soil data and management information in a sufficient spatial resolution for the application of the CANDY model.

2.2.4. Selection of scenario sites

For the selection of adequate and representative scenario sites for future climate simulations, several criteria were set. On a first level, the largest possible national coverage and representativeness of both climatic regions and soil types by the selected simulation sites were sought. For this purpose, the natural landscape units were grouped by soil regions. This was achieved by assigning each of the 506 polygon units of the natural landscape map to one of 12 soil regions according to BÜK200 by the largest area share the soil region takes up in the respective natural landscape.

In the next step, within each grouping of natural landscapes by soil region, the two largest natural landscape units were selected. A scenario site was determined within a selected natural landscape unit which simultaneously fulfils the following requirements: 1) it was under continuous, long-term grassland use according to the available grassland masks of remote sensing analyses from 2016 to 2019 [dataset] [64–66] as well as land-use information provided by the BÜK200 and 2) it is the most widespread soil type of grassland in the selected natural landscape unit (excluding organic soils) according to the BÜK200.

2.2.5. Scenario climate data

The required ambient climate data (1990–2019) as model input were acquired from climate stations in the closest proximity to the selected scenario sites (see supplementary material S1 for a detailed summary of ambient climate data, including site coordinates and weather stations). For the future climate (2020–2099), site-specific data from RCP (representative climate pathways) scenarios, which were created as part of the IPCC Fifth Assessment Report [57] and served as basis for the creation of the recent SSP (shared socio-economic pathways) scenarios [92], were used. While the recent SSP scenarios describe different socioeconomic pathways that could lead to different levels of GHG emissions as an effect of population growth, economic development, energy use, and technological change, the RCP scenarios quantify the increase in the radiative forcing in comparison to pre-industrial times by 2.6, 4.5 and 8.5 W m^2, which corresponds to an increase in CO_2-equivalents to 490, 650 and 1370 ppm respectively by the year 2100, resulting in different patterns of precipitation, air temperature and global radiation. The RCP scenarios do not explicitly consider socioeconomic factors or policy decisions and avoid uncertainties associated with assumptions about socioeconomic drivers, making them useful to assess the potential physical impacts of climate change.

Each RCP scenario is realized by multiple projections resulting from simulations of different climate models and are therefore possible outcomes of future climate conditions [57]. In order to reduce the impact of climate model-specific errors and uncertainties, the selection should include as many projections (i.e. climate models) per RCP scenario as possible. From all available climate models, those models were selected that provided site-specific climate data (global radiation, precipitation, air temperature) for each day of the year in all three RCP scenarios (see supplementary material S2 for list of climate models used for future data and supplementary material S3 for a summary of future climate data). A detailed description of the RCP scenarios can be found in Refs. [68–72,93]. The raw data can be accessed at the [dataset] [72].

2.2.6. Closing data gaps

The validation data contained various gaps concerning climate, soil and management data, which were closed using a variety of open access sources as described below.

A total of 10.6% of climatic data (largely sunshine duration; see also supplementary material S4a) were completed using the online and open-access DWD Climate Data Centre [60–62]. Site-specific coordinates of the grassland sites served to identify the spatially closest climate stations for each site. In some cases, data from more than one climate station had to be used, if the closest climate station did not provide a complete data set.

Management data of the validation sites displayed partly large gaps (35.8%) due to the descriptive character of provided management information (see also supplementary material S4b). Missing dates for management operations, such as cutting, grazing and fertilization, were completed with available long-term means for the respective site. Missing stocking densities, fertilizer input and yield were derived from long-term means and descriptive management documentations of the original dataset while considering yield levels and nitrogen (N) requirements based on the German fertilizer ordinance [63], which is the state of the art guideline for agricultural practices.

18.9% of soil data (see also supplementary material S4c), largely for subsoil horizons, were initially missing. Texture, BD and rock contents for the validation sites were completed with the BÜK200 soil map by using site-specific coordinates. Therein, the soil profile parametrizations were provided as classes, grouped by specific ranges of the properties, according to the soil mapping guideline KA5, which is predominantly used across Germany [91]. For each class of sand-, silt-, clay- and rock content as well as BD the median between the upper and lower delimiter was used. The BÜK200 usually provides several parametrizations per location. From all available parametrizations, we chose the one that shared the same soil type according to the KA5 classification with the to-be-completed soil and, if possible, was designated as a grassland soil within the BÜK200. If no parametrization with the same soil type was available, the most similar soil type based on expert knowledge was selected instead. Missing data on the PWP, FC and Ks were acquired by using the PTFs integrated within the CANDY model. For Hc the standard value of 0.16 J cm^3 K^1 was assumed, which proved reliable in prior simulations [42,52,75]. Similarly, a PD of 2.6 g cm^3 was assumed for all horizons due to the predominance of quartz in soil particles [20,94], which was also validated by prior measurements and simulations [95]. Soil properties for the 24 scenario sites were acquired analogously (see supplementary material S5 for detailed soil properties as well as soil classification according to the KA5 classification guideline [91] and WRB [96]). In four cases simulations with the median values of texture classes caused unrealistic soil water simulations in the scenario sites, since the BD and texture class provided by the BÜK200 resulted in a pore volume below the simulated FC according to the internal PTFs. In these cases, the texture was lowered to the lower limit of the respective class.

2.3. Management regimes

Ten management regimes representing an intensification gradient from low (1) to high (10) were set up based on management data from long-term survey sites as well as remote sensing data provided by [dataset] [64–66] as model input for scenario simulations. The regimes include management with 1) one to five cuts per year and mineral fertilization (meadows), 2) one to four cuts and grazing, mineral as well as organic fertilization (mown pastures) and 3) grazing only (pasture) (see supplementary material S6 for detailed information of the management regimes). The management operations within each regime were kept constant across all years to allow assessing the effect of different land use practices across changing environmental conditions within each scenario.

The yield levels and fertilizer quantities correspond to the yields of the respective intensity of grassland use as well as their N requirement [63]. The stocking rates of grazing cattle were derived from available management data on grazed grasslands from the validation sites. Therein, the stocking rates as well as grazing duration varied highly. The management regimes account for this

variability by reducing the stocking rate under regimes with a longer grazing duration and thus keeping the grazing intensity constant across all regimes. With regard to the N requirement, grazing was considered as an additional cut, in accordance with the fertilizer ordinance [63]. The N content of organic fertilizer was determined based on data provided by the Lower Saxony Chamber of Agriculture [97].

2.4. Initialization of scenario simulations

The simulations were carried out for each combination of the 24 selected scenario sites, ten management regimes and the three RCP scenarios with six climate models each. This resulted in 4,320 individual simulations over a period of 110 years (1990–2099), which allows to minimize the uncertainty of future simulations caused by the application of too few scenarios [56]. Due to the conceptual character of the selected scenario sites, it was not possible to initialize them based on the measured SOC content, as was done for the parametrization- and validation sites. Therefore, a uniform initial SOC content of 2 M-% in the topsoil (0–30 cm depth) was assumed, which was the mean value of the validation sites. While a site-specific SOC parametrization is usually more desirable as it produces optimal results, a generalized approach comes with the benefit of the comparability of the individual sites with varying pedogenetic and climatic characteristics [81]. As a result of a sensitivity analysis on CANDY-based simulations [79], it was found that the initial SOC content is the most dominant factor in the first year of simulations and it affects long-term SOC modelling, but its dominance decreases with time, while other site properties increase. To avoid this initialization error and to determine the (model) site specific SOC starting value, simulations were conducted with a 30 year long initialization period, starting simulations in 1990 using ambient climate, following simulations using future climate based on the RCP scenarios from 2020 until 2099. Reference [79] successfully implemented a similar approach with an initialization period of seven years. Due to the relatively slow turnover, SOC changes need to be studied over years and decades, while monthly and daily changes are of relatively minor importance [95]. SOC simulations were therefore conducted in annual time steps and were limited to the topsoil (upper 30 cm), since the majority of vegetation- and land use related effects and nutrient turnover can be observed in this layer [98–101].

2.5. Statistical analysis

Statistical analysis was conducted using the open source software R version 4.0.2 and R Studio version March 1, 1073 [102]. Visualization of data was performed with the R package *ggplot2* [103]. Correlation of model input data was explored with the *corrplot* package [104].

2.5.1. Parametrization and validation

To validate the parametrization of the CANDY model, the root mean square error (RSME), relative RMSE (RRMSE) and coefficient of determination (R^2) were calculated from observed vs predicted SOC contents using the *modeval* function within the *sirad* package [105]. The RMSE, RRMSE and R^2 are mathematically expressed as follows:

$$\mathrm{RMSE} = \sqrt{\frac{\sum_{i=1}^{n}(P_i - O_i)^2}{n}} \tag{1}$$

$$\mathrm{RRMSE} = 100\frac{\sqrt{\frac{\sum_{i=1}^{n}(P_i - O_i)^2}{n}}}{\bar{O}} \tag{2}$$

$$R^2 = \frac{\sum_{i=1}^{n}(P_i - O_i)\cdot(O_i - \bar{O})}{\sqrt{\sum_{i=1}^{n}(P_i - \bar{P})^2 \cdot \sum_{i=1}^{n}(O_i - \bar{O})^2}} \tag{3}$$

with P = predicted value, O = observed value, $\bar{P}$ = mean of predicted values, $\bar{O}$ = mean of observed values, n = number of P/O pairs, i = individual P/O pairs [77,106,107].

2.5.2. Scenario simulations

To analyse the effect of management, soil group and climate on the SOC balance, mixed effects models were fitted using the *nlme* package [108]. For the evaluation of the scenario simulation results, the simulated SOC for each of the 4,320 simulation runs across the simulation period as well as management-, soil- and climate data were aggregated by different means.

The SOC balance of every simulation run was determined as the difference between the last simulated value in the year 2099 and the first simulated value at the beginning of the simulations with the future climate, in the year 2020.

The management regimes were considered as a factor with ten levels, due to a high co-linearity of management operations across the ten management regimes (see also supplementary material S7 for a correlation matrix of the management data).

Soil data were aggregated into four levels by their texture class into sands, silts, clays and loams, in accordance to the KA5 soil mapping guideline [91] (see also supplementary material S5). This enables a practical interpretation of the simulation results.

Climate data were taken into account both as qualitative predictors (RCP scenario (three levels) and its projections (six levels) with

a total of 18 levels), and as quantitative predictors, in form of site-specific long-term (2020–2099) means. These were calculated in three ways: 1) as averages of annual means (air temperature and global radiation) and of annual sums (precipitation) (annual average means – AAM), 2) as the average standard deviation within all years (mean intra-annual variability – MIV) and 3) as the variance of annual means (air temperature and global radiation) and annual sums (precipitation) across the entire simulation period (inter-annual variance of means – IVM). The AAM is a relatively static descriptor of climatic conditions, the MIV in turn expresses how much the air temperature, precipitation and global radiation fluctuate due to seasonality, while the IVM describes how much the annual means/sum change over the long-term (80 years). The aggregation by three different methods was motivated by studies reporting that the magnitude of absolute climate metrics may be second to the effects of seasonal fluctuation and long-term change [109,110] as well as a requirement to investigate the impact of seasonality, particularly growing season precipitation, on SOC sequestration [46].

2.5.3. Model inference

A multimodel inference approach was applied [58], which allows to compare and rank several models representing competing hypotheses using information criteria such as the Akaike information criterion (AIC; [111]). At first four global models (GMs) with the fixed effects management (10 levels), soil group (4 levels), climate (see below) and two- and three-way interactions and the random effect site (24 levels) were fitted. The structure of all four GMs can be summarized as follows:

$$\mathrm{GM} = \mathrm{SOC\ balance} \sim (\mathrm{management} + \mathrm{soil\ group} + \mathrm{climate})\hat{}3 + \mathrm{random\ effect}$$

The explanatory variable climate was included either as qualitative predictors (RCP scenario (3 levels) and its projections (6 levels), with a total of 18 levels, GM-SP), or as quantitative predictors air temperature, precipitation and global radiation aggregated by the same method (GM-AAM, GM-MIV, GM-IVM). Interactions between climate variables were not included into the models due to high complexity (see also supplementary material S8 for a correlation matrix of climate data). The variance inflation factor (VIF) was calculated for these models and resulted in values < 2, attesting minor co-linearity between explanatory variables.

Subsequently, candidate models (CMs) were fitted for each of the four GMs, which included all possible combinations of predictors and their interactions using the *dredge* function of the *MuMIn* package [112]. The best CMs within dAIC<2 and non-spatially correlated residuals were interpreted using the *effects* package to produce confidence intervals [113], and the *emmeans* package [114] and the *multcomp* package [115] to perform the post hoc test at an alpha of 0.05. Adjusted R^2 are reported for the model in total, for the fixed effects as well as for the random effects using the *rsq* package [116].

The model performance was assessed by plotting the residuals and several adjustments were made: A variance structure was incorporated into the GMs to account for different heterogeneity between soil type groups using the *varIdent* function of the *nlme* package [108,117]. This variance function improved the fit of models considerably (see supplementary material S9 and S10 for residuals plots).

The spatial distribution of sites was taken into account by fitting spatial correlation structures (none, Gaussian, spherical, rational, exponential, linear) in the GMs using the site coordinates and comparing them by means of the AIC. For this purpose, the site coordinates were modified by a small random number in order to avoid problems related to null-distances due to the usage of multiple climate- and management scenarios per site (see also supplementary material S11 for plotted residuals in dependence of latitude and longitude). Furthermore, spatial autocorrelation (SAC) in residuals was tested for the best CMs at the site-level using the Moran's I at $p < 0.05$ and the *ape* package [118].

3. Results

3.1. Parametrization and validation

In comparison to the initial CANDY parametrization, the transpiration coefficient was increased from 0.8 to 8.0 kg mm^1. In all cases, the re-parametrization resulted in an increase of the performance in comparison to the initial version. The duration of the vegetation period displayed a minor effect on the performance statistics. The overall best performance was achieved by increasing the duration of the vegetation period from initially 254 to 256 days. The model performance across all parametrization data displayed a good fit with an RRMSE of 8.22% and R^2 of 0.955 (Table 1, Fig. 2). The new parametrization was validated by 23 long-term soil survey sites. Despite a poor R^2 on silts, likely due to relatively low SOC spread, the data produced an overall RRMSE of 15.99% an R^2 of 0.845, confirming a reliable parametrization across all sites (Table 2, Fig. 3).

Table 1
Model performance statistics after complete parametrization of the CANDY model based on the experimental sites of the GCEF-, GiFACE- and Jena-datasets, including number of available SOC measurements (n).

Dataset	RMSE [M-% SOC]	RRMSE [%]	R^2	n
GCEF, Bad Lauchstädt	0.101	4.96	0.359	80
GiFACE, Giessen	0.377	9.22	0.793	27
Jena Experiment, Jena	0.155	8.21	0.472	20
All data	0.200	8.22	0.955	127

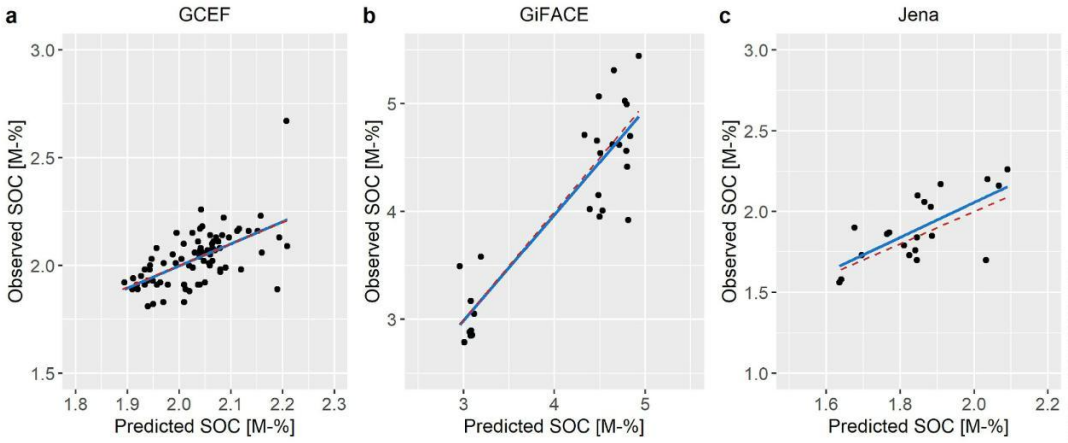

Fig. 2. Observed and predicted SOC contents in the topsoil layers of the parametrization dataset. Continuous line shows the fitted linear regression, dashed line shows the theoretical ideal 1:1 fit.

Table 2
Model performance statistics for the validation sites in Lower Saxony, Brandenburg, Baden-Württemberg and Thuringia, grouped by soil type and including number of available SOC measurements (n).

Soil type	RMSE [M-% SOC]	RRMSE [%]	R^2	n
Sand	0.650	21.31	0.692	31
Silt	0.473	8.68	0.002	11
Clay	0.723	16.16	0.784	76
Loam	0.312	5.10	0.995	3
All data	0.677	15.99	0.845	121

3.2. Scenario simulations

Fig. 4 shows the simulated SOC balance for the 24 scenario sites with averages of three RCP scenarios, six projections and ten management regimes. The only site with an overall negative mean SOC balance (a clay-rich soil on site 7, see also supplementary material S5) was located in the soil region "loess and sandy loess area" in Thuringia. This site displayed low precipitation amounts (<620 mm) and moderate to high mean air temperatures (10.30–11.63 ◦C) across all three RCP scenarios (see also supplementary material S3). The majority of sites with a strong SOC increase were located in the Alps regions in southern Germany, particularly Bavaria (sites 20–24). These fine-textured sites (30–45 M-% silt and 20–40 M-% clay content in the topsoil) were characterized by high precipitation amounts (partially above 1.000 mm) and low mean air temperatures (<10.0 ◦C for RCP 2.6 and 4.5). Sites in northern Germany (sites 1, 5 and 6) displayed in average only minor increases as a result of moderate precipitation (780–840 mm across all scenarios) and air temperature (9.48–11.35 ◦C across all scenarios).

3.2.1. SOC-balance in relation to climate, soil and management

The multimodel inference identified site properties (climate, management and soil type) and interactions that substantially affect the simulated SOC balance. The best fit for the GM-AAM and GM-MIV models was achieved with an exponential and Gaussian structure respectively, while the GM-IVM model produced the best fit using no spatial correlation structure. In all cases, only one CM was within dAIC<2 (see Table 3). Among the models which use qualitative climate predictors (projection and scenario, GM-SP), the best CM included all terms of the GM except for the interaction between soil group and climate scenario and all three-way interactions. Among the AAM and MIV models respectively, the best CMs were the GMs. The best CMs, CM-AAM-1 and CM-MIV-1, therefore solely differed by their aggregation of climate data and SAC structure, with the same predictor terms and interactions (but different coefficients). Among the IVM models, the best CM included all terms of the GM except for the three-way interaction between soil group, management and the IVM of temperature. From all GMs, GM-AAM displayed the highest empirical support with the lowest AIC, while GM-SP displayed the lowest, with the exception of the null model M0.

Among the SP, AAM, MIV and IVM models, all models displayed a similar overall performance, judged by the total R^2 and AIC, and thus outperform M0. The proportion of variation explained by the fixed effects was highest for the MIV models, and similar for the AAM and IVM models. Conversely, the proportion of variation explained by the random effects was lowest for the MIV models.

SAC within residuals persisted for the models CM-AAM-1, CM-IVM-1 and CM-SP-1 at $p < 0.05$, while no SAC in residuals could be detected for CM-MIV-1 at $p > 0.05$ (see Table 4). From all available models, only CM-MIV-1 fulfilled the requirements for model interpretation of having a dAIC<2 and non-spatially correlated residuals. Its suitability was supported by the higher R^2 for fixed effects.

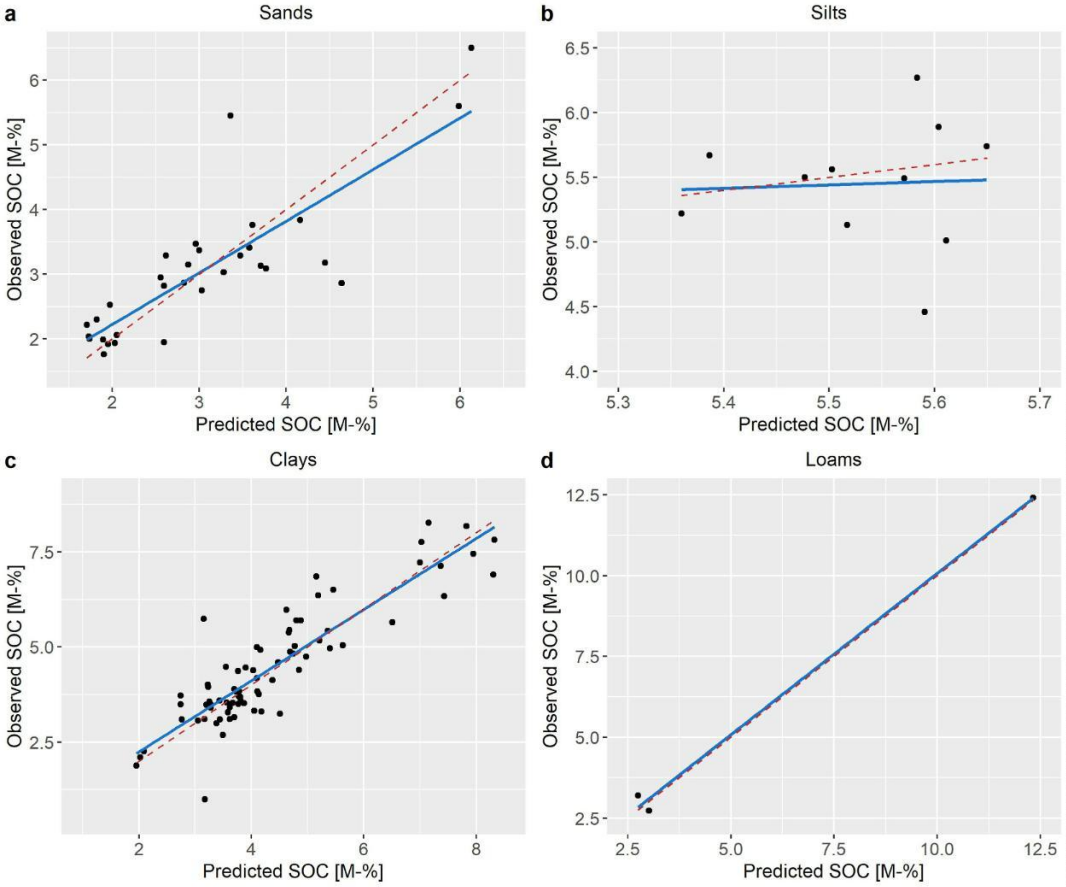

Fig. 3. Observed and predicted SOC contents in the topsoil of the long-term soil survey sites in Germany by soil group. Continuous line shows the fitted linear regression, dashed line shows the theoretical ideal 1:1 fit.

3.2.2. Interpretation of effects

Based on the model CM-MIV-1, changes in the MIV of temperature exercised minor effects on the SOC balance at lower to moderate management intensity (Fig. 5, regimes 1–5). Under an extensive grazing regime (regime 1) with increasing MIV of temperature, the SOC balance of clays remained fairly stable, with an increase by 0.027 M-% per ◦C (see also supplementary material S12 for the SOC change per one unit increase of air temperature, precipitation and global radiation). Loams and silts increased by 0.101 and 0.091 M-% per ◦C respectively, while sands decreased by 0.013 M-% per ◦C. Differences between management regimes and soil groups became more pronounced at higher management intensity. At five cuts (regime 10), the SOC balance of clays increased by 0.176 M-% per ◦C, the balance of loams, sands and silts decreased by 0.032, 0.024 and 0.102 M-% per ◦C respectively.

An increase in the MIV of precipitation resulted in a decreasing SOC balance at lower management intensity (Fig. 6, regimes 1–3) and an increasing SOC balance at higher management intensity (regimes 7–10). Under extensive grazing management (regime 1), the SOC balance decreased by 0.058 M-% per mm on clays, by 0.033 M-% per mm on loams, and by 0.028 M-% per mm on sands. Silts were the only soil group experiencing a slight increase by 0.021 M-% per mm. Under intensive management with five cuts (regime 10), clays showed the highest increase, by 0.105 M-% per mm, followed by sands increasing by 0.024 M-% per mm and loams increasing by 0.009 M-% per mm. The SOC balance of silts in turn decreases by 0.022 M-% per mm.

Changes due to increasing MIV of global radiation displayed similar effects across all management regimes (Fig. 7). Under an extensive grazing regime (regime 1) and increasing MIV of global radiation, the SOC balance increased by 0.024 M-% per 100 J cm2 on clays, by 0.010 M-% per 100 J cm^2 on sands and by 0.093 M-% per 100 J cm^2 on silts, while it decreased by 0.012 M-% per 100 J cm^2 on loams. Under a five-cut regime (regime 10) with increasing MIV of global radiation, the SOC balance increased by 0.067 M-% per 100 J cm^2 on loams, by 0.073 M-% per 100 J cm^2 on sands and by 0.040 M-% per 100 J cm^2 on silts, while it decreased by 0.025 M-% per 100 J cm^2 on clays.

Regardless of soil group, the highest mean SOC increase and simultaneously highest variability during the simulation period was simulated for the regimes 8 (+0.31 ± 0.23 M-%) and 9 (+0.30 ± 0.22 M-%), which represent a land use with four cuts, with regime 9 additionally including a grazing period (Fig. 8a). An increase to five cuts (management regime 10) resulted in a lower SOC increase across all sites (+0.23 ± 0.20 M-%) in comparison to a four-cut use. SOC losses were determined under management regimes 2 (0.15 ± 0.11 M-%) and 3 (0.11 ± 0.06 M-%), which represent a meadow and mown pasture respectively, both under a single cut regime. A

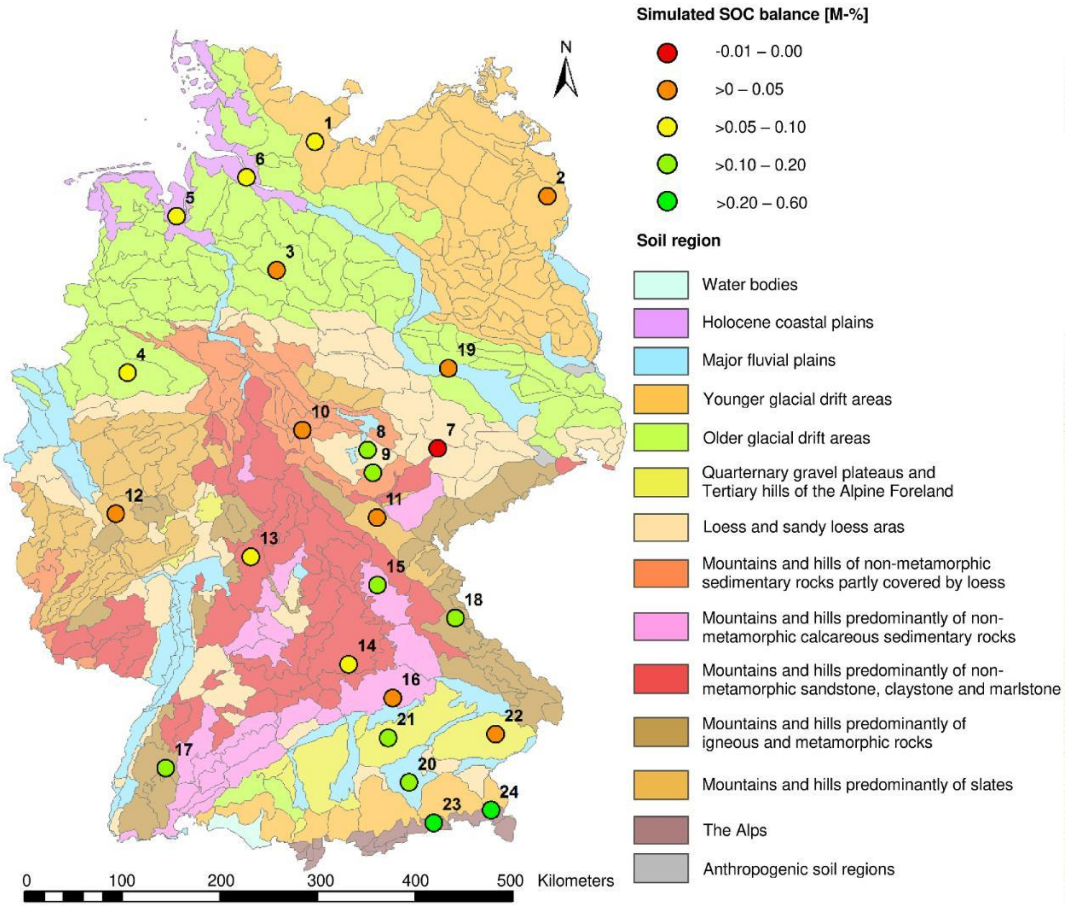

Fig. 4. Simulated SOC balance (averaged across three RCP scenarios, six projections and ten management regimes) for the 24 scenario sites. Soil regions and natural landscapes in accordance to Ref. [67] and [dataset] [59].

Table 3
Summary statistics of the global models (GMs) and null model (M0) in comparison to the best fitting candidate models (CMs) based on a dAIC<2. sg = soil group, mng = management regime, scn = scenario, prj = projection, pre = precipitation, temp = air temperature, rad = global radiation. The models are grouped by the method by which climate data were aggregated (SP, AAM, MIV, IVM). The dAIC always refers to the best CM within the climate aggregation groups.

Model	Explanatory variables	df	AIC	dAIC	R^2		
					total	fixed	random
GM-SP	(sg + mng + scn + prj)^3	455	-9162.7	526.7	0.94	0.52	0.42
CM-SP-1	sg + mng + scen + prj + sg:mng + sg:prj + scen:mng + scen:prj + mng:prj	140	-9689.4	0.00	0.95	0.57	0.38
GM-AAM	sg * mng * (pre + temp + rad)	166	-13145.0	0	0.97	0.66	0.31
CM-AAM-1							
GM-MIV	sg * mng * (pre + temp + rad)	166	-12772.2	0	0.97	0.83	0.13
CM-MIV-1							
GM-IVM	sg * mng * (pre + temp + rad)	165	-11904.8	8.4	0.96	0.68	0.28
CM-IVM-1	sg * mng * (pre + rad) + temp + mng:temp + sg:temp	138	-11913.2	0	0.96	0.68	0.28
M0	-	6	-2332.6	0	0.28	0.00	0.28

low to moderate increase in SOC balance was determined under regimes 1 (+0.04 ± 0.09 M-%), 4 (+0.01 ± 0.09 M-%), 5 (+0.15 ± 0.12 M-%), 6 (+0.08 ± 0.10 M-%) and 7 (+0.13 ± 0.17 M-%). Overall, according to estimated marginal means provided by CM-MIV-1 (Fig. 8b), clays displayed either a similar or a significantly higher SOC balance than loams, sands and silts under all management regimes. Silts in turn displayed the overall lowest, though not significantly lower than sands and loams, SOC balance across all management regimes.

Table 4

Observed and predicted Moran's I values for the best candidate models. Significant spatial autocorrelation persists at $p < 0.05$.

Model	CM-AAM-1	CM-MIV-1	CM-IVM-1	CM-SP-1
observed	0.062	-0.036	0.055	0.032
predicted	0.043	-0.043	-0.043	-0.043
sd	0.035	0.043	0.036	0.028
p	0.003	0.868	0.006	0.008

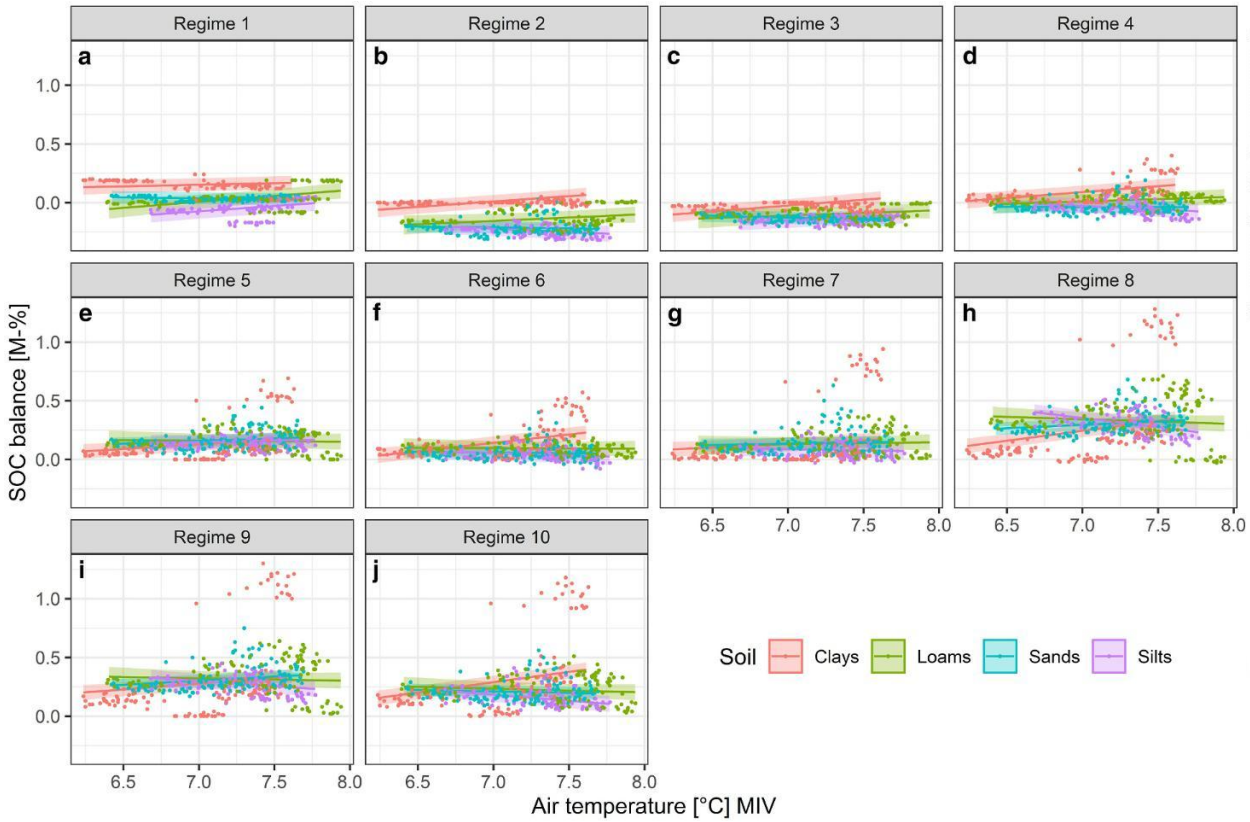

Fig. 5. Linear regressions (lines) and 95% confidence intervals (ribbons) of the effect of the mean intra-annual variability (MIV) of temperature on the SOC balance [M-%] in dependence of the soil type and management regime according to the CM-MIV-1 model.

4. Discussion

In this study, the SOC balance obtained from the CANDY model was analyzed in relation to the effects of management intensity, soil type and climate using 24 representative grassland sites. Management was identified as the most important driver affecting SOC sequestration, though different responses of soil types to the effects of climate were detected. Among climatic properties, the precipitation during growing season exerted the highest impact on the SOC balance.

4.1. Model parametrization and validation

The CANDY model performed well in predicting the SOC content, as shown by the error statistics for the parametrization and validation data. While the R^2 was fairly small (<0.5) in the GCEF and Jena parametrization datasets, likely due to a high variability of SOC measurements (see e.g. Refs. [78,119]) and a relatively short period of available measurements, for all data a value of >0.9 was achieved, attesting to a high reliability across wide pedoclimatic conditions and land use regimes. The RMSE was overall within a good range on the parametrization sites. Reference [81] reported similar RRMSEs (5.4–10.5%) for CANDY-based SOC simulations in Austria. The RMSEs of this study were also in accordance with values reported for prior CANDY-based simulations with the GCEF-dataset conducted by Refs. [74,76]. Therein, the RMSE ranged between 0.05 and 0.15 M-% SOC.

The R^2 for the validation sites confirmed the reliability of long-term simulations on a variety of climates, soil- and management types across Germany, despite RRMSEs up to 22% for sandy soils in the validation data and despite the number of available sites and

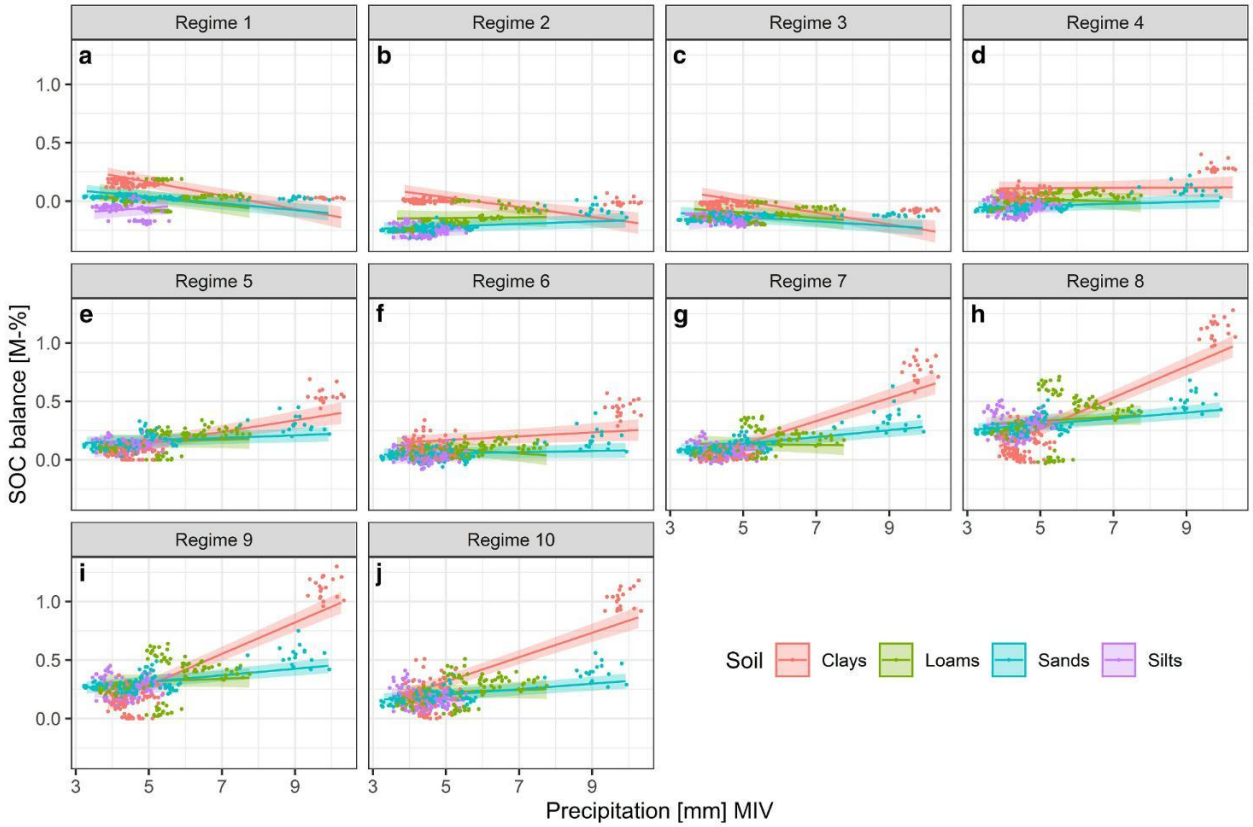

Fig. 6. Linear regressions (lines) and 95% confidence intervals (ribbons) of the effect of the mean intra-annual variability (MIV) of precipitation on the SOC balance [M-%] in dependence of the soil type and management regime according to the CM-MIV-1 model.

years of data across the three validation datasets varying highly (see data availability in the material and methods section). A high spatial and temporal variability in SOC measurements is well known to result in a relatively high error [31,120,121], particularly when considering the high fluctuations of measured SOC contents and the necessity to complete missing management-, soil- and climate data from third party sources on the validation sites. The single low R^2 on silts was likely a result of the relatively small range of measured SOC contents on the only two available sites in the dataset. Considering the overall good performance of the validation dataset ($R^2 >$ 0.8), the CANDY model was considered reliable for the evaluation of long-term future climate scenarios.

4.2. Scenario simulations

4.2.1. Importance of seasonal fluctuations

The multimodel inference approach and the investigation of spatial autocorrelation substantiated the importance of seasonal fluctuations of climate metrics (precipitation, air temperature, global radiation): The mean intra-annual variability (MIV), which reflects on the seasonality, was the most reliable aggregation method for the prediction of the SOC balance. Using annual sums and means (AAM) produced a higher empirical support judged solely by the AIC. However, only the MIV model successfully accounted for spatial autocorrelation, which was a central issue detected in our data, and furthermore displayed the best performance, as per the higher fixed R^2 and lower random R^2.

Changes of the seasonal weather regimes were therefore more relevant to the SOC sequestration than the mere increase or decrease of the annual mean of temperature and annual sum of precipitation [82]. Several studies also report the prevalence of seasonal effects and changes in temperature or precipitation regimes over the effects of uniform changes [43,110,122–125]. Particularly unusually long periods between rainfall events result in water stress, thus decreasing plant productivity and sequestration potential [125]. An increasing MIV coincides with higher seasonal amplitudes, and may therefore indicate stress caused by dryness, wetness, heat or cold [19,82]. However, in this study, precipitation aggregated by AAM and MIV was highly correlated (see supplementary material S8). This implies an increasing seasonality of precipitation with an increasing annual sum of rainfall. All but one site showed increasing precipitation under the RCP 2.6 scenario, with even higher increases under the RCP 8.5 scenario, in comparison to ambient climate (see supplementary material S1 and S3). Consequently, an increasing MIV of precipitation did not result in dry periods in our dataset, since in large parts of Germany the majority of rainfall occurs during the growing season (see e.g. Refs. [82,126]). Summer dryness is thus more likely to occur in regions in Germany with a low seasonality of precipitation. The relative importance of the variability of

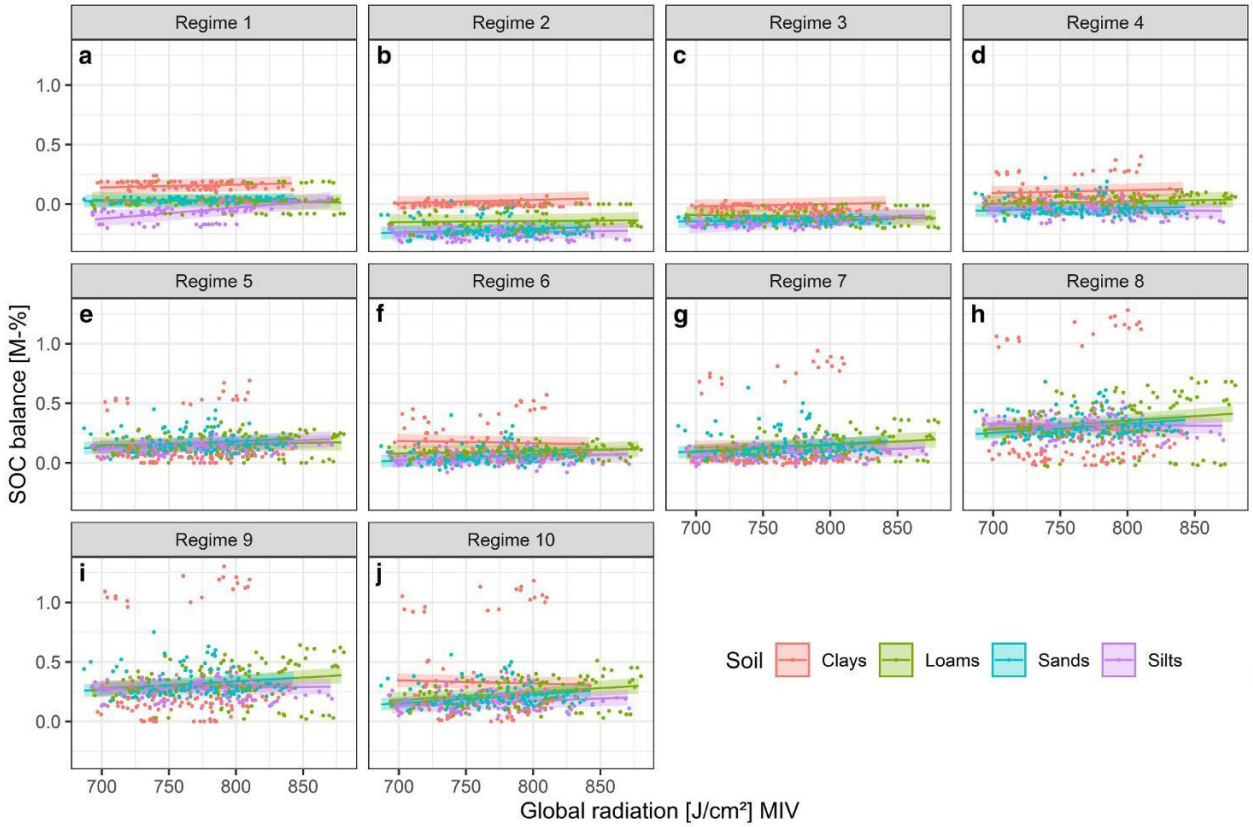

Fig. 7. Linear regressions (lines) and 95% confidence intervals (ribbons) of the effect of the mean intra-annual variability (MIV) of global radiation on the SOC balance [M-%] in dependence of the soil type and management regime according to the CM-MIV-1 model.

precipitation regimes may furthermore differ on wet, intermediate and dry sites [125]. In our study, the seasonal distribution of rainfall events was most impacting on sites experiencing excessive dryness. A low MIV of precipitation, and therefore its seasonality, by proxy indicates a risk of decreasing SOC. Accordingly, a simulation study by Ref. [49] using the PaSim model revealed that the net primary production was promoted by increased precipitation particularly at dry sites. Reference [124] also reason that the distribution and magnitude of rainfall events play a dominant role for the recovery of water contents in deeper horizons primarily in arid and semi-arid regions. Within the scope of this study however, it was not further possible to identify locations-specific differences regarding the relative importance of either MIV or AAM on the sequestration potential, which presents a future research topic.

4.2.2. Impact of changing precipitation and air temperature

Our results showed the highest climate-related variability of SOC sequestration due to changes in precipitation, whereas air temperature and particularly global radiation exerted a smaller impact. According to Ref. [19] soil moisture was the main factor governing sequestration in German soils. A dominance of mean precipitation over mean temperature on turnover rates was also documented by Ref. [10], with sequestration increasing from arid and cold to warm and humid climates. Hence, in our study the highest sequestration potential was ascertained in the southern region of Germany, which coincided with the highest precipitation rates (see also Fig. 4 and supplementary material S1 and S3). Similarly, reference [44] detected the highest sequestration rates in Europe in the alpine region, based on simulations with the ORCHIDEE-GM model. A review by Ref. [26] attributed correspondingly lower SOC accumulation rates to arid climates, and reference [127] reported that increasing altitude, and therewith precipitation, coincided with higher SOC contents. A similar effect was demonstrated by Ref. [128], who showed that topsoil SOC stocks increase with elevation and rainfall amounts. Nevertheless, increasing air temperature with future climate change may promote SOC sequestration [31]. In accord with our results, reference [129] conclude that temperate grasslands would likely become C sinks with future climate change as a result of increasing precipitation and temperature. Grasslands in Germany are predominantly situated in the alpine and low mountain regions and will therefore benefit from increasing precipitation and temperature: As an effect of a specific warming level above 1 ◦C in comparison to pre-industrial climate (1881–1910) reference [3] presented evidence suggesting that most German grassland soils would experience an increased net biome productivity and a longer growing season, presenting potential for SOC sequestration, though increasing temperature may result in increasing microbial activity, higher mineralization rates and therefore SOC losses [127]. At warming levels reaching 3.5 ◦C reference [3] reported that increasing droughts would shorten the vegetation period, turning former C sinks to sources. Favourable effects of increasing temperature are counteracted by reduced

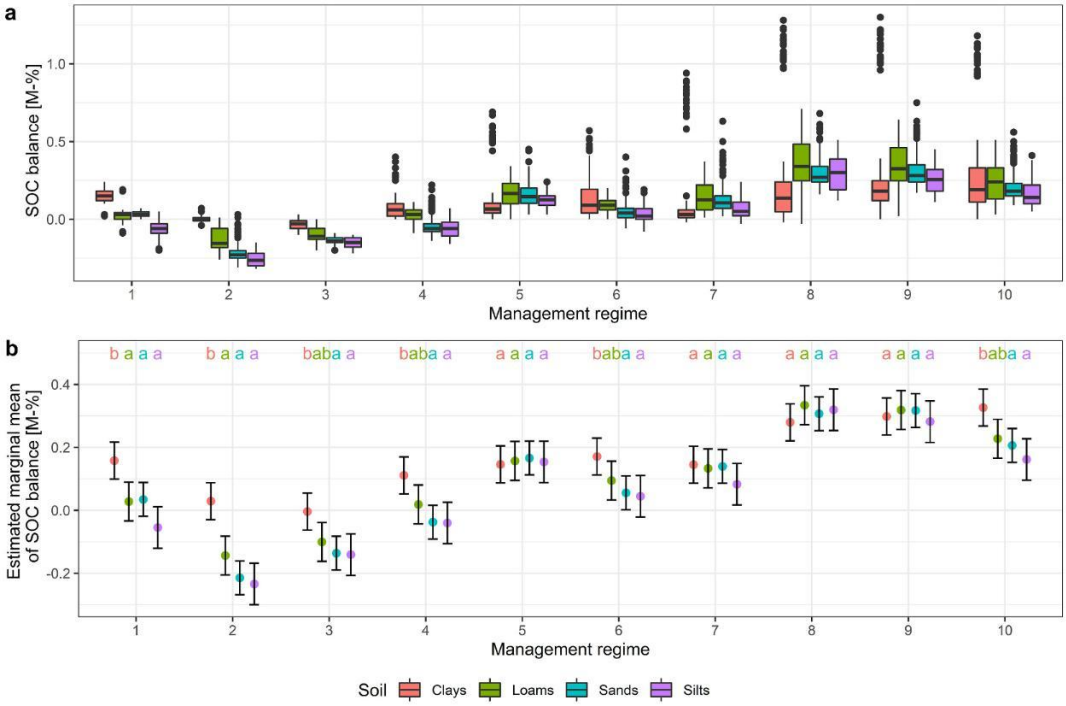

Fig. 8a: Simulated SOC balance [M-%] in the topsoil (0–30 cm) for the period 2020–2099 across 24 sites, grouped by soil group and management regimes. **Fig. 8b**: Estimated marginal means and 95% confidence intervals (error bars) obtained from the CM-MIV-1 model for the SOC balance, grouped by management regime and soil type. Post hoc test was applied within each management regime. Soil groups that do not share a letter differ significantly ($p < 0.05$).

precipitation particularly in the summer months [82]. This applies in particular for sites with already low summer precipitation. Positive feedbacks on the radiative forcing caused by anthropogenic climate change may further limit the terrestrial sequestration potential [130]. Reduced vegetation cover changes the surface albedo, causing higher amounts of radiation hitting the soil surface and increasing soil temperature, exacerbating already present dryness stress [82].

4.2.3. Interaction of climate and management practices

In our simulations, the effect of climate change on the SOC balance was largely driven by management practices. A prevalence of the effect of land use over the effect of climate was determined both by measurements [19] and by long-term scenario simulations [43, 50] and confirmed in a recent review [15]. However, as our simulations illustrated, due to the complex interaction of management and precipitation, simulation-based assessments are required to include a multitude of management- and climate regimes, as it is not possible to deduce ubiquitous rules with only one or few land use regimes. Reference [49] for instance found temperature and precipitation dominated the effect of management, which is in strong contrast to our results. In our study, management pronounced the climate-related differences in SOC sequestration between different sites. The study by Ref. [49] however included only two management regimes (cutting and grazing) on three sites with moderate to high precipitation (625, 995, 1033 mm), which simply may not suffice to explore the effect of varying management intensity, as illustrated by our results. Since the management regimes used in our study were created artificially and cover a wide spectrum from extensive to intensive management, based on information from long-term survey sites, the German fertilizer ordinance [63] and remote sensing data [dataset] [64–66], uncertainties largely stem from the application of future climate datasets. Reference [56] further substantiated this, arguing that high uncertainties may arise as an effect of simulations with a single climate scenario. Our study accounted for this by the application of three contrasting climate scenarios, each realized by six different climate projections [57,68–71,93]. As evident by the relatively small effect sizes of changing climate metrics in comparison to the effects of management (see Figs. 5–7) as well as small differences between climate projections (i.e. climate models that realize a given scenario) within each scenario (see supplementary material S12 and S13), the uncertainty associated with predicting the impacts of future climate scenarios is small. We therefore consider our results robust to the assumptions underlying our simulations (i.e. future climate change). As a caveat though, the applied RCP scenarios merely quantify the future radiative forcing and the resulting change in air temperature, precipitation and global radiation [57]. They represent a best case, worst case and middle of the road scenario of the efforts undertaken to mitigate future climate change. The exact emissions and therefrom resulting changes in the radiative forcing, however, are subject to various and complex socio-economic and political processes. These processes in turn need to be explored with the aid of the recent SSP scenarios [92] in order to guide decision making in policies aiming to mitigate climate change.

With increasing management intensity (up to five cuts), all scenario sites became C sinks on average. Grassland management

practices that increase yield, such as increasing cutting frequency, organic and mineral fertilization and grazing intensification, generally also promote SOC sequestration by vegetation growth and rhizodeposition [29–31], thus explaining higher sequestration rates under a four- and five-cut regime in our simulations. The beneficial effect of a higher management intensity prevails at increasing precipitation, since additional vegetation regrowth due to cutting requires additional water input. Considering precipitation increases as an effect of climate change in our simulation sites (see supplementary material S3), a moderate intensification of extensively used grassland, as investigated in our scenario simulations, is an adequate method to increase SOC sequestration [21,31,32]. An extensive use with two or less cuts per year (regimes 1 to 4) in turn is detrimental to long-term sequestration. Nevertheless, an increasing cutting frequency typically requires additional N-fertilizer input and more machine traffic, which may result in higher GHG emissions [3,44]. The production and application of mineral N fertilizer furthermore causes high CO_2 emissions, requiring life cycle assessment (e.g. Ref. [131]) in order to gauge the benefits and risks. Increasing SOC sequestration at the cost of increased N emissions is highly questionable, as N_2O, which is a by-product of denitrification, has the 296-fold global warming potential of CO_2 over a period of 100 years and is therefore a high impact for global warming [4,132,133]. Intensification of land use may furthermore be in conflict with the preservation of biodiversity [115,134–137], requiring a thorough consideration of a landscapes potential for achieving the respective goals at the farm scale.

4.2.4. Varying soil type related responses to climate and management

According to the meta-analysis by Ref. [129] and the review by Ref. [138], different soil types respond differently to the effects of climate and management. But in our study sands, silts and loams displayed a similar response to management on average, since the differences between the three soil types under the same management were not significant. Clays in turn showed the highest SOC gains on average. According to the CANDY-simulations, clays in Germany may accumulate up to 1 M-% SOC across the period of 80 years under four cuts (regimes 8 and 9) annually. The effects of land use generally predominate pedogenetic properties [27]. In general, to allow carbon sequestration enough mineral surface is required [139]. This limits the potential of coarse soils even under optimal management and explains higher SOC gains in clay soils. An increase of the cutting frequency at increasing MIV of temperature resulted in a lower SOC balance on most soil types except clays. Differences due to temperature between soil types furthermore largely persisted at higher management intensity (four to five cuts), with silts being most susceptible to temperature increases.

However, our analysis also revealed that the potential for SOC sequestration of clays can be negated under certain conditions: With an increasing MIV of precipitation, i.e. higher summer precipitation, clays displayed the highest potential for both C losses and gains, depending on management. A positive SOC balance was achieved with two or more cuts annually, while extensification resulted in losses. Under increasing precipitation, clays are therefore high-risk/high-gain-sites. An interpretation of the effect of precipitation on silts is hardly viable due to their prevalence in dry regions, therefore occupying only a small spectrum of the overall investigated precipitation pattern. However, according to the estimated marginal means, silts will gain SOC as a result of increasing precipitation with future climate change. The relatively minor changes due to changing precipitation patterns across different management regimes on loams and sands imply a lower response to climate change effects, though increasing the number of cuts also promoted sequestration with higher precipitation on these two soil types, albeit to a lesser extent than on clays. The smaller apparent effect is likely due to high infiltration rates and relatively low water storage capacity [139,140], which reduces the beneficial effect of higher precipitation. Although to a lesser extent, sandy soils benefited from increasing precipitation under future climate in our simulations. However, sands do not constitute typical grassland soils due to their overall low water storage abilities.

5. Conclusion

Using a broad dataset consisting of 24 representative grassland sites across Germany, ten management regimes applied to all sites, and 18 site-specific future climate datasets, we investigated the combined effects these properties exert on the future SOC sequestration potential of grasslands. Our results showed that changes in seasonal weather patterns were a more reliable predictor for the SOC sequestration than their absolute means. Precipitation, which increased predominantly at all sites and in all climate scenarios, was more relevant to the SOC sequestration in our survey region than temperature or global radiation. However, in contrast to many reports, an increasing seasonality of rainfall (i.e. higher MIV) did not reduce SOC in our dataset, since the majority of rainfall in Germany occurs during the growing season. Consequently, increasing seasonality promoted SOC sequestration.

The effect of management was dominant over the effect soil type and amplified climate-driven differences between soils. Averaged across all climate scenarios, under an extensive use (two or fewer cuts) all four soil types (sands, silts, clays, loams) experienced minor changes or even SOC losses. At higher management intensities (four to five cuts per year) all soil types increased in SOC. A moderate intensification (three or more cuts, with an optimum at four cuts) therefore promoted sequestration in our representative sites, though the gain of increased sequestration at a higher risk of N emissions due to higher required fertilization rates must be carefully considered. Overall, clay soils displayed the highest potential for long-term SOC accumulation across all climate regimes and land uses. However, under extensive use (two or fewer cuts) in combination of increasing precipitation, the potential for SOC losses was also highest on clays as a result of SOC mineralization surpassing sequestration. Loams and sands in turn displayed the least changes due to varying seasonality of precipitation under all management regimes as an effect of a low particle surface limiting sequestration. Increasing management intensity promoted SOC sequestration particularly at higher precipitation. In our dataset, the highest sequestration rates were thus determined in Southern Germany, which coincides with the highest precipitation amounts. Our study showed that substantial SOC gains can be expected under grassland under adequate management over the next decades. However, due to the long time until SOC changes are measurable, management practices that aim at fostering SOC sequestration need to be implemented early in order to achieve the desired sequestration effect.

For future studies scenario simulations should include a multitude of management regimes, since the effect that climate, particularly precipitation, exerts on SOC varies highly with management intensity. Simulations including management regimes that merely represent two extremes may be insufficient to detect those interdependencies. In conclusion, particularly decreasing summer precipitation is a central factor threatening future SOC sequestration, though the majority of the representative grassland sites selected for our study will experience increasing summer precipitation. Lower summer rainfall can partly be counteracted by management practices, though there is a demand to quantify the locations-specific relative importance of seasonality and absolute means of climate metrics on the sequestration potential. As a result of increasing seasonality of precipitation, grassland management is recommended on the representative sites under future climate change, since 1) the permanent vegetation provides a protective cover against erosion, 2) increasing precipitation amounts aid the recovery of soil water and 3) grassland management with three to five cuts annually contributes to the mitigation of climate change via increased SOC sequestration.

Author contribution statement

Matthias Robert Filipiak: Conceived and designed the experiments; Performed the experiments; Analyzed and interpreted the data; Contributed reagents, materials, analysis tools or data; Wrote the paper.

Doreen Gabriel: Analyzed and interpreted the data; Contributed reagents, materials, analysis tools or data; Wrote the paper.

Katrin Kuka: Conceived and designed the experiments; Analyzed and interpreted the data; Contributed reagents, materials, analysis tools or data; Wrote the paper.

Data availability statement

Data included in article/supplementary material/referenced in article.

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Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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4. Assessment of the carbon sequestration potential in temperate grasslands of the Hainich Biodiversity Exploratory in dependence of future climate and management scenarios

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Highlights

- According to long-term simulation models, management practices have a more significant impact on soil organic carbon (SOC) sequestration than soil texture and climate.
- The models indicate that moderate intensification enhances SOC sequestration.
- Model projections suggest that grazing is generally more effective than mowing in promoting SOC sequestration.
- Based on the models, grasslands in the Hainich region are expected to see slight improvements in SOC sequestration due to projected future climate changes.
- The simulation models predict that SOC sequestration rates progressively increase across different texture classes, with the lowest rates in silts, followed by loams, and the highest in clays.
- Model outcomes suggest that soil compaction is likely to reduce SOC sequestration rates.

Abstract

Soil organic carbon (SOC) sequestration is crucial for mitigating climate change and regulating soil functions. Temperate permanent grasslands thereby have a high potential to sequester SOC due to an extended growing season under future climate conditions. Using the CANDY model, this study evaluates the SOC sequestration potential of 50 grassland sites in the Hainich Biodiversity Exploratory by subjecting each site to six contrasting management options under 18 future climate datasets spanning a period from 2008 to 2100. Simulations indicate that increasing management intensity through higher livestock units (up to 4 LU ha^{-1}) or high mowing frequency (up to 5 cuts per year) leads to substantial SOC gains, especially on less intensively used sites. However, beyond a certain intensity threshold, the benefits of intensification diminish. This reduction in effectiveness may be due to factors such as increased water demand, soil compaction and disruption of nutrient cycles. Moderate grazing intensification (2 – 3 LU ha^{-1}) and moderate mowing intensification (up to 3 – 4 cuts per year) are recommended in the Hainich region to maximise SOC sequestration while minimising negative impacts and maintaining resilience to future climate change. Soil compaction was identified as a central threat to future SOC sequestration, particularly under high management intensity, whereby soils with clay contents >15 % were most susceptible. Continuous monitoring and adaptive management are essential for sustainable SOC sequestration. This study highlights the need for balanced management strategies to enhance SOC sequestration in temperate grasslands. The CANDY model's findings align with broader literature, underscoring the importance of integrated, sustainable land management to achieve long-term climate mitigation and soil health goals.

4. Assessment of the carbon sequestration potential in temperate grasslands of the Hainich Biodiversity Exploratory in dependence of future climate and management scenarios

Keywords:

Soil organic carbon; sustainable agriculture; climate change mitigation; soil functions; grassland; future scenarios; management practices

Abbreviations

AIC: Akaike information criterion; BAT: biologic active time; BD: bulk density; BÜK200: national soil map of Germany (Bodenübersichtskarte) in scale of 1:200.000; C: carbon; CAN: calcium ammonium nitrate; CANDY: Carbon And Nitrogen Dynamics; CDC: climate data centre; CM: candidate model; dAIC: delta of the AIC; DM: dry matter; DüV: German fertilisation ordinance (Düngeverordnung); DWD: German meteorological service (Deutscher Wetterdienst); FC: field capacity; FM: fresh matter; G1, G2 and G3: extensive, moderate intensity and intensive grazing respectively (management regimes); GHG: greenhouse gas; GM: global model; Hc: heat capacity; IPCC: Intergovernmental Panel on Climate Change; KA5: German soil classification guideline; Ks: saturated hydraulic conductivity; LU: livestock units; LUI: Land Use Intensity; M1, M2 and M3: extensive, moderate intensity and intensive mowing respectively (management regimes); N: nitrogen; OM: organic matter; PD: particle density; PTF: pedotransfer function; PWP: permanent wilting point; RCP: representative climate pathways; RMSE: root mean square error; RRMSE: relative RMSE; R^2: coefficient of determination; SAC: spatial auto-correlation; SOC: soil organic carbon; SOM: soil organic matter; VIF: variance inflation factor.

4.1. Introduction

Temperate grasslands provide critical ecosystem services such as maintaining biodiversity, water retention, nutrient cycling regulation, nitrogen (N) emission reduction, and soil organic carbon (SOC) sequestration [1–6]. Long-term SOC sequestration in grasslands is consistent with global and national climate change mitigation policies [7–10]. However, balancing these goals with the need for productivity for feed production, especially in biodiversity-maintaining regions, is challenging [11–19]. Climate change thereby presents both opportunities and risks, affecting grassland regions in Germany with warmer temperatures, longer growing seasons, and shifts in precipitation [16, 20–22].

Given the changing climate and therefrom resulting challenges, it is critical to identify grassland management strategies that ensure high yields and sequestration rates. Quantifying SOC changes under different pedoclimatic conditions is essential for understanding carbon (C) sources and sinks to support sustainable land use strategies [23–25]. However, measurable changes in SOC typically require long-term field experiments that account for uncertainties associated with spatial heterogeneity and sampling issues [22, 26–28].

4. Assessment of the carbon sequestration potential in temperate grasslands of the Hainich Biodiversity Exploratory in dependence of future climate and management scenarios

Process-oriented simulation models provide a way to conduct long-term scenario simulations under different conditions [16, 21, 26, 29–32]. However, modelling studies are often limited by relatively short simulation times or insufficient consideration of sites, climate scenarios, or management regimes [21, 33–39].

In the present study, an attempt was made to counter these limitations of earlier simulation studies by conducting long-term SOC simulations with the CANDY model [40, 41]. Validations of the model at various long-term field experiments and study sites support its reliability [41–47]. First, using data from 50 grassland sites, the model was validated under recent climate and land use conditions. We then developed future management regimes representing an intensification gradient and applied them under different climate scenarios on each site. This research aims to provide management recommendations for Central European grasslands, taking into account resilience to climate change and SOC sequestration.

Our research is driven by the following hypotheses:

1. SOC sequestration will be influenced by management intensity.
2. SOC sequestration rates will differ between grassland management types (grazing and mowing).
3. Increased growing season precipitation under future climate scenarios will enhance SOC sequestration under future conditions.
4. The responses of SOC sequestration rates will differ by soil texture classes (clay, loam, silt).

4.2. Materials and methods

Site description

The SOC simulations were based on data from 50 plots in the Hainich Biodiversity Exploratory, a typical middle-elevation grassland region in central Germany [48, 49]. Established in 2006, the Biodiversity Exploratories investigate the effects of land use intensity and change on biodiversity, soil properties and ecosystem processes [49]. The Hainich Biodiversity Exploratory is located in northwest Thuringia around the city of Mühlhausen [48]. During the ambient simulation period (2008 – 2017), mean annual air temperature was 8.8 °C and mean annual precipitation was 537 mm across all 50 sites. The sites are Cambisols (n = 28), Stagnosols (n = 18), and Vertisols (n = 4) developed from limestone with varying degrees of loess cover (Figure 1) [48].

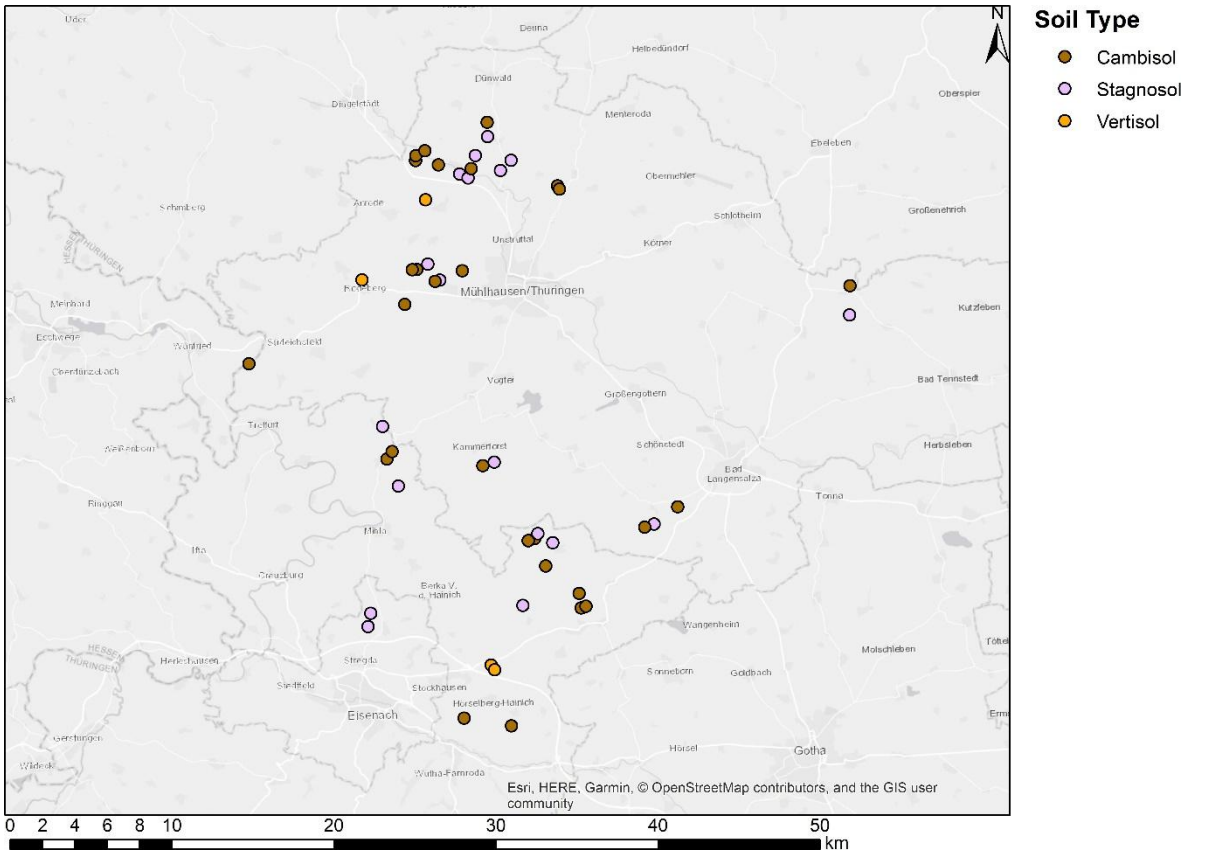

Figure 1: Soil type [113] and distribution of the 50 grassland sites around the city Mühlhausen (Thuringia).

The grassland sites are continuously managed in a practice-oriented manner by their landowners or tenants as meadows, pastures or mown pastures, receiving mineral or organic fertilisation, both, or none at all [48–51]. Detailed information on experimental design and management can be found in publications by [49], [51] and on the biodiversity exploratories website [48]. The site-wise varying land use intensity in Hainich is quantified using the land use intensity index (LUI), which standardises and aggregates mowing frequency (LUI mowing), stocking intensity (LUI grazing), and fertiliser application rates (LUI fertiliser) into one dimensionless value [50]. The LUI indices for the 2008 – 2017 period were calculated according to [50], based on information from the land owners and tenants on mowing, grazing and fertilisation [51] using the LUI calculation tool [52] implemented in the Biodiversity Exploratories Information System (BExIS) [53].

Description of the CANDY model and required input data

The simulations were performed with CANDY (Carbon and Nitrogen Dynamics, version 3.20.17.40) [40, 41], a one-dimensional dynamic soil model. CANDY simulates C, N, water, and temperature turnover, dynamics, and fluxes in mineral agricultural soils in up to daily resolution. This study used a modified

CANDY parametrisation with a transpiration coefficient of 8.0 kg mm^{-1}, developed in a previous national-scale simulation study [45]. The model incorporates the concept of Biological Active Time (BAT) to quantify site-specific influences on SOC turnover conditions, including soil temperature, moisture, aeration, and texture [54], which impacts processes in the soil and their dynamics. CANDY conceptualises soil organic matter (SOM) in fresh, active, stabilised, and long-term stabilised pools, making it suitable for long-term SOC simulations across diverse sites and conditions without requiring a spin-up period. The calculation of the long-term stabilised SOM pool follows the "particle surface" approach of [55] and [56].

CANDY requires soil, management and meteorological data for the SOC turnover simulation. Data were primarily obtained from the BExIS database [53], supplemented in some cases by secondary data sources to fill gaps, as outlined in the following subsections.

Meteorological data

CANDY requires daily values of mean air temperature at 2 m, precipitation sum, and either global radiation or sunshine duration. Different datasets were used for the ambient and future simulations.

Ambient data

For the ambient period, original measurements (ID 24766, [57]) were used, supplemented by data from nearby weather stations of the German National Meteorological Service (DWD Climate Data Centre) [58–61]. Specifically, stations 6305 and 5593 provided missing precipitation and air temperature data, while stations 2925 and 7368 contributed sunshine duration data.

Future climate scenarios

For future period, three Representative Climate Pathway (RCP) scenarios from the IPCC Fifth Assessment Report [30] were used. These scenarios quantify increases in radiative forcing of 2.6, 4.5, and 8.5 W m^{-2} by the year 2100, corresponding to 490, 650, and 1370 ppm CO_2 equivalents, respectively. To address uncertainties, projections from six independent climate models were used, corresponding to 18 datasets, which are representative of the Hainich Biodiversity Exploratory at long. 10.4978 and lat. 51.2061 (see Additional file 1, supplementary material 1). More details on the RCP scenarios can be found in [62].

Management data

The required management data include the start and end of grazing periods, stocking rates, dates of fertiliser application, type and amount of fertiliser, and harvest dates and corresponding yields.

4. Assessment of the carbon sequestration potential in temperate grasslands of the Hainich Biodiversity Exploratory in dependence of future climate and management scenarios

Ambient management

For the ambient period [51] provided the majority of data, which was supplemented by long-term averages for missing management actions. To address limitations in available grazing data, the start and end of each grazing period was standardised to the first and last day of a given month to ensure consistent calculations for grazing duration and stocking rates. In cases where only annual fertiliser application rates were available instead of event-based application rates, two-thirds of the annual input were allocated to the first fertilisation event, with the remainder allocated evenly to subsequent events. Organic fertilisers as required in the CDYOPSPA table of the CANDY database were parameterised using data from the Lower Saxony Chamber of Agriculture [63] and the Thuringian State Institute for Agriculture [64] (see Additional file 1, supplementary material 2).

Future management regimes

Future management regimes were derived from actual management data from the Hainich Biodiversity Exploratory [51] and classified into each three regimes for both mown (M1, M2, M3) and grazed (G1, G2, G3) grasslands, representing a gradient of intensification. A summary of the management regimes and operations is provided in Table 1. The yields of the M1 (one cut per year) and M2 (three cuts per year) regimes correspond to the rounded reported average yield for sites in Hainich with one and three cuts per year on average respectively. The yield of the M3 regime (five cuts per year) was determined by consulting the German fertilisation ordinance (Düngeverordnung – DüV) [65] due to a lack of actual management data for a five-cute use in the Hainich sites. Thereby, the expected yield for a five-cut use according to [65] was reduced by 10 dt ha^{-1} to accommodate the lower average yields in Hainich (see Additional file 1, supplementary material 3). Grazing regimes were created from all pasture sites (i.e. mown less than once per year on average) and used the average reported start and end dates of the grazing period. Stocking rates correspond to the rounded minimum (G1), mean (G2) and maximum (G3) of reported stocking rates (see Additional file 1, supplementary material 4 and 5). Fertiliser inputs followed recommendations from [65] as well as [66] for extensive, moderate and intensive grazing. Cattle slurry was selected as the organic fertiliser due to the larger quantities available in this region.

Table 1: Description of management regimes developed for future simulations.

Name	Type	LUI	Operation	Quantity and type	Date
M1	1 cut	0.99	Organic fertilisation Mowing	41 kg N via 89 dt FM ha^{-1} cattle slurry 30 dt DM ha^{-1}	01.04. 01.08.

M2	3 cuts	1.81	Organic fertilisation Mowing Organic fertilisation Mowing Organic fertilisation Mowing	110 kg N via 239 dt FM ha^{-1} cattle slurry 35 dt DM ha^{-1} 28 kg N via 61 dt FM ha^{-1} cattle slurry 17.5 dt DM ha^{-1} 28 kg N ha^{-1} via 61 dt FM ha^{-1} cattle slurry 17.5 dt DM ha^{-1}	15.03. 01.05. 15.05. 01.07. 15.07. 01.09.
M3	5 cuts	2.35	Organic fertilisation Mineral fertilisation Mowing Mineral fertilisation Mowing Mineral fertilisation Mowing Mineral fertilisation Mowing Mineral fertilisation Mowing	170 kg N via 370 dt FM ha^{-1} cattle slurry 18 kg N ha^{-1} via CAN 50 dt DM ha^{-1} 23.5 kg N ha^{-1} via CAN 12.5 dt DM ha^{-1} 23.5 kg N ha^{-1} via CAN 12.5 dt DM ha^{-1} 23.5 kg N ha^{-1} via CAN 12.5 dt DM ha^{-1} 23.5 kg N ha^{-1} via CAN 12.5 dt DM ha^{-1}	01.03. 01.03. 01.04. 15.04. 15.05. 01.06. 01.07. 15.07. 15.08. 01.09. 15.10.
G1	extensive grazing	1.17	Organic fertilisation Start grazing End grazing	65 kg N via 141 dt FM ha^{-1} cattle slurry 1 LU ha^{-1} 1 LU ha^{-1}	01.04. 01.06. 30.09.
G2	moderate grazing	1.56	Organic fertilisation Start grazing End grazing	95 via 207 dt FM ha^{-1} cattle slurry 2 LU ha^{-1} 2 LU ha^{-1}	01.04. 01.06. 30.09.
G3	intensive grazing	2.10	Organic fertilisation Start grazing End grazing	130 kg N via 283 dt FM ha^{-1} cattle slurry 4 LU ha^{-1} 4 LU ha^{-1}	01.04. 01.06. 30.09.

Soil properties

CANDY requires detailed information for each homogeneous horizon of a soil profile up to a depth of 200 cm. This includes the texture (silt and clay content), bulk density (BD), particle density (PD), water content at field capacity (FC), water content at permanent wilting point (PWP), heat capacity of the soil (Hc), saturated hydraulic conductivity (Ks), initial SOC contents, and optionally the rock content. In addition, a time series of topsoil SOC measurements is required for validation.

Soil data on physical properties obtained from the BExIS database [53] encompassed topsoil measurements of PD (ID 16566, [67]), BD (ID 17086, [68]; ID 20266, [69]), and texture (ID 14686, [70]). In the absence of measurements of PD in the subsoil, a default value of 2.6 g cm^{-3} was assumed due to the prevalence of quartz [71–74]. Subsoil BD and texture, rock contents, and horizon boundaries were extracted from the BÜK200 digital soil map of Germany (Bodenübersichtskarte 200) [75]. Based

on the range classes provided by the BÜK200 according to the German soil classification guideline KA5 [76] median values of soil properties were used. FC, PWP and Ks were calculated using pedotransfer functions (PTFs) integrated into CANDY, and a default value of 0.16 J cm^{-3} K^{-1} was assumed for Hc [40, 41, 77]. The earliest available SOC measurements from the BExIS database [53], in most cases from the years 2008 and 2009, were used for profile parameterisation, and the full time series of topsoil SOC measurements (0 – 10 cm) from 2008 to 2017 were used for validation (ID 6223, [78]; ID 14446, [79]; ID 18787, [80]; ID 23846, [81]).

Model set up

Model setup includes optimising the initial SOC content, specifying the length of the growing season, and setting model run conditions such as the initialisation period and transition from ambient to future simulations.

Initial SOC content

Measured SOC values are often unavailable for the start date of simulations (see "Initialisation period and transition to future simulation"), as was the case in the Hainich Biodiversity Exploratory, and may have a high uncertainty due to spatial heterogeneity and sampling errors [82, 83]. To address this, optimised initial values for the ambient period were obtained using the CANDY Optimiser version 20.15.2.2 [84]. The optimiser adapts initial SOC values through up to 50 simulation runs by minimising the root mean square error (RMSE) between simulated and measured values. This iterative process increased the reliability of the results and enabled starting simulations with an initialisation period prior to the first available measurements and thus prior to the validation period [85].

Duration of the vegetation period

The length of the growing season in CANDY, which affects nutrient uptake, water uptake, and rhizodeposition, is an important model parameter. This value, which is calibrated to local specifications to ensure more reliable results, is typically established once at the beginning of the simulation period. It was set to 219 days in this study, representing the average duration calculated across all 50 grassland sites in the Hainich region for the first year of the ambient period.

Initialisation period and transition to future simulation

The simulations started with an initialisation period from 01.09.2005 to 31.12.2007 in order to establish the grassland vegetation within the model, followed by the ambient period, for which SOC measurements were available, starting on 01.01.2008 and ending on 31.12.2017. Future simulations started on 01.01.2018 using the last simulated SOC values from the ambient period as initial values,

and ended on 31.12.2100. SOC simulation results were generated in annual resolution, and additionally for all days in the ambient period with available SOC measurements. Each of the 50 grassland sites underwent 108 individual simulations in the future period, combining 18 climate scenarios and six management regimes. All SOC simulations for both ambient and future conditions were performed in the topsoil layer (0 – 10 cm) due to availability of the SOC measurements required for validation.

Statistical analysis

All statistical analyses were performed using the freeware and open source software R version 4.0.2 and R Studio version 1.3.1073 [86]. Data visualisation was performed using the packages *ggplot2* [87], *ggpubr* [88], *data.table* [89] and *corrplot* [90].

Validation

The *modeval* function from the *sirad* package [91] was used to calculate the RMSE, RRMSE, and R^2 to validate the parametrisation at a regional scale by comparing the simulated SOC from the ambient period (2008 – 2017) at all Hainich grassland sites with periodic measurements of SOC in the 0 to 10 cm layer. These metrics provided insight into the absolute fit, relative fit, and explained variance of the simulated values [92–94]. Further assessments were performed separately for each texture class (clay, loam, silt). Additionally, the simulated SOC balance across the ambient period was compared with the balance estimated by a linear regression across available measured SOC contents from the ambient period (2008 – 2017).

Assessment of SOC balance

The SOC balance for the ambient and future periods was evaluated with the multi-model inference approach, which ranks and compares competing statistical models to identify the most relevant predictors and interactions [95]. Global models (GMs) incorporating a preselection of predictor variables were fitted using the *lmer* function of the *nlme* package [96]. Subsequently, candidate models (CMs) representing subsets of the GMs with all possible combinations and interactions of predictors were fitted using the *dredge* function of the *MuMIn* package [97]. These CMs were then ranked using the Akaike Information Criterion (AIC) [98] to select the best model for identifying the essential drivers of future SOC sequestration.

Data preparation and predictor pre-selection

Before analysis, data were aggregated and pre-assessed to align with the criteria of the multi-model inference approach. The dependent variable was the SOC balance, representing changes in topsoil (0

– 10 cm) SOC contents from the beginning to the end of the relevant simulation periods (2008 – 2017 for the ambient period and 2018 – 2100 for the future). The initial SOC content at the start of the respective simulation period served as a predictor to account for variations across the 50 Hainich sites. Additional predictors from available meteorological, management, and soil data for the GMs were chosen based on the variance inflation factor (VIF), which indicates collinearity, using the *vif* function of the *car* package [99]. Only predictors with no or weak collinearity (VIF<2) were considered for further analysis.

For the ambient dataset, meteorological data were not included as predictors since the simulations were conducted with actual meteorological data and did not include scenario replications. Differences in management were represented by the mean and slope of the LUI index, which represent the average land use intensity of the ambient period and its long-term change respectively. Soil properties were represented by BD and clay content, identified as key factors influencing SOC turnover [27, 100–102].

For the future dataset, daily meteorological data in the form of six projections across three climate scenarios (Additional file 1, supplementary material 6) were aggregated as long term means, seasonality, and long-term change as outlined in detail in [45]. Aggregated values of air temperature and precipitation were selected as predictors following VIF guidelines and due to their reported impact on SOC turnover [16, 103–107]. Future management was represented as a factor variable, based on a high VIF (>5) for individual management actions. Unlike in the ambient dataset, the larger dataset size for the future period allowed grouping sites by soil texture classes (silts, clays, loams), facilitating a practical interpretation and derivation of recommendations based on established soil properties.

Model selection, diagnosis and interpretation

Based on predictor selection following the VIF criteria, the ambient GM displayed the following structure:

$$GM\_ambient = SOC\ balance \sim (initial\ SOC\ content + BD + clay\ content + LUI\ mean + LUI\ slope)\text{^}3 + random\ effect \quad (1)$$

Up to three-way-interactions were included in the ambient GM in order to identify complex interactions in the smaller ambient dataset.

The structure of the future GM in turn was as follows:

$$GM\_future = SOC\ balance \sim (initial\ SOC\ content + BD + texture\ class + management\ regime + air\ temperature + precipitation)\text{^}2 + random\ effect \quad (2)$$

The future GM was fitted with long-term averages of the annual precipitation sum and mean air temperature instead of their seasonality or long-term change since this resulted in a substantially lower AIC (dAIC>2).

In order to account for possible variance heterogeneity, multiple variants of the ambient and future GM were fitted using the *varIdent* function of the *nlme* package [96]. Weights applied to the variance structure of the ambient GM were soil texture class (3 levels), while the future GM additionally applied the management regime (6 levels), RCP scenario (3 levels) and the climate projection (6 levels). Implementing a variance structure did not improve the ambient GM, while the fit of the future GM was substantially improved (dAIC>2) by using the management regimes as weights.

Spatial autocorrelation (SAC) in residuals was tested for the ambient and future GMs at the site-level using the Moran's I at a $p<0.05$ and the *ape* package [108]. Since the Moran's I test revealed no sufficient evidence for SAC ($p>0.05$), no further adjustments were required.

After model selection, the adjusted R^2 was reported for the best three ambient and future CMs using the *rsq* package [109]. Model interpretation was performed with the overall best ambient and future CM (as per the lowest AIC) with no SAC in residuals, as determined by the Moran's I test [108]. The residuals of these models were plotted against the fitted values and visually examined for patterns and outliers.

Confidence intervals were calculated using the *effects* package [99]. The effect sizes, i.e. change in the SOC balance per one-unit-change of independent variables (e.g. BD, clay content, precipitation and temperature), were calculated using the *spread* function of the *tidyr* package [110] and the *effects* package [99]. The *emmeans* package [111] and the *multcomp* package [112] were used to perform post hoc tests at an alpha of 0.05 and identify significant differences in the SOC balance between different soil texture classes.

4.3. Results

Ambient period

Validation of the CANDY model and ambient SOC balance

Comparing the measured and the simulated SOC values of the ambient period (2008 – 2017), the CANDY model achieved an RRMSE of 14.83 and R^2 of 0.72 across all sites, whereby among the three soil texture classes silts performed best, followed by loams and clays (see Table 2 and Figure 2). The majority of the soils in the Hainich Biodiversity Exploratory are clays (42 sites), which is also reflected by the number of available measurements for validation (169). Loams (2 sites) and silts (6 sites) represent a minor fraction of the total available measurements (8 and 24 respectively).

Table 2: Model performance statistics for the ambient validation period (2008 – 2017) in the 50 Hainich sites.

Group		**RMSE [M-% SOC]**	**RRMSE [%]**	**R^2**	**sites**	**n**
Texture class (KA5)	Clays	0.68	15.37	0.62	42	169
	Loams	0.62	13.82	0.80	2	8
	Silts	0.50	11.00	0.93	6	24
All sites		0.66	14.83	0.72	50	201

Results were grouped by texture class according to the KA5 guideline [76], including number of sites with the corresponding soil texture class and the number of available measurements (n). Note that no sandy soils were present in the Hainich Biodiversity Exploratory.

On average, both the experimental (Figure 3a) and the simulated (Figure 3b) SOC contents followed similar trends across the ambient period, resulting in moderate to high SOC gains across varying management intensities. Disaggregating the sites by their soil type according to [113] shows largest increases on sites with a high management intensity (LUI mean >2) for both the experimental an simulated values on Cambisols and Stagnosols. A similar trend could not be determined on Vertisols since only four out of the 50 Hainich sites were located on Vertisols (see also Figure 1).

A detailed comparison reveals some differences between the simulated SOC balance on the one hand and the estimated experimental balance as per a linear regression across the measured SOC contents from 2008 – 2017 on the other hand. The highest simulated increase in SOC was +0.82 M-% from an initial SOC of 4.47 M-% on a site with a LUI mean of 2.7 and a BD of 0.68 g cm^{-3}. In contrast, the highest estimated experimental SOC gain was +3.40 M-% from an initial SOC of 3.96 on a site with a LUI mean of 1.2 and a BD of 0.43 g cm^{-3}. According to the simulations, a total of 13 sites, with initial SOC contents ranging between 2.48 and 5.81 M-%, experienced a low to moderate increase in the simulated SOC (<0.1 M-%) over the ambient period, while seven sites, with initial SOC contents between 2.99 and

5.01 M-%, experienced an SOC increase >0.5 M-%. As per the estimated balance from the measurements, two sites experienced minor increases (by 0.07 and 0.10 M-%) in the experimental SOC balance, starting from an initial SOC of 4.58 and 3.79 M-% respectively, while the remaining sites experienced measured SOC gains ranging between 0.1 and 3.4 M-%, with initial SOC contents ranging between 1.51 and 6.72 M-%. According to the CANDY simulations only one site experienced a minor decrease in the simulated SOC by 0.04 M-%, starting from the highest initial SOC level of 7.54 M-%, with a LUI of 0.9 and a BD of 0.87 g cm^{-3}. In contrast, according to the estimated balance from experimental data, the highest SOC loss (by 1.64 M-%) occurred on a site with an initial SOC content of 6.48 M-%, a LUI of 1.7 and a BD of 1.04 g cm^{-3} and twelve other sites experienced SOC losses between 0.02 and 0.70 M-% (with initial SOC contents between 3.00 and 5.68 M-%).

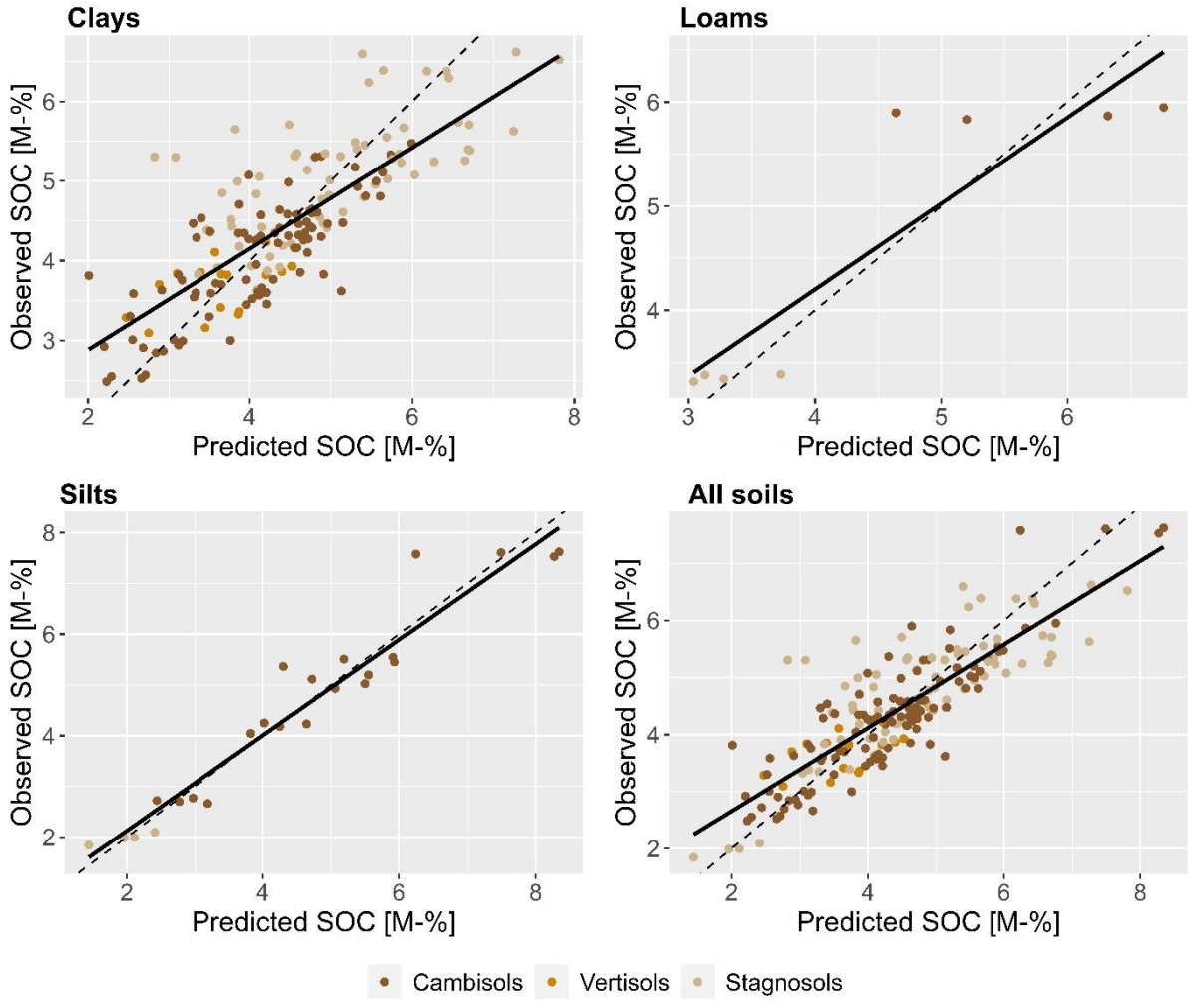

Figure 2: Observed and predicted ambient SOC contents in the topsoil of the 50 Hainich grassland sites.

The continuous line shows the fitted linear regression, the dashed line shows the theoretical ideal 1:1 fit. The data was grouped by soil texture class in accordance to [76], while the soil type according to [113] was visualised by colouring.

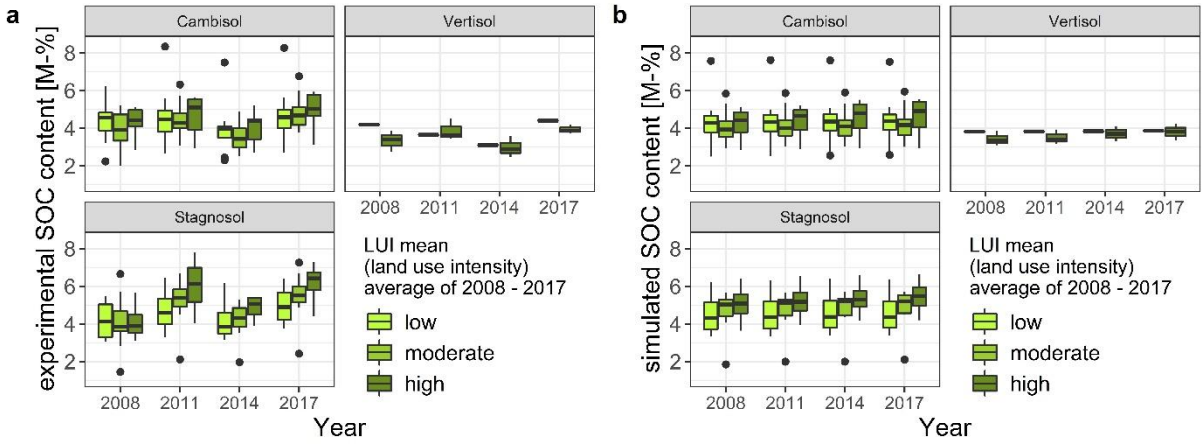

Figure 3: Experimental (3a) and simulated (3b) topsoil SOC content across the ambient period.

Simulated values were plotted only for years with available experimental values. The sites were grouped by the average land use intensity (LUI mean) across the ambient period into low (LUI mean <1), moderate (LUI mean 1 – 2) and high (LUI mean >2) as well as by their soil type into Cambisols, Stagnosols and Vertisols according to [113]. Note that due to the low number of sites on Vertisols (n = 4; see also Figure 1) the range of management intensity is correspondingly small for that soil type.

On a site-to-site comparison, the site with the highest BD (1.29 g cm^{-3}) had a simulated increase of 0.38 M-% and an experimental increase by 0.94 M-%. The site with the lowest BD of 0.43 g cm^{-3} had simulated SOC gain of 0.45 M-% and an experimental gain by 3.40 M-%, which was the highest experimentally determined gain. The highest mean LUI recorded was 3.14, with a simulated and experimental SOC increase of 0.29 M-% and 0.81 M-% respectively, while the lowest mean LUI value was 0.66, whereby SOC balance decreased by 0.20 according to experimental data and increased by 0.15 according to simulations.

Statistical model selection and diagnostics

Several competing statistical models with a preselection of predictors were fitted to the simulated ambient SOC balance in order to identify significant predictors as well as their effect sizes. This allowed identifying key causes for changes in the SOC balance and their relative contribution. The ambient GM, i.e. the model containing all predictors, and the corresponding three best CMs, containing a subset of predictors, performed substantially better (dAIC>2) than the null model M0 (see Table 3). The overall best ambient CM, as determined by the AIC, displayed a simple model structure. Significant predictors of the ambient SOC balance were two two-way interactions: the first between the LUI mean and LUI slope, and the second between BD and clay content. The interaction between the LUI mean and LUI

slope was also contained in the other two best ambient CMs. The ambient GM and the best three CMs displayed a similar goodness of fit as judged by the R^2. The Moran's I test revealed no strong evidence for spatial autocorrelation in residuals in the best CM model at $p<0.05$.

Table 3: Summary statistics of the model selection of the ambient period (2008 – 2017).

Model	Predictors	df	AIC	dAIC	R^2		
					total	fixed	random
GM	(SOCi + BD + clay + LUIm+ LUIs)^3	28	-24.5	20.4	0.93	0.71	0.22
CM 1	BD + clay + LUIm + LUIs + BD:clay + LUIm:LUIs	9	-45.0	0	0.94	0.53	0.41
CM 2	SOCi + BD + clay + LUIm + LUIs + BD:SOCi + LUIm:LUIs	10	-38.5	0.8	0.94	0.60	0.35
CM 3	SOCi + BD + clay + LUIm + LUIs + BD:LUIm + BD:SOCi + clay:LUIs + clay:SOCi + LUIm:LUIs + LUIs:SOCi + clay:LUIs:SOCi	15	-30.0	0.8	0.95	0.67	0.28
M0	-	3	32.1	24.6	0.88	0	0.88

The table contains the ambient global model (GM), the best three candidate models (CM), the ambient model designated to the LUI disaggregated by grazing, mowing and fertiliser (MLUI), and the null model (M0) for the ambient period, with initial SOC (SOCi), bulk density (BD), clay content (clay), mean LUI (LUIm), slope of LUI (LUIs), LUI grazing (LUIg), LUI mowing (LUIm) and LUI fertiliser (LUIf). Note that no model selection was applied to the MLUI model, thus AIC and dAIC are not applicable (n.a.).

Contribution of individual effects to the ambient SOC balance

The interaction between the LUI mean and the LUI slope extracted from the best ambient CM showed a positive response of the simulated SOC balance to increasing the LUI slope (corresponding to intensification) at a lower LUI mean level (<1.4) and a decreasing response to increasing the LUI slope at moderate and higher LUI mean levels (≥1.4). As such, increasing the LUI slope by one unit at a LUI mean of 0.5 increased the simulated SOC balance by 0.18 M-%. At a LUI mean of 1.0, the simulated SOC balance still increased by 0.07 M-% when the LUI slope increases by one unit. However, at a LUI mean of 2.5 increasing the LUI slope by one unit decreased the simulated SOC balance by 0.23 M-%. Nevertheless, despite the decreasing beneficial effect of the LUI slope at higher LUI mean levels, the simulated SOC balance remained overall positive (see Figure 4 and Additional file 1, supplementary material 7).

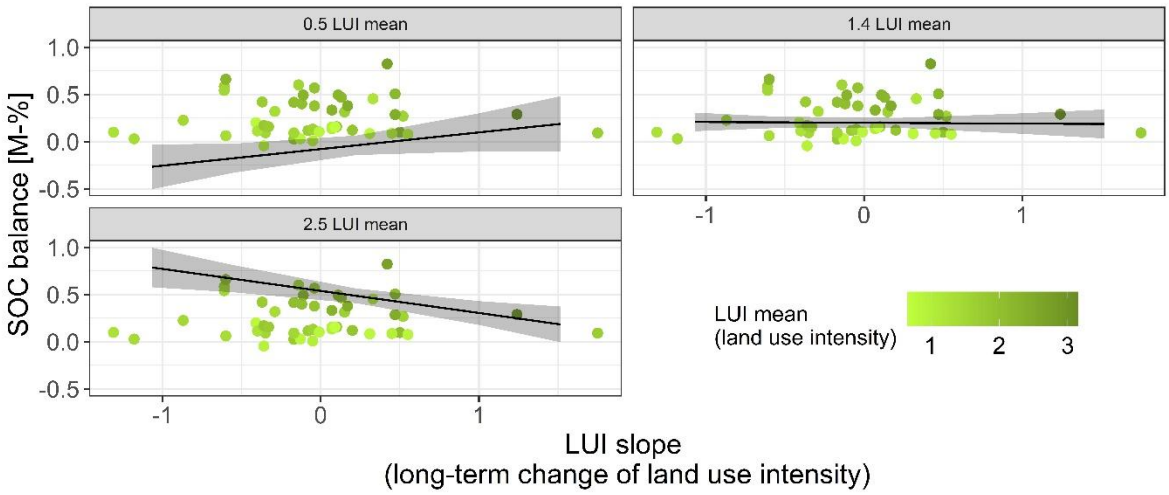

Figure 4: Effect of the LUI slope (intensification) on the topsoil SOC balance of the ambient period.

The estimated marginal means (regression line) and 95 % confidence interval (ribbon) of the LUI slope effect were obtained from the best ambient CM. The effect of the LUI slope was plotted across three representative levels of the LUI mean (0.5, 1.4, 2.5) to visualise the changing impact of further intensification across varying levels of mean management intensity across the ambient period. The SOC balance simulated with CANDY is additionally plotted (point features) for reference with the mean LUI as colour scale.

The second significant interaction in the best CM model showed a decreasing simulated SOC balance at higher soil clay content with increasing soil BD. At low clay contents increasing the BD by one unit increased the simulated SOC balance (by 0.25 at 5 M-% clay content). Changing the BD did not have an impact on the simulated SOC sequestration at moderate clay contents (15 M-%). However, especially at high clay contents the simulated SOC balance sharply decreased with the BD increasing by one unit (by 0.73 M-% at 50 M-%; see Figure 5 and Additional file 1, supplementary material 8).

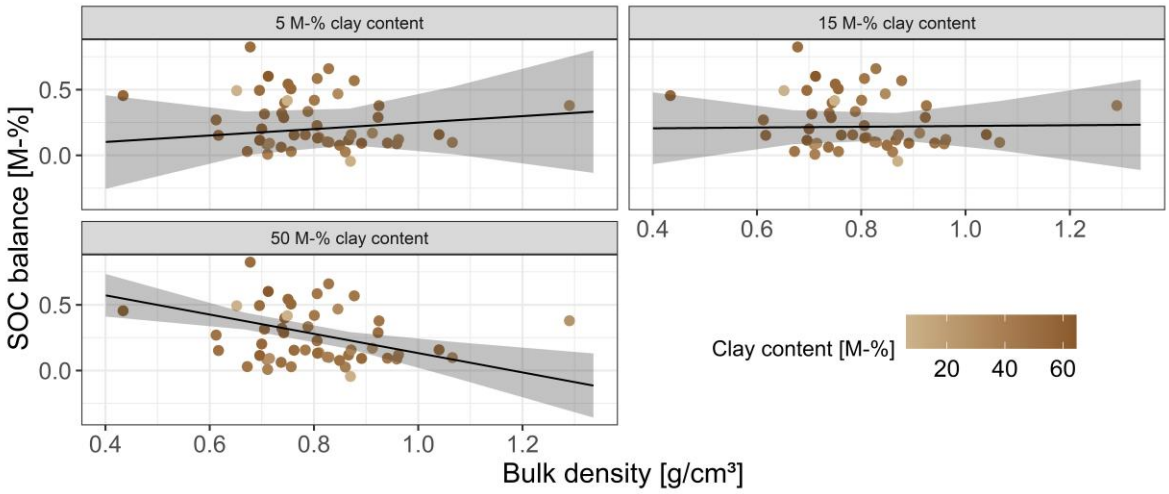

Figure 5: Effect changing bulk density on the topsoil SOC balance of the ambient period.

The estimated marginal means (regression line) and 95 % confidence interval (ribbon) of the effect of bulk density were obtained from the best ambient CM. The effect of increasing bulk density was plotted across three representative levels of the topsoil clay content (5, 15, 50 M-%) to visualise the changing impact of soil compaction in dependence of soil texture. The SOC balance simulated for all 50 sites with CANDY is additionally plotted (point features) for reference with clay content as colour scale.

Future period

Future SOC-balance

Figure 6 summarises the expected change in the simulated SOC balance under future climate change. Generally, averaged across all future climate scenarios and soil texture classes, the CANDY model predicted increasing SOC contents at higher management intensity, for both the mown and the grazed regimes. In a direct comparison, higher simulated SOC gains were thereby determined for the grazed regimes than for the mown regimes.

According to the CANDY model, across the entire future period (2018 – 2100) SOC losses at the Hainich sites under future scenarios are expected only under the M1 and M2 mowing regimes, which involve one and three cuts per year, respectively. Under these regimes, across the 83 years of the future period, the CANDY simulations showed the largest SOC losses on silty soils (-0.64 ±0.56 M-% for M1 and -0.75 ±0.60 M-% for M2), followed by loamy soils (-0. 45 ±0.25 M-% for M1 and -0.56 ±0.26 M-% for M2), and least on clayey soils (-0.07 ±0.14 M-% for M1 and -0.17 ±0.13 M-% for M2). Notably, simulated SOC losses on silty and loamy soils were significantly greater than those on clayey soils in both M1 and M2 regimes. In contrast, under the five-cut regime M3, all soil types showed a positive

simulated SOC balance (1.83 ±0.41 M-% on clays, 1.55 ±0.34 M-% on loams, 1.16 ±0.73 M-% on silts), whereby the simulated SOC balance of clays was significantly higher than that of loams and silts.

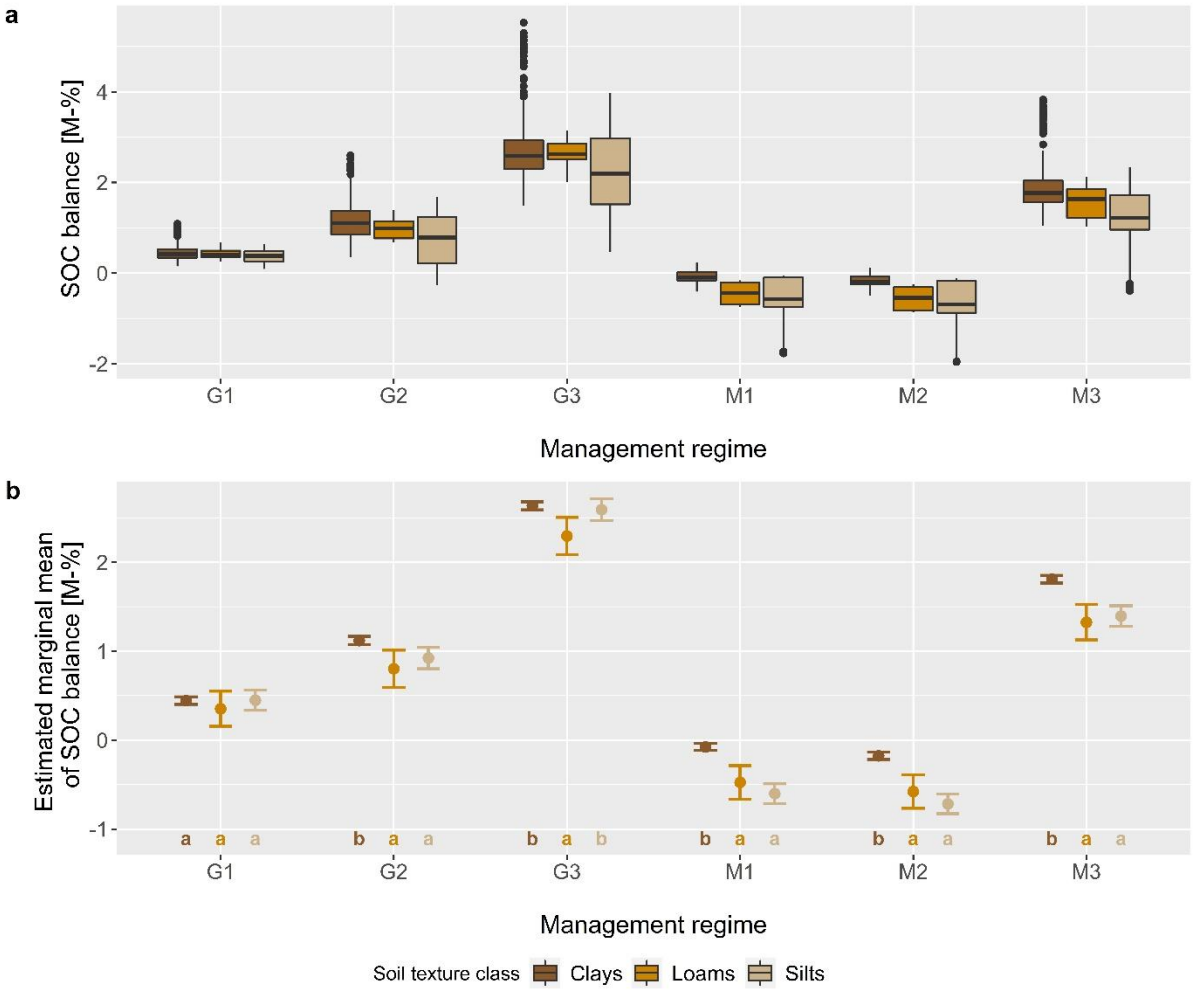

Figure 6: Simulated future SOC balance (6a) and estimated marginal means and 95 % confidence intervals (6b).

Results were grouped by texture class and management regimes (G: grazing; M: mowing; 1: extensive; 2: moderate; 3: intensive). Estimated marginal means and confidence intervals for the SOC balance were obtained from the best future CM. The post hoc test was applied within each management regime. Soil texture classes that do not share a letter differ significantly (p<0.05).

Positive SOC balances according to CANDY were determined for the grazing regimes on all intensity levels, whereby the SOC gains increased with an increasing stocking density (G1<G2<G3). The overall smallest simulated gains (0.45 ±0.15 on clays, 0.43 ±0.11 on loams and 0.37 ±0.13 M-% on silts) were

determined under the extensively grazed regime (G1), whereon silts, clays and loams were not significantly different. Moderate gains in the SOC balance were predicted under the moderate intensity grazing regime (G2). Thereby, clays had a significantly higher SOC gain (1.13 ±0.40 M-%) than loams (0.98 ±0.21 M-%) and silts (0.75 ±0.55). Nevertheless, the overall largest gains are expected to occur under the intensive grazing regime (G3), especially on clays (2.66 ±0.57 M-%) and loams (2.67 ±0.25 M-%), followed by silts (2.22 ±0.98 M-%).

Statistical model selection and diagnostics

Analogously to the ambient period, using a pre-selection of predictors fitted to the simulated future SOC balance allowed to identify significant predictors and their effect sizes. All models for future scenarios performed substantially better (dAIC>2) than the null model and displayed a good fit with a high R^2 (Table 4). The best future CM included all individual effects of the corresponding GM as well as a total of nine two-way interactions, which can be grouped into three categories: 1) the interaction between management regime on the one hand and soil texture class, BD, initial SOC content, precipitation and air temperature on the other; 2) the interaction between soil texture class on the one hand and initial SOC content, precipitation and air temperature on the other; and 3) the interaction between precipitation and initial SOC content. The second-best CM also included the interaction between BD and initial SOC content, while the third best CM included the interaction between BD and precipitation instead. The Moran's I test revealed no strong evidence for SAC in the residuals of the best CM at $p<0.05$.

Table 4: Summary statistics of the model selection of the future period (2018 – 2100).

Model	**Predictors**	**df**	**AIC**	**dAIC**	**R^2**		
					total	**fixed**	**random**
GM	(SOCi + BD + txc + mng + tmp + prc)^2	63	-8976.0	8.5	0.97	0.96	0.01
CM 1	prc + tmp + BD + txc + mng + SOCi + prc: txc + prc:mng + prc:SOCi + tmp: txc + tmp:mng + BD:mng + txc:mng + txc:SOCi + mng:SOCi	56	-8984.5	0	0.97	0.96	0.01
CM 2	prc + tmp + BD + txc + mng + SOCi + prc:txc + prc:mng + prc:SOCi + tmp:txc + tmp:mng +	57	-8983.3	1.2	0.97	0.96	0.01

	BD:mng + BD:SOCi + txc:mng + txc:SOCi + mng:SOCi						
CM 3	prc + tmp + BD + txc + mng + SOCi + prec:BD + prc:txc + prc:mng + prc:SOCi + tmp:txc + tmp:mng + BD:mng + txc:mng + txc:SOCi + mng:SOCi	57	8983.0	1.5	0.97	0.96	0.01
M0	-	3	16135.1	25119.6	0.08	0	0.08

The table contains the future global model (GM), the best three candidate models (CM), and the null model (M0) for the future period, with initial SOC (SOCi), bulk density (BD), soil texture class (txc), management regime (mng), mean air temperature (tmp) and annual precipitation sum (prc).

Contribution of individual effects to the future SOC balance

The following subsection describes the contribution and effect size of the three categories of effect interactions acquired from the best future CM to the SOC balance simulated by CANDY, as described above.

The effect sizes of the interaction between management regimes on the one hand and soil texture class, BD, initial SOC content, precipitation sum and mean air temperature on the other are summarised in Table 5. The largest causes for changes in the simulated SOC balance according to the best future CM were a result of the interaction between texture class and management regime (Figure 6). The negative effect of extensive mowing management (M1) was thereby highest on silts (-0.60 M-%), followed by loams (-0.48 M-%) and clays (-0.08 M-%). The beneficial effect of intensive grazing (G3) was in turn highest on clays (+2.63 M-%), followed by silts (+2.59 M-%) and loams (+2.29 M-%). An increasing BD resulted in a decreasing simulated SOC balance across all regimes. The effect was particularly high under regimes with a high management intensity (-2.76 M-% under M3, -4.48 M-% under G3), whereby it was substantially higher on the G3 regime than on the M3 regime (Figure 7). The effect of the initial SOC content in turn on the simulated SOC balance was negative on all regimes but the G1 regime. A high initial SOC value thus had a negative effect on further SOC sequestration predicted by CANDY. The effect size of increasing precipitation was small across all regimes and detectable only when annual precipitation amounts increases by an order of magnitude in the hundreds of millimetres, with minor differences between contrasting regimes (see also Additional file 1, supplementary material 9). The largest effect occurred on the M3 and G3 regimes. Increasing temperature resulted in an increase of the simulated SOC balance under the G2 regime, while under

the G3 and M3 regimes it resulted in a decrease, and only minor changes occurred under the G1, M1 and M2 regime (see also Additional file 1, supplementary material 10).

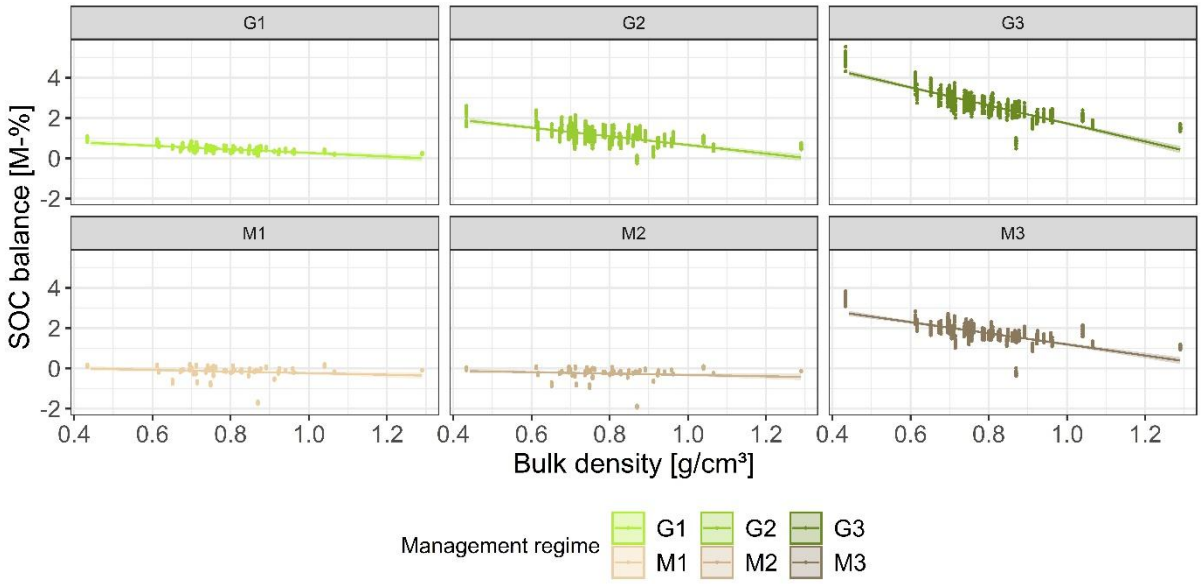

Figure 7: Linear regression (lines) of the effect of changing bulk density on the future SOC balance.

Effect sizes were extracted from the best candidate model and grouped by the management regimes (G: grazing; M: mowing; 1: extensive; 2: moderate; 3: intensive). The SOC balance simulated with CANDY is additionally plotted (point features) for reference.

Table 5: Estimated change in the future (2018 – 2100) topsoil SOC balance grouped by management regime.

Management regime	**Estimated change in SOC balance [M-%] for different texture classes**			**Estimated change in SOC balance [M-%] per one unit increase of…**			
	Clays	**Loams**	**Silts**	**Initial SOC [per 1 M-%]**	**BD [per 1 g cm^{-3}]**	**Precipi-tation [per 100 mm]**	**Air temperature [per 1 °C]**
G1	0.44	0.35	0.45	0.05	-0.90	0.02	0.08
G2	1.12	0.80	0.92	-0.03	-2.14	-0.13	0.30
G3	2.63	2.29	2.59	-0.10	-4.48	0.09	-0.06
M1	-0.08	-0.48	-0.60	-0.07	-0.40	0.07	0.00
M2	-0.18	-0.58	-0.72	-0.08	-0.34	0.05	0.02
M3	1.81	1.33	1.39	-0.08	-2.76	0.28	-0.10

Results are presented 1) in dependence of different texture classes and 2) per one unit increase of the initial SOC content, BD, precipitation sum and mean air temperature across the 50 Hainich grassland sites (based on the best future CM).

The interaction between soil texture class on the one hand and precipitation, temperature, and initial SOC content on the other are summarised in Table 6. Increasing precipitation amounts resulted in minor positive effects across the three soil texture classes in the order silts < loams < clays. Increasing air temperature displayed slightly larger, positive effects size on all three texture classes, with the largest gains on loams and lowest on silts. The effect of an increasing initial SOC content was negative on all three texture classes, with the magnitude increasing in the order clays < loams < silts.

Table 6: Estimated change in the future (2018 – 2100) topsoil SOC balance grouped by soil texture class.

Texture class	**Estimated change in SOC balance [M-%] per one unit increase of...**		
	initial SOC [per 1 M-%]	**precipitation [per 100 mm]**	**air temperature [per 1 °C]**
Clays	-0.01	0.07	0.04
Loams	-0.18	0.06	0.04
Silts	-0.28	0.03	0.03

Results are presented per one unit increase of the initial SOC content, annual precipitation sum and mean air temperature across the 50 Hainich grassland sites (based on the best future CM).

As the last interaction in the best future CM, with the smallest overall effect size, increasing precipitation amounts by 100 mm while increasing the initial SOC content by 1 M-% increased the simulated SOC balance by <0.01 M-% (see also Additional file 1, supplementary material 11).

4.4. Discussion

In this study, the CANDY model was applied to simulate changes in the topsoil (0 – 10 cm) SOC contents in 50 grassland sites of the Hainich Biodiversity Exploratory in Thuringia, central Germany. An ambient period spanning 10 years (2008 – 2017) was used as a baseline to determine recent SOC changes as well as to validate the simulated SOC contents. Following this, six management regimes and 18 climate datasets from representative climate pathway scenarios [30] were used to predict future changes (2018 – 2100) in the SOC balance. Multiple competing statistical models were subsequently fitted to the simulated ambient and future SOC balance in order to identify key predictors and their individual contribution to the overall SOC balance.

Model validation

Applied to simulations on the regional scale in the Hainich Biodiversity Exploratory the CANDY model displayed a good performance with an R^2 of 0.72, RMSE 0.66 M-% SOC and RRMSE 14.83 % when averaging across all sites. The simulated and experimental values follow an overall similar trend in the ambient period and suggest similar dynamics of the SOC changes as a result of management intensity, by proxy of the LUI, and soil texture. However, a detailed comparison reveals partially pronounced differences in the SOC balance when comparing measured and simulated values of individual sites. The experimental values thereby imply both higher losses and gains than predicted by CANDY. A similar observation, i.e. larger measured than simulated SOC decreases, was made by [42] when applying the CANDY model, and [29] also reported a slight over-estimation of SOC contents on loamy soils based on the CANDY carbon balance model. Such discrepancies when comparing simulated and measured SOC contents on a site-to-site level can be explained by several causes. Most importantly, available measurements of topsoil SOC contents in the Hainich sites were limited to two measurements per site for the years 2008 and 2009, and only one measurement per site for each of the subsequent sampling years (2011, 2014, 2017) respectively. A high spatial and temporal variability, and sampling uncertainty of measured SOC contents are thereby known to cause relatively large errors and apparent dynamics, which consequently cannot be captured by simulation models and may result in large discrepancies [26, 29, 55, 56, 114–116]. [82] substantiated that an underestimated spatial heterogeneity, manifested as too few replicate samples, is a major source of uncertainty in experimental SOC assessments and may result in an RRMSE of 30 % under grassland. [83] furthermore found that month-to-month variability in SOC contents may be larger than the variance between replicate samples taken at the same time and thus conclude that samples taken at a single point in time may not be representative of the long-term trend. In Hainich, where clay-rich soils are dominant, this high spatio-temporal uncertainty is additionally exacerbated by the fact that high clay contents are associated with large errors in SOC measurements [117, 118]. Accordingly, in this study the largest RMSE between simulations and measurements was determined for clays. The partially pronounced differences between individual sites when comparing measured and simulated values substantiate the necessity to perform SOC simulations across a high number of contrasting and replicated sites to increase the robustness of simulations to spatial heterogeneity, as was done in this study.

Uncertainties on the modelling side might stem from initially missing soil properties for subsoil horizons, gaps in the original ambient meteorological data as well as uncertainties in reported management data (see also [51, 52, 119]). [9] showed that poorly defined pedo-climatic conditions are

the most common cause for a poor performance of soil C and N models. Using soil maps to acquire missing soil properties has thereby been successfully implemented and resulted in a satisfactory performance in past simulations with the CANDY model [45, 120, 121]. Still, climatic differences between the individual 50 sites in the Hainich Biodiversity Exploratory could not be reflected by the simulations due to an overall high vicinity of the 50 sites on the one hand and relatively few weather stations distributed across the small region on the other hand. Under the future scenarios these uncertainties due to climate were accounted for by the application of three contrasting climate scenarios realised by six projections each. The overall high R^2 and acceptable RMSE attests that the abovementioned limitations were adequately accounted for, particularly considering the fairly recent application of the CANDY model to simulations of grassland sites.

The CANDY parametrisation applied to the simulations in this study was originally developed for national scale simulations on grasslands across Germany [45]. While a dedicated re-parametrisation specifically for the Hainich region most likely would have produced better, and possibly slightly different, results, particularly with an individual parametrisation for each site, a generalised parametrisation has the benefit of being applicable to a wide range of sites and pedo-climatic conditions without sacrificing comparability [85, 122]. Nevertheless, further improvements of the CANDY grassland parametrisation may be performed in future studies, e.g. by detailed calibrations considering also the dynamics of N, temperature and soil moisture in order to achieve a better agreement of long-term and short-term processes [56]. Overall though, the results of the validation performed in this study are well within the range of previous SOC simulations using the CANDY model [55, 56, 85, 114] as well as other simulations performed on grassland sites e.g. with the CENTURY model [36, 37], the RothC model [122, 123], the CN-SIM model [4] as well as a statistical predictive model [124].

Evaluating the ambient and future SOC sequestration potential

By applying the multi-model inference approach to the SOC balance determined by the CANDY model, several significant predictors for the ambient and future SOC sequestration potential, and thereby properties causing a site to become either a sink or source of C, were identified. Under the ambient period, the effect of the LUI as a predictor was dominant to all other site properties (i.e. silt and clay content, BD, initial SOC content). The largest changes in the simulated SOC balance were a result of management intensity, represented by the LUI, followed by the clay content and BD. Similarly, under the future scenarios, largest changes in the simulated SOC balance occurred due to management and its intensity, whereby partially pronounced differences were determined between contrasting soils

(clays, loams, silts). As evident from the future simulations, changing precipitation amounts and air temperature induced only minor changes, which largely depended on the soil texture and management intensity (see also Additional file 1, supplementary material 12). The overall minor contribution of changing climate on the simulated SOC balance is corroborated by the application of a wide variety of contrasting climate scenarios and projections [125]. The model selection further substantiates the reliability of the predictors, since all individual effects and the same nine interactions were retained in the best three future CMs.

Response of the SOC balance to management and potential trade-offs

Impact of management intensity

In line with our simulation results, the predominance of management effects over those of climate in agricultural soils has been widely reported [37, 101, 103, 126]. The ambient simulations using the CANDY model revealed an interaction between the LUI mean, which is the average level of land use intensity across the ambient period (2008 – 2017), and the LUI slope, which is the change of the LUI across the same period and thus represents intensification at positive and extensification at negative values. According to the effect sizes acquired form the best ambient CM, particularly extensively managed sites (LUI<1.4) will benefit from intensification with an increasing SOC balance according to our simulations. At a LUI mean of ≥1.4 the predicted benefit is smaller and further intensification becomes less beneficial. Nevertheless, despite a decreasing effect of further intensification, as per the LUI slope, even at a LUI mean of 3.0 the overall simulated SOC balance remained positive under ambient conditions. As per the CANDY simulations, the only SOC losses (by 0.04 M-%) in the ambient period occurred on a site with a low LUI (0.87), while highest SOC gains (by 0.90 M-%) were a result of a high mean LUI (2.67).

Similarly to the ambient period, under the future period the overall simulated SOC gains were highest for regimes G3 and M3, which had the highest management intensity (see Table 1, Figure 6 and Additional file 1, supplementary material 12). For the future period and averaged across all soil texture classes and climate scenarios, CANDY predicts that SOC increases by an average 2.6 ± 0.6 M-% in the top 10 cm, (minimum: 0.5, maximum 5.5 M-%) and corresponding to 7.4 ±1.0 kg CO_{2eq} m^{-2}, can be expected in the Hainich region over the course of the next eight decades under an intensive grazing management (G3 regime; see Figure 6). Under an intensive mowing regime (M3), CANDY predicts an average increase by 1.7 ± 0.5 M-% (minimum: -0.4, maximum: 3.8 M-%), corresponding to 5.0 ± 1.1 CO_{2eq} m^{-2}, in the top 10 cm. Not accounting for differences due to soil texture or climate scenarios, which were relatively minor compared to the management effects, at these sequestration rates a

grassland area of approximately 1,100 m² per capita under a four month long grazing period with 4 LU ha^{-1} (G3) or a grassland area of approximately 1,500 m² under a five cut mowing regime (M3) would be required to offset the average per capita GHG emissions of the European Union, which were 7.8 t CO_{2eq} in 2021 [127].

When averaged across all soil texture classes and climate scenarios, SOC losses under future simulations occurred exclusively under the M1 and M2 regimes. On the one hand this implies that in the CANDY simulations beneficial effects of intensification under mowing management do not manifest at three cuts per year or lower, yet. On the other hand this also implies overall larger simulated C outputs by harvest than the returns and inputs via residues, organic fertilisers, and vegetation regrowth under the M1 and M2 regime, even at the highest input rates of organic fertilisers permitted by the DüV [65] for the corresponding yield level of the respective regime. The overall SOC losses predicted by CANDY were even slightly larger under the M2 regime (-0.18 M%) than under the M1 regime (-0.08 M-%), despite larger inputs of C via organic fertilisers under the M2 regime (982 kg C ha^{-1} per year) than under the M1 regime (242 kg C ha^{-1} per year). Since the additional C input via organic fertilisers under the M3 regime (1006 kg C ha^{-1} per year) was not substantially larger than under the M2 regime (982 kg C ha^{-1} per year) due to regulations imposed by the DüV [65], this corroborates the assumption that as per CANDY beneficial effects of repeated vegetation regrowth via mowing intensification become apparent above a management including three cuts per year.

Supporting our results, other studies also reported that intensification (in this case increasing the LUI to ≥1.4) may promote SOC sequestration particularly on less productive sites, while especially suboptimal grazing management under extensive grazing and low OM inputs were shown to result in SOC losses in comparison to a higher grazing intensity [38, 128–132]. However, under field conditions overly high intensification and overstocking may cause land degradation and result in unfavourable trade-offs, such as soil compaction, topsoil erosion, surface run-off, imbalance of microbial activity, degradation of SOM, and increased GHG emissions, thus turning former C sinks to sources [18, 133–137]. A comprehensive assessment of the sustainability and productivity of intensively managed grassland in the context of future climate change must furthermore consider trade-offs such as water and energy consumption, eutrophication, acidification, land area required, and emissions in form of GHGs such as methane (CH_4) and nitrous oxide (N_2O), by the application of life cycle assessments (LCA) on a regional or site level [138–142]. Particularly emissions of N_2O account for the majority of agricultural GHG emissions and are highly correlated with application rates of N fertilisers [143–145]. As a result of a wetter and warmer future climate, emissions of N_2O are thereby expected to increase

with increasing grazing intensity [146]. The application and production of mineral N fertilisers in turn is another major factor in the energy consumption and the overall GHG balance of intensive grassland systems [140, 141], emphasising the importance of practices aimed at reducing the requirement for mineral fertilisers. In the overall GHG balance, the abovementioned trade-offs might counterbalance the predicted SOC sequestration under intensive grazing, which had the highest SOC sequestration potential in the 50 representative Hainich sites.

Despite a highly beneficial effect to the SOC balance according to our simulations, increased intensification may furthermore be in conflict with the preservation of biodiversity [147], which is a central goal in the Hainich Biodiversity Exploratory. Particularly extensive and grazed grassland promotes biodiversity, while intensification coupled with high mineral fertiliser inputs, especially under dry conditions, is adverse [18, 148–150]. Some evidence thereby suggests a smaller decline in biodiversity due to intensification under grazing management than under mowing management [151]. A decreased diversity of soil flora and fauna in turn may be adverse to the nutrient cycling and storage abilities of the soil, reducing its fertility and ability to sequester C on the long term [152–156]. It is therefore crucial to identify a reasonable middle of the road solution that considerably contributes to SOC sequestration efforts as well as maintains biodiversity while minimising undesirable trade-offs [11, 16, 157, 158]. Overall though, while the current management intensity of the selected study sites in the Hainich Biodiversity Exploratory is at a low to moderate level [51] and potential adverse effects of intensification are not expected to occur yet, it is necessary to be aware of potential trade-offs, particularly those that are not reflected by simulations.

Comparison of the SOC balance of grazing and mowing regimes

The future management regimes, which were derived based on the ambient management data, allowed to identify individual effects due to grazing and mowing management, which was not feasible under the ambient management due to grazing and mowing being confounded and not standardised within the management of all sites. The CANDY simulations under future conditions show that the grazing regimes are more beneficial to SOC sequestration than the mowing regimes. The more beneficial effect of grazing under future scenarios persists even despite a higher LUI of the moderate and intensive mowing regimes than the respective grazing regimes (see Table 1). In case of the intensive regimes (G3 and M3), clays, loams and silts had a higher simulated SOC balance under the G3 regime than under the M3 regime, despite the LUI of the G3 regime being 0.25 lower than that of the M3 regime (see Table 5). This implies beneficial effects beyond the mere increase or decrease of the LUI.

In a previous study, where mixed management regimes were applied, i.e. regimes combining mowing and grazing as well as both organic and mineral fertilisers, differences between mixed regimes on the one hand and exclusive mowing regimes on the other hand on the simulated SOC balance were smaller, although the mixed grazing regimes overall provided a larger benefit than the exclusive mowing regimes [45]. Under field conditions, the benefit of management practices that at least partly include grazing periods is largely caused by overall higher C returns with animal excreta than the C returns under exclusive mowing management [33, 129, 159–165]. This might prevail even despite larger additional C inputs via organic fertilisers, which are of high importance to the SOC balance of agricultural soils, under exclusive mowing management [165–169]. In our simulations, the M1, M2 and M3 regimes received 242, 982 and 1006 kg C ha^{-1} via organic fertilisers per year respectively; in contrast, the G1, G2 and G3 regimes received an input of 384, 563 and 770 kg C ha^{-1} via organic fertilisers (see Additional file 1, supplementary material 2 for properties of organic fertilisers) and additionally 212, 425 and 849 kg C ha^{-1} directly via excrements of grazing cattle (at a daily C input of 1.74 kg C ha^{-1} per LU as per the default CANDY parametrisation), resulting in higher C inputs under grazing management. Overall, this means that the aggregation of complex management information into one simplified index, such as the LUI, does not capture all information relevant to the prediction of the SOC balance, such as the amount of C returns and inputs. Nevertheless, other effects might also persist under field conditions, such as the involvement of microorganisms in both the turnover processes and the stabilisation of SOC under grassland [170], which, however, is not reflected by our CANDY simulations.

In the future CANDY simulations, differences in the SOC balance between grazing and mowing were particularly pronounced at a low management intensity (M1 and G1), where the total C input by organic fertilisers and return by grazing animals was more than twice as high under the G1 regime than under the M1 regime. This indicates that the magnitude of additional C inputs on the SOC sequestration is assumed to differ in the CANDY model among a land use intensity gradient. Consequently, this also implies that the simulated C return by grazing is more prominent at low management intensity, whereas at high management intensity, effects associated with C input via vegetation regrowth and additional organic fertiliser input are more dominant. However, at least within the scope of this study, this is highly speculative and requires further investigation, particularly under field conditions in order to either corroborate or refute this assumption. Overall, under field conditions, substituting mineral with organic fertilisers allows both meeting the N requirement and providing additional C inputs [166, 171, 172], although the application rates of organic fertilisers must

follow good farming practice in order to avoid over-fertilisation with other nutrients contained in the organic fertilisers [65, 173].

Response of the SOC balance to soil properties

Impact of soil texture

Despite a prevalence of management effects over the effects of soil properties and climate in our simulations, some interactions and varying responses of different soil texture classes could be determined. The interactions and effect sizes are particularly prominent under future conditions. Due to the dependence of SOC increases on the available mineral surface, under field conditions the largest sequestration effects are usually expected on fine-textured soils (i.e. clays), whereas the effects are smaller on coarse soils [100, 101, 105, 174–176]. Our simulation results under future scenarios are in accord with this, showing the largest management-induced increases on clays (see Figure 6 and Additional file 1, supplementary material 12). [105] presented evidence suggesting that clays may store a relative >30 % more SOC than loams or silts, which is also in accord with our results. The clay content, and texture in general, is therefore a relevant factor explaining the sequestration potential in long-term simulations [36, 45, 102]. However, the results presented for loamy soils are accompanied by a fairly large uncertainty due to a low number of measurements (n = 6, see Table 2), which is manifested in the wide error bars of the estimated marginal means (Figure 6).

Impact of bulk density

As a result of severe compaction under field conditions, the interparticle space where storage of water, transportation and transformation processes occur, as well as the contact area for adsorption of particles, decreases, thus reducing the incorporation of organic matter (OM) into the soil and limiting sequestration [177–180]. In CANDY, this process is mirrored by the BAT, which quantifies the effect of soil moisture, aeration, particle surface area, and temperature on the decomposition and storage of SOC, whereby the BD is used to calculate the pore volume as one of the required parameters [54]. Accordingly, under ambient simulations, increasing BD was adverse to sequestration, whereby the effect increased with increasing clay contents along a gradient from 5 to 70 M-% (see Figure 5). This is in accord with the increasing susceptibility to compaction with increasing clay content widely reported under field conditions [179, 181, 182]. The effect of BD being adverse to sequestration was also confirmed in future simulations, whereby the effect increased with management intensity and was particularly high under the grazing regimes (see Figure 7). Since on average sequestration rates increased with management intensity in the future simulations and were higher under the grazing than under the mowing regimes, these regimes were consequently also most prone to reduced

sequestration rates with increasing BD in our simulations. A high grazing density and frequent use of heavy machinery are thereby central causes for soil compaction under field conditions, requiring adapted management strategies to counteract or avoid this [183–185]. Otherwise, the beneficial effect of increased SOC inputs via vegetation regrowth under intensive management may be denied or even counterbalanced under field conditions in case of severe compaction.

Impact of the initial SOC content

The initial SOC content has a deciding impact on the effect other site properties play in the overall sequestration potential under field conditions, since a high initial SOC content reduces the remaining particle surface for further SOC accumulation [36, 131, 176, 186–188]. The initial SOC content was thereby not a significant predictor for the simulated SOC balance in the ambient period, while it was significant in the future period. This is because the relevance of the initial SOC content increases as the SOC content approaches saturation in simulations and under field conditions alike [121, 189–191], whereby SOC saturation is reached sooner when starting from a high initial SOC content than from a lower initial SOC content [36, 176, 186–188]. Vice versa, a longer simulation period offers more time for the SOC contents to reach saturation than a shorter period does.

The initial SOC content plays a role in the context of intensification in the simulation results presented in this study. On the one hand, high initial SOC contents mean high potential for losses under extensive to moderate use (M1 and M2 regimes in particular) according to the CANDY simulations, while on the other hand low initial SOC contents present an opportunity for high sequestration rates under moderate to intensive management when combined with high inputs of C via fertilisers (M3 and G3 regimes). Consequently, in the future simulation period high initial SOC contents reduced the SOC gain across all management regimes, with a notable exception of the extensive grazing regime (G1). The latter exception is likely caused by a combination of two effects. First, as per our simulations, a low management intensity (M1 and G1) resulted in overall low management-related SOC gains. This causes a longer duration until SOC contents reach saturation compared to a higher management intensity, and thus no limiting effect of a higher initial SOC occurring within the simulated period. Second, differences between total C-inputs and returns between mowing and grazing regimes (see “Comparison of grazing and mowing regimes”) resulted in overall simulated SOC losses under the extensive mowing (M1) and gains under extensive grazing (G1) management. The predicted losses under the M1 regime thereby increased at higher initial SOC contents. Consequently, under the G1 and M1 regime the higher initial SOC contents did not limit the simulated SOC gains, yet, whereby under the M1 regimes they even resulted in potentially higher losses. Nevertheless, the differences in

the impact of changing the initial SOC across contrasting management regimes are relatively minor compared to the overall changes due to management.

The reported negative effect of increasing the initial SOC content on the overall SOC gain was highest on silts in the future simulations. Since the specific surface area available for SOC storage decreases under field conditions in the order clays > loams > silts, increasing the initial SOC content by the same absolute amount leaves proportionally less available surface area for further SOC accumulation on silts than on clays or loams [192–195], which was accordingly reflected by the CANDY simulations.

Curiously, the best future CM suggests that increasing the initial SOC content while simultaneously increasing precipitation is beneficial to the simulated SOC balance. However, as a result of that interaction the predicted SOC balance increased by <0.01 M-% when precipitation amounts increased by 100 mm and the initial SOC content increased by 1 M-%. Such an SOC gain would be undetectable by measurements and is thus negligible [26, 28, 196], particularly considering that at such a small effect size uncertainties associated with gaps and inaccuracies of input data might be prevalent [9, 121, 190, 197], rendering the apparent effect to be most likely a model artefact.

Response of the SOC balance to future climate

Impact of precipitation

The future simulations revealed a minor response of the Hainich grassland sites to changing precipitation, despite increases of the annual precipitation sum from 542 mm in the ambient period to up to 710 mm under future climate (see Additional file 1, supplementary material 6). Under field conditions, increasing precipitation allows for increasing the cutting frequency, since repeated vegetation regrowth results in higher water demand but also increases SOC sequestration [12, 16, 122, 130, 132, 176]. In our simulations the largest SOC increases with increasing precipitation indeed occurred under a five cut regime (M3), though the beneficial effect of precipitation was relatively minor.

According to the CANDY simulations the Hainich region experiences no dryness stress considering the mean air temperature. Nevertheless, the simulated SOC balance under the intensive regimes (M3 and G3) showed a positive response to increasing precipitation, likely due to increasing water uptake at this management intensity [26, 102, 122, 176, 198]. Accordingly, based on simulations with the PaSim model, [33] found that increasing precipitation promoted the net primary production, and by extension the SOC sequestration, under dry conditions. [176] furthermore reported that the largest benefit to SOC sequestration occurred by increasing annual precipitation from <400 mm to 800 – 1.200

mm, while no significant effects occurred beyond that. The soil texture thereby largely determines the water storage abilities and how turnover rates react to changing climate [102, 199, 200]. Correspondingly, in our simulations clays benefited the most from increasing precipitation amounts due to their better water storage abilities and subsequent reduction of SOC mineralisation [102, 199, 200]. However, the magnitude of the effect was small, due to the abovementioned overall sufficient precipitation amounts.

Impact of air temperature

Since rainfall amounts appear to be at an optimum in Hainich and since further increases do not substantially promote the simulated SOC gains due to reduction of the mineralisation, especially if the soil moisture is above field capacity, changes in the mean air temperature are slightly more relevant in our study. Under field conditions, increasing temperatures may thereby result in increasing microbial activity, evapotranspiration and thus increased SOC mineralisation and losses, unless the increased water requirements are compensated by increased precipitation as well [26, 102, 122, 176, 198]. In fact, precipitation amounts and air temperature are slightly co-linear under the future scenarios in the Hainich region, as evident by the VIF.

In our simulations, increasing air temperature resulted in a differentiated effect. First, silts benefited the least from increasing air temperature. Under field conditions, this is explained by a lower water storage ability and higher susceptibility to dryness of silts when compared to clays and loams [102, 199, 201, 202]. The CANDY simulations reflect on these pedoclimatic field conditions by quantifying the impact of aeration, temperature and texture on the turnover conditions via the BAT concept [54]. Second, the moderate intensity regimes (G2 and M2) benefited the most from increasing temperature, while the intensive regimes (G3 and M3) displayed an adverse effect of increasing temperature to SOC gains. This suggests that the simulated overall water losses due to increased evapotranspiration under intensive management and increasing temperatures surpasses the water availability. Similar results based on simulations with the PaSim-Model were reported by [33]. Generally though, under field conditions increasing air temperatures increase plant growth and promote sequestration in mid- and high-altitude regions where the vegetation period is limited by low temperatures and the snow cover [21, 126]. [33] reason that increasing global air temperatures will cause an altitudinal shift in the grassland phenology and productivity, whereby mid- to high-altitude grasslands will become more temperate. Based on simulations with the same RCP scenarios as used in our study, [21] showed that as a consequence of a warming by up to 2 °C until 2100 the Alps region and Northern Europe are expected to experience an increase in the duration of the vegetation period, which will result in a

higher sequestration potential. In Hainich, the average duration of the vegetation period across all 50 grassland sites for the reference year 2005 was determined to 219 days, which is considerably less than the 256 days determined for national-scale simulations across Germany [45]. Warming temperatures in Hainich will therefore likely cause an extension of the vegetation period, resulting in larger SOC inputs via roots, unless negated by insufficient rainfall [27, 203]. Nevertheless, [21] also predicted that a warming level of 3.5 °C will present a tipping point in large areas of Europe, resulting in increased droughts and likely turning former SOC sinks to sources. A meta-analysis by [105] reported that the change from a temperate to warm climate with a mean air temperature >15 °C would decrease SOC by 60 % across arable and grassland sites. However, such extreme changes in air temperature and climate are not expected in Hainich according to the RCP scenarios, which renders it a suitable and resilient region for future sequestration even under the worst-case scenarios.

4.5. Conclusion

By applying the CANDY model to a dataset of 50 grassland sites in the Hainich Biodiversity Exploratory, first the SOC balance was simulated under ambient conditions (2008 – 2017), representing the status quo of management. Then, 18 climate datasets and six contrasting management regimes representing a three-step intensification gradient for mown and grazed grassland respectively were applied to each site to simulate future conditions (2018 – 2100), in order to investigate the individual contribution of management, climate and pedogenic properties to the simulated SOC sequestration potential as well as their interactions. We investigated 1) whether increasing the management intensity is a viable mean to promote sequestration in a representative cool, mid-altitude grassland, 2) whether sequestration rates differ between grazing and mowing regimes, 3) whether increased temperatures and growing season precipitation under future climate will bolster SOC sequestration and 4) whether different soil texture classes (clays, loams, silts) respond differently to the effects of climate and management, resulting in an overall varying potential for SOC sequestration and losses.

According to our simulations, the effect of management predominated the effects of climate and soil properties, with intensification largely promoting SOC sequestration. Under ambient simulations, intensification was proven beneficial to SOC sequestration, whereby gains occurred even under the most intensive management investigated (LUI of 3.07). Under future simulations, increasing the management intensity to five cuts per year, corresponding to a LUI of 2.35, or implementing a grazing management with a four month long grazing period with a stocking density of 4 LU ha^{-1}, corresponding to a LUI of 2.10, was shown to promise the largest SOC increases. According to the CANDY simulations performed over the next eight decades, SOC gains under an intensive grazing regime may amount to

2.6 ±0.6 M-% SOC on average (minimum: 0.5, maximum 5.5 M-%), corresponding to 7.4 kg CO_{2eq} m^{-2}, while an intensive mowing regime may result in average gains of 1.7 ±0.5 M-% (minimum: -0.4, maximum 3.8 M-%), corresponding to 1.4 kg C m^{-2} and 5.0 CO_{2eq} m^{-2} in the upper 10 cm. In future simulations, exclusive grazing regimes had a higher SOC balance than mowing regimes, despite larger organic fertilisers inputs under medium and high intensity mowing regimes. Nevertheless, these simulated balances do not account for potentially increased GHG emissions via N_2O, CH_4, higher water and energy demand, and higher risk of erosion and soil compaction, which may at least partly counterbalance these benefits.

In contrast to multiple studies reporting increased SOC sequestration as an effect of higher precipitation amounts, this effect was small in Hainich. In our simulations, increasing precipitation slightly promoted sequestration at higher management intensity via reducing mineralisation and compensating for the higher evapotranspiration due to also increasing temperature. The small magnitude of this effect is likely a result of fairly low mean air temperature and no dryness stress occurring during the entire year. Instead, in Hainich the productivity and SOC sequestration potential is limited by the duration of the growing season. Increasing mean air temperatures under future climate change in the cool mid-altitude climate will likely cause an extension of the vegetation period, allowing for more biomass growth, higher management intensity and thereby promote SOC sequestration. Thereby, medium intensity management regimes (grazing with 2 CU ha^{-1} over four months or mowing three times a year) benefited the most from the temperature increase under future climate.

Besides the management effects, several effects on the SOC balance predicted by CANDY related to varying pedogenic properties were also found. For one, the soil texture largely determined how the simulated SOC balance of a site reacted to varying climate conditions. In our simulations, clays benefited the most from increasing precipitation and offered the overall highest potential for SOC sequestration due to their high water storage ability and particle surface area, while silts were most prone to dryness, and thereby to SOC losses. In contrast to the ambient period, the simulated SOC gain during the future period also depended on the initial SOC content, which can be explained by the longer simulation period and the resulting SOC contents approaching saturation. High initial SOC contents thereby resulted in a lower predicted SOC gain on silts due to a lower particle surface than clays and loams, and thus lower remaining available surface area for SOC storage until saturation. A certain threat to SOC sequestration according to the CANDY simulations, both under ambient and

future conditions, was caused by an increasing BD, which reduces the effective particle surface available to SOC storage.

According to our simulations, the Hainich region will remain largely resilient to future climate change. As per the CANDY simulations, clay-rich sites with initially low to moderate SOC contents (<5 M-%) under moderately high management (grazing with 2 – 3 LU ha^{-1} or 3 – 4 cuts per year) and high inputs of organic fertilisers can constitute considerable C sinks while benefiting the most from increasing air temperatures, although a comprehensive evaluation of the entire life cycle is necessary in order to assess the overall GHG balance, including e.g. emissions via N_2O and CH_4 as well as water and energy required. Conversely, extensive management (grazing with 1 CU ha^{-1} or 1 cut per year) with low inputs of organic fertilisers poses the highest risk of C losses, especially on silty soils or when the SOC content is near saturation. Based on our results, we recommend increasing the LUI to approximately 1.4 in the Hainich region in order to further foster sequestration, particularly on clay-rich sites with still low SOC contents, while simultaneously maintaining soil structure and biodiversity. While simulations still imply increasing SOC sequestration rates at a LUI >1.4, the benefit decreases and other adverse effects, such as compaction and topsoil erosion, might become prevalent. Overly high intensification furthermore poses a potential conflict of interest with the preservation of biodiversity, which is a central goal in the Hainich Biodiversity Exploratory.

Availability of data and materials

This work is based on data elaborated by several projects of the Biodiversity Exploratories program (DFG Priority Program 1374). The datasets are publicly available in the Biodiversity Exploratories Information System (http://doi.org/10.17616/R32P9Q). The datasets are listed in the references section.

Competing interests

The authors declare that they have no competing interests

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Authors' contributions

Matthias Filipiak: Conceptualisation, methodology, performing simulations and analyses, interpretation, data curation, writing – original draft, writing – review and editing, visualisation.

Katrin Kuka: Conceptualisation, methodology, interpretation, writing – review and editing, supervision.

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5. General discussion

Considering the expected future climate change scenarios, agriculture will be faced by multiple forms of challenges, such as meeting sustainability goals, maintaining soil health, providing ecosystem services and meeting productivity demands. In order to contribute to national and global efforts of a sustainable agriculture, this thesis aimed at providing management recommendations to protect the physico-chemical quality of arable and grassland soils as well as investigate the soil organic carbon (SOC) sequestration potential under representative permanent grassland sites, which typically have a substantial sequestration potential, across Germany in the coming decates. This chapter will provide a short summary of the results of the previous chapters and discusses further possible means of increasing SOC stocks under agricultural soils as well as limitations of the performed modelling approaches in order to provide a baseline for future research.

5.1. Summary of the results

Chapter 2 investigated the impact of a long-term arable and grassland use on the chemical soil properties. The literature review substantiated an increased risk of degrading chemical soil properties under arable land when compared to permanent grassland under otherwise similar site conditions. These include an increased risk of nitrate (NO_3) leaching, gaseous nitrogen (N) emissions, loss of SOC, and off-site damages such as eutrophication by N or phosphorus (P). These changes are largely a result of topsoil erosion and increased macroporosity due to regular tillage, compaction due to heavy field trafic particularly during wet phases, wind erosion during bare periods, and a more open nutrient cycle, requiring high inputs of among others C, N and P to compensate for the higher uptake and removal under arable cultivation. Elevated nutrient contents, particularly N and P, due to past intensive arable cultivation can thereby persists for decades to centuries after conversion to grassland, which have a lower nutrient output rate, and thereby even prevent the establishemnt of a biodiverse vegetation.

Chapter 2 also provided management recommendations aimed at decision makers and farmers. The majority of chemical soil properties can be improved by introducing a biodiverse, permanent vegetation, practicing reduced or conservation tillage on arable land, avoiding bare phases on arable land e.g. by growing catch crops, increasing organic matter (OM) inputs e.g. by retaining crop residues or substituting mineral with organic fertilisers, including legumes in crop rotations and grassland mixtures, implementing grassland phases within an arable crop rotation, and by prioritising grazing over mowing on grassland while simultaneously avoiding overstocking. Especially high contents of SOC, which are typical for grassland sites, are associated with improvements of the nutrient status of soils, since SOC has a affinity to store N, P, potassium (K) and micronutrients. The chemical status of the soil,

particularly its pH level, should thereby be monitored regularly on arable and grassland alike, especially if acidifying fertilisers are applied. Overall, bridging the gap between arable and grassland sites can substantially improve the chemical soil properties of arable sites, thus maintaining the soil quality and the ecosystem services it provides.

The simulations of climate change and management scenarios on representative grassland sites across Germany (chapter 3) substantiated the largely reported dominance of management effects over the effects of climate as well as soil properties. The simulations revealed that sites under a management with three to five cuts per year have a high potential to sequester SOC by the end of the century under a variety of climate scenarios. Management with two or fewer cuts per year in turn poses a risk of SOC losses. On the national scale, the most important meteorological metric impacting SOC sequestration and mineralisation according to the simulations was the seasonality of rainfall. Therby, increasing precipitation during the growing season reduced the simulated mineralisation an accomodated the increasing water requirement due to increasing mean air temperatures under future climate, as well as the increasing water uptake at higher management intensity due to repeatet vegetation regrowth. On the national scale, clay-rich soils were identified as “high-risk-high-gain” soils, having both the highest SOC sequestration potential at high management intensity (i.e. up to five cuts per year) and sufficient rainfall, as well as highest SOC losses at low management intensity and dryness. Across all 24 investigated sites, only one representative site displayed a negative simulated SOC balance, i.e. SOC losses, when averaged across all evaluated 18 future climate change scenarios as well as all ten management regimes applied. The site was a clay-rich soil with precipitation amounts <620 mm across all climate change scenarios, in the soil region “loess and sandy loess area” in Thuringia, in central Germany. In contrast, the majority of sites with high SOC gains were located particularly in the sub-alpine and low mountain regions, which were characterised by high clay contents (>20 M-%), high precipitation amounts (partially >1.000 mm per year) and mean air temperatures rarely higher than 10 °C.

The simulations across a mid-altitude grassland region in the Hainich Biodiversity Exploratory in central Thuringia (chapter 4), which was coincidentally identified as a region of interest by the results presented in chapter 3, show partially similar but also partially different trends on a regional scale than on a national scale. Just as on the regional scale, intensification of land use, both in form of increasing cutting frequency as well as increasing stocking rates, were substantial to a positive SOC balance both in the ambient (2008 – 2017) as well as future period (2018 – 2100). As per evidence by chapter 3, the “loess and sandy loess” region of Thuringia is particularly prone to SOC losses under suboptimal, i.e. extensive (less than three cuts per year), management. This is in accord with the results of chapter 4,

wherein according to the simulations especially extensively managed sites (two or fewer cuts per year) will benefit from moderate intensification (grazing with 2 – 3 livestock units (LU) ha^{-1} or mowing with to 3 – 4 cuts per year). However, above a certain intensity threshold this effect diminished, as other effects, e.g. soil compaction, became more dominant. Exclusive grazing regimes were identfied as substantially more beneficial to the SOC balance than exclusive mowing regimes due to overall larger SOC returns with animal excreta as per the simulations, despite larger inputs of C-rich organic fertilisers and a higher land use intensity (LUI) index under comparable mowing regimes. Interestingly, the beneficial effect of intensitication appeared to be non-linear in the simulations under the exclusive mowing regimes, since both the medium intensity mowing (i.e. three cuts per year) and the low intensity mowing regime (i.e. one cut per year) displayed fairly similar and minor SOC losses, while high intensity mowing (i.e. five cuts per year) displayed considerable gains. Clay soils were again identified as having the largest potential for SOC gains, while silts were most prone to losses. Unlike on the national scale, however, the seasonality of precipitation was not a significant predictor in the Hainich region. The impact of changing precipitation amounts, and also changing temperature, as a result of future climate change across the same 18 climate change scenarios was overall negligible when compared to the effect of soil properties and particularly management. This is largely due to no dryness stress occuring in the Hainich region considering the fairly low mean air temperature, whereby the vegetation period is rather limited by the snow cover rather than by rainfall amounts. Consequently, the Hainich region is resilient to the expected climatic changes, displaying rather beneficial changes in the SOC sequestration potential as a result of increasing temperatures. The simulations in the Hainich region also revealed that, besides extensification and thus low C input via vegetation regrowth and rhizodeposition, increasing soil bulk density (BD) presents a high risk of SOC losses and reduced sequestration potential. This effect was particularly prominent on clay-rich soils with a high management intensity, which, paradoxically, were also the properties promising the highest SOC sequestration. Finally, the degree of SOC saturation at the start of the future period was shown to limit further sequestration, as sites with high SOC contents displayed the lowest potential for further accumulation, particularly on silty soils, due to a lower specific particle surface available for SOC storage compared to loams or clays.

5.2. Sites of primary interest and associated management challenges

Chapter 3 revealed the “loess and sandy loess” soil region as overall most susceptible to SOC losses under future climate change and chapter 4 substantiated that in this region, particularly sites under

extensive mowing management are at highest risk of SOC losses under future climate change. In the same region, as evident by chapter 4, moderate intensification (i.e. 3 – 4 cuts per year or grazing with 2 to 3 LU ha^{-1} over a period of four months) particularly on clay-rich soils was shown to have a high potential to increase SOC according to the simulations, especially under grazing management. However, clay contents >15 M-% and intensive grazing (4 LU ha^{-1}) were also associated with the highest predicted SOC losses under increasing soil compaction. Such regions should thus be in the focus of future sustainability and carbon (C) farming strategies to maintaint their productivity and resilience to external stressors under changing climate. The intensification of land use on clay-rich soils should thereby go hand in hand with monitoring the soil BD and introducing counter measures and strategies to avoid it, e.g. by optimising duration and timing of grazing periods, livestock species and breed, as well as diversification of vegetation composition (Milazzo et al., 2023; Trimarco et al., 2023). For instance, rotational grazing systems across multiple subdivided grassland paddocks may aid in homogenising the C returns via animal excreta and also optimise spatial distribution of cattle, thus reducing spatially concentrated compaction by trampling (Byrnes et al., 2018; Mosier et al., 2022; Probo et al., 2014; Trimarco et al., 2023). At least temporarily housing livestock in barns during periods of increased wetness might also prove essential to avoiding soil compaction (Bilotta et al., 2007; Milazzo et al., 2023; Trimarco et al., 2023). The review by Milazzo et al. (2023) furthermore substantiated the importance of a diverse grassland species composition to increase the soil's resilience to physical stressors, such as compaction, aggregate disruption, and erosion. This might in turn present a novel challenge considering the expected shift of vegetation zones under future climate (Chang et al., 2017; Milazzo et al., 2023; Riedo et al., 2000) and thereby caused requirement for farmers to use new adapted plant species. Finally, substantiating the socio-economic complexity of the overall issue, it is worth noting that the most efficient management with the optimal SOC balance may not necessarily be in accord with animal well-being, as evident by a study reporting lower GHG emissions in feedlot production than in adaptive multi-paddock grazing (Stanley et al., 2018).

5.3. Exploring the potential of deep tillage: bridging the gap between arable and grassland

As outlined in chapter 2, conversion of grassland to arable land typically causes large SOC losses and the subsequent SOC gains in the arable phase often can not compensate for the losses caused by the conversion process itself (Guo and Gifford, 2002a; Janssens et al., 2003; Oberholzer et al., 2014; Springob et al., 2001). However, the production of one calorie by arable crops usually requires less land area, energy, and water and causes less greenhouse gas (GHG) emissions than the production of

one calorie by animal products, such as milk, meat, and cheese (Boke Olén et al., 2021). It is therefore necessary to bridge this gap by developing farming practices and policies that on the one hand meet the demands of providing sufficient food, fodder, and biomass and on the other hand sequester atmospheric C while minimising GHG emissions and the impact of agricultural land use on terrestrial habitats, particularly considering the increasing competition of land area for production of food, energy or practices dedicated to nature conservation and climate protection (Guyomard et al., 2021; Harvey and Pilgrim, 2011; Jeanneret et al., 2021; Taube et al., 2014; van de Ven et al., 2021).

One such strategy applicable to arable land, as identified in chapter 2, is deep tillage, a one-time practice that transports SOC-rich topsoil material below the regular ploughing layer and in turn brings SOC-poor subsoil material to the top. Assuming no deep tillage has been implemented before, the buried old topsoil is subsequently subject to little disturbance, presenting favourable conditions for long-term SOC stabilisation, while the old subsoil displays a high available mineral surface area to store new SOC (Alcántara et al., 2016; Feng et al., 2020; Lawrence-Smith et al., 2021; Madigan et al., 2022; Schiedung et al., 2019). While this concept has already been widely explored on arable land (e.g. Alcántara et al., 2016; Feng et al., 2020), only few studies investigate the implementation of deep tillage on grassland, mostly on sites in New Zealand (Calvelo-Pereira et al., 2022; Lawrence-Smith et al., 2021; Schiedung et al., 2019). In these studies, implementing deep tillage on intensive grassland with topsoils near SOC saturation was thereby shown to have a potential to substantially increase the SOC storage in the new topsoil as well as across the entire soil profile, given no disturbance of the subsoil after the initial deep tillage event as well as beneficial conditions promoting topsoil sequestration as well (Lawrence-Smith et al., 2021; Schiedung et al., 2019), which presents a promising research subject for the European temperate zone as well. Furthermore, by converting the deep tilled grassland to arable land it is, in theory, simultaneously possible to maximise the production of food per land area, given sufficient political, ecological, economical, and ethical incentive to reduce the consumption of animal products. While the conversion of particularly species rich grassland to arable land is currently discouraged in the EU as it poses a conflict of interest with the preservation of biodiversity and maintaining regulatory ecosystem services (European Commission, 2022), such an approach could focus on the conversion of intensive grassland sites, which provide less regulatory and more production ecosystem services (Baer and Birgé, 2018; Barrios, 2007; Hasan et al., 2020; Paudel et al., 2021; Power, 2010; Zhao et al., 2020) and furthermore were shown to display a potential to sequester large amounts of SOC, as per the simulations and research results presented in this thesis. This in turn could provide an opportunity to substantially increase both the SOC storage and biomass output per land area of agricultural land while simultaneously reducing the required land area for

agricultural production, opening new opportunities to concepts such as sparing land from agricultural use to provide other ecosystem services, particularly maintaining biodiversity on extensive grasslands. Nevertheless, this approach would require extensive testing to assess its potential for SOC storage, in form of both field experiments as well as long-term SOC simulations coupled with life-cycle assessments (LCA) in order to include saved emissions of carbon dioxide (CO_2), methane as well as nitrous oxide; an LCA would furthermore allow to also consider the SOC lost during the conversion process and consequently allow to identify the timespan after which the SOC balance turns positive after the initial soil disturbance and SOC loss (Carrilo Quijano et al., 2024).

5.4. Approaching the aggregation of complex management information into indices

Chapter 4 opend up a discussion about the difficulty of aggregating complex management information into one dimensionless land use intensity (LUI) index. Therein, the LUI index was used as an explanatory variable in a statistical model applied to the CANDY simulation results to identify significant contributors to the simulated SOC balance under the ambient period (2008 – 2017). The results revealed that a lot of substantial information is lost when aggregating stocking rates, grazing duration, fertiliser application rates and type, and mowing frequency and yield into one value, which was also already suggested by Blüthgen et al. (2012). However, even the individual components of the aggregated LUI, i.e. the LUI grazing, LUI mowing and LUI fertiliser, do not carry sufficient relevant management information. For example, the LUI fertiliser aggregates both mineral and organic fertilisers into one value, the LUI mowing does not consider yields, and the LUI grazing does not distinguish between short but intensive and long but extensive grazing periods, with the grazing intensity used for the calculation of the LUI grazing being the product of the stocking rate and the grazing duration. Generally, mineral fertilisers, which are rich in N but lack C, typically reduce root growth and rhizodeposition, and thereby SOC sequestration, since the necessity for biomass build-up belowground for nutrient uptake decreases due to a high N availability (Pausch and Kuzyakov, 2018). In contrast, organic fertilisers provide the required N and are an additional source of C and thus substantial to the total C balance of agricultural soils (Jacobs et al., 2020; Maltas et al., 2018; Wei et al., 2021). By aggregating both mineral and organic fertilisers into one value, this crucial information is lost. However, a standardised quantification of management intensity into few but meaningful indices, which ideally can also be easily implemented into SOC predictions, is necessary in order to reliably compare contrasting management options, particularly between different studies (Milazzo et al., 2023). While according to the simulations performed in Hainich (chapter 4), and per implication in

representative sites across Germany (chapter 3) as well, grazing was indeed more beneficial than mowing and the key cause for this was identified (i.e. larger C input by returns via animal excreta than by harvest residues and organic fertiliser input according to the simulations), these results are only valid across the investigated pedoclimatic properties and management intensities. The lack of an ubiquitously valid approach of standardising management indicators severely impedes the comparability with studies under different site conditions and management. Consequently, the elephant in the room is the still lacking consensus on the effect of grazing being largely beneficial or adverse to sequestration across literature (see chapter 1).

5.5. The relevance of changing precipitation patterns across varying spatial scales

The results of chapter 3 suggested a larger relevance of the average seasonality of rainfall on the simulated SOC sequestration potential on a national scale in Germany. However, the result of chapter 4 suggest that on a regional scale, at least in the Hainich Biodiversity Exploratory, the mean annual precipitation sum was more relevant instead, although the effect was largely negligible in comparison to the impact of overall management. This discrepancy suggests different relative magnitudes of the seasonality of rainfall, its long-term change and mean annual sum across different spatial scales and site properties, and warrants further investigation. Formulating generalised but reliable management recommendations aiming at increasing SOC sequestration without the requirement of specialised, time-consuming simulations requires the identification of key influential factors that are both easy to determine and valid across a broad spectrum of pedoclimatic conditions. Precisely quantifying the impact of changing climate thus contributes to robust predictions of the SOC sequestration potential across varying site properties and management options.

Despite the abovementione discrepancy, the available data and generated results still allow a few conclusions. On the national scale (chapter 3) several sites with insufficient summer rainfall as well as higher sand contents were present and an increasing seasonality of rainfall resulted in more summer precipitation, thereby reducing dryness stress, aeration, and mineralisation, particularly on the sandy sites. This was not the case on the regional scale in the Hainich Biodiversity Exploratory due to the dominance of clay-rich soils and their overall good water storage abilities (Basset et al., 2023; Bormann and Klaassen, 2008; Moyano et al., 2011). Generally in simulations, a central cause for higher SOC mineralisation and losses under future scenarios is a higher water depletion and reduced moisture during the growing season because of a lower summer precipitation (Bai and Cotrufo, 2022; Chang et al., 2017; Puissant et al., 2017), particularly in already arid regions (Guo and Gifford, 2002b; Post and

Kwon, 2000; Swemmer et al., 2007). However, the seasonality of rainfall, which was defined as the mean standard deviation of the annual precipitation sum in chapter 3, hardly exceeded 4 mm in the Hainich region even under the most extreme future climate, resulting in evenly distributed rainfall across the entire growing season. Conducting a comprehensive analysis across multiple sites that takes into account both the seasonality of rainfall and mean annual precipitation sums to predict SOC changes might shed more light on the relationship between rainfall patterns and SOC dynamics and thus can inform better land management practices. In particular identifying thresholds at which the seasonality of rainfall dominates the absolute rainfall amount regarding their impact on the SOC sequestration potential could contribute to to identifying sites and regions prone to adverse impacts of future climate change and to SOC losses as well as sites promising high SOC gains on both the national and regional scale.

5.6. Reflecting on static soil properties in the model within a dynamic environment

As outlined in the introduction section of this thesis, a common challenge with simulations of SOC changes over time is a robust and sufficient consideration of soil propertiers, climatic variability, and management (Brilli et al., 2017; Congreves et al., 2016; Liu et al., 2017; Smith et al., 2010). Predictions and deductions of key influential factors based exclusively on field data may, however, be complicated if the management is not standardised, i.e. if there is large within-site variability and a lack of replication of management practices across multiple contrasting sites, as was the case in the Hainich region (see also Blüthgen et al., 2012). For these reasons, in this thesis in chapters 3 and 4, it was opted to keep the management operations constant and unchanging from year to year within each future simulation run, while the temporal variability was reflected by the climate data. Variability due to management in turn was represented by replicating each site and climate scenario with multiple contrasting management regimes. In the simulations performed for this thesis the soil properties were also conceptualised as static in the CANDY model. This is reliable particularly for the grain size composition, and thus texture, due to being largely stable on the long term, unless subjected to heavy erosion by tillage or by wind and water in case of bare phases witout vegetation cover, which in turn are not typical for grassland (Abbasi et al., 2007; Başaran et al., 2008; DuPont et al., 2010; Wang et al., 2012; Zhang et al., 2013). However, the BD, and therewith associated hydraulic properties, typically experieces fluctuations under field conditions, e.g. due to seasonal swelling and shrinking of the soil due to wettness or freeze-thaw cycles, due to compaction by cattle trampling and field traffic, or by loosening by burrowing activities of soil biota (Augeard et al., 2008; Franzluebbers et al., 2000;

Jozefowska et al., 2016; Nemček-Korenkova and Urík, 2012; Wuest, 2015). While this variability was not accounted for directly in the CANDY simulations, by including it as a predictor in the statistical evaluation in chapter 4 it was possible to quantify the impact the BD has on the SOC sequestration potential in dependence of varying climate and management, and thus the uncertainty steming from the variability of BD across the entire dataset. Most importantly, this approach allowed identifying the tipping point of 15 M-% clay content, above which the susceptibilty of a soil to SOC losses due to increasing BD drastically increases.

5.7. Conceptualising the impact of management on soil biome and vegetation in simulations

One final consideration in the context of predicting SOC changes using simulation models pertains the soil microbiome and vegetation composition. Microorganisms, such as bacteria and fungi, dominate soil nutrient turnover (Filser et al., 2002; French et al., 2017; Strecker et al., 2016) and have a crucial effect on the soil's ability to sequester C (Gosling et al., 2017). Transformation of OM inputs and C losses are controlled by microbial biomass, since photosynthetically fixed C is eventually emitted as CO_2 through microbial respiration (Rees et al., 2005), which is why changes in microbial communities are an immediate indicator for changes in SOC stocks. The vegetation species composition and microbiome are thereby greatly affected by management (Filser et al., 2002). For instance, a perennial and diverse vegetation and continuous supply of plant residues promotes the development of microbial biomass, while monocultures and a high uptake of biomass and nutrients are adverse (Attard et al., 2016; DuPont et al., 2014; Jozefowska et al., 2016; Robertson et al., 1994; van Eekeren et al., 2008). However, most soil simulation models with a focus on C and N dynamics do not capture the influence of management on the soil microbiome, and the effect the microbiome exerts on the turnover and mineralisation of OM is typically represented indirectly, if at all, e.g. by concepts like the biologic active time (BAT) implemented in the CANDY model (Brilli et al., 2017; Franko et al., 1995; Franko et al., 1997). While such approaches are both useful and necessary to conceptualise complex interactions in the soil biome and their impact on the turnover conditions, they might present potential limitations. Chapter 4 argued that while the Hainich region is a promising candidate for SOC sequestration under a variety of future climate change scenarios, the impact of management intensification on the soil biome and biodiversity, which are key components of the terrestrial C cycle, could not be reflected on by the simulations, which might pose a potential conflict of interest in a region dedicated to the conservation of biodiversity. Developing modelling concepts and approaches that involve feedback-mechanisms between management on the one hand and biodiversity and

species composition on the other hand, and their corresponding impact on C dynamics, could increase the robustnes of simulation models (Brilli et al., 2017; Le Noë et al., 2023; Riedo et al., 2000). Recent modelling approaches tackle this by including one or several microbial organism pools that reflect on the activity of microbiota, though the required amount of pools and their impact on C turnover are an ongoing topic of research and discussion (Schimel, 2023).

5.8. References

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6. Conclusion

This thesis provided several strategies for farmers and decision makers aimed at improving chemical soil properties under arable and grassland while maintaining a variety of regulatory and provisioning ecosystem services, and furthermore contributed to national efforts aimed at increasing the SOC sequestration potential under grassland under a wide variety of climate scenarios, management options, and site properties, while also considering potential trade-offs with goals aimed at conservation of biodiversity.

The results generated based on simulations on both the national and regional scale substantiated the overall dominant effect of management practices on SOC dynamics, whereby moderate intenification of land use, particularly under management that at least partly includes grazing periods, was shown to be a viable method to increase SOC sequestration, provided sufficient rainfall amounts. The simulations performed in this thesis furthermore identified representative regions of high priority due to either high risks for SOC losses under suboptimal management or a potential for high SOC gains. This in turn may guide future decision-making in environmental policies. Particularly in dry regions the seasonality of rainfall will largely determine the effect of intensification of grassland use on the SOC gains, while cool, mid-altitude or alpine grassland regions with clay-rich soils will benefit from increasing temperatures under future climate by prolonging the vegetation period. Nevertheless, despite a high SOC sequestration potential, intensively-managed clay-rich sites were also most prone to SOC losses due to increasing soil compaction, substantiating the relevance of management practices aimed at avoiding overcompaction under such sites.

A full evaluation of both the benefits of intensification and specific differences between grazing and mowing management must consider trade-offs of intensification, such as potentially adverse impacts on biodiversity, as well as GHG emissions due to a higher requirement for fertiliser application. The robustness of future simulations on national and regional scales will greatly benefit from including a multitude of contrasting management options replicated under a wide variety of climate scenarios under different sites. Conceptualising the impact managagement practices exert on the biodiversity and the soil microbiome within simulation models may thereby both aid in evaluating potential trade-offs as well as increase the robustness of SOC predictions due to the key role the soil microbiome plays in the turnover of OM. Finally, the evaluation and comparison of independently performed studies on the impact of management practices on the sequestration potential would greatly benefit from the introduction of standardised indicators of management intensity that separately consider additional inputs of C and N, grazing intensity, and mowing intensity.

Publications and conference contributions

Peer-reviewed publications

Filipiak, M., Gabriel, D., Kuka, K., 2023. Simulation-based assessment of the soil organic carbon sequestration in grasslands in relation to management and climate change scenarios. Heliyon 9 (6), e17287. doi:10.1016/j.heliyon.2023.e17287.

Publications under peer-review

Carrilo Quijano, C., **Filipiak, M.**, Nieder, R., Stichnothe, H., 2024. Unlocking the Carbon Sequestration Potential of Agricultural Soils through Deep Tillage —A Dynamic Life Cycle Analysis Perspective. doi:10.21203/rs.3.rs-3910570/v1.

Filipiak, M.; Kuka, K. (2024): Assessment of the carbon sequestration potential in temperate grasslands of the Hainich Biodiversity Exploratory in dependence of future climate and management scenarios. Submitted to: Carbon Balance & Management.

Conference contributions – oral presentations

Filipiak, M., Mallast, J., Margraf, V., Stichnothe, H., 2021. Presentation of the Win-N project: N-utilisation efficiency and N-emissions when using inhibited ammonium sulphate urea. Annual conference THeKLa (Treibhausgasbilanzierung und Klimaschutz in der Landwirtschaft – Greenhouse gas balancing and climate protection in agriculture), 10.11.2023, web-conference.

Augustin, J., Döhler, J., Eißner, F., **Filipiak, M.**, Flessa, H., Hoffman, M., Klein, J., Klein, J., Klein, M., Kramp, K., Kreuter, T., Mallast, J., Margraf, V., Monzon Diaz, O., Pacholski, A., Pamperin, H., Rücknagel, J., Ruser, R., Schmid, W., Schuster, C., Skodras, D., Stichnothe, H., Wetzel, S., Zederer, D., Ziehe, D., 2023. Emission reduction potential for gaseous N compounds through the use of inhibitors. Interim status of the Win-N project. 3rd Annual Conference THeKLa (Treibhausgasbilanzierung und Klimaschutz in der Landwirtschaft – Greenhouse gas balancing and climate protection in agriculture), 28/29 June 2023, Braunschweig.

Declarations

1. I hereby declare in lieu of oath that this thesis has not been submitted to other examination authorities in the same or a modified form. I further declare that I have not applied for a doctoral degree at any other university.

   Hiermit erkläre ich an Eides statt, dass diese Arbeit weder in gleicher noch in abgewandelter Form bereits anderen Prüfungsbehörden vorgelegen hat. Weiter erkläre ich, dass ich mich an keiner anderen Hochschule um einen Doktorgrad beworben habe.

Göttingen, 11.01.2025

_____

(Matthias Robert Filipiak)

2. I hereby declare in lieu of oath that this dissertation was prepared independently and without unauthorised assistance. All passages that have been taken from other works in terms of wording or meaning have been identified as borrowings in each individual case by stating the source.

   Hiermit erkläre ich an Eides statt, dass diese Dissertation selbständig und ohne unerlaubte Hilfe angefertigt wurde. Sämtliche Stellen, die anderen Werken dem Wortlaut oder dem Sinn nach entnommen wurden, wurden in jedem einzelnen Fall durch Angabe der Quelle als Entlehnung kenntlich gemacht.

Göttingen, 11.01.2025

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(Matthias Robert Filipiak)

Supplementary Material to chapter 3

The supplementary material contains ambient climate data (supplementary material 1), climate models used for simulations (supplementary material 2), future climate data (supplementary material 3), summary of available data (supplementary material 4), soil properties of the scenario sites (supplementary material 5), summary of management regimes (supplementary material 6), correlation matrices for management- and climate data (supplementary material 7 and 8), residual plots of global and best candidate models (supplementary material 9, 10, 11), estimated changes in SOC per one-unit-increases of independent variables (supplementary material 12) and simulated changes in SOC from the CANDY model (supplementary material 13).

Supplementary material 1: Ambient (1990 – 2019) climate data for the 24 scenario sites as per Fig. 4, including climate station name and number as well as coordinates of the scenario site.

Displayed climate data are: Mean air temperature in 2 m (Temp., °C), mean annual precipitation sum (Prec., mm) and mean daily sunshine duration (Sun., h). The data was provided by the DWD in Brunswick.

Site nr.	Climate data			Station number and name	Coordinates (ETRS 89 Zone 32 N)	
	Temp.	Prec.	Sun.		Lat	Long
1	9.47	633.48	4.59	4625 Schwerin	5981232.791	600632.6635
2	9.38	525.12	4.75	1869 Grünow	5925692.302	838739.8694
3	9.40	783.15	4.24	4745 Soltau	5850022.054	561354.6638
4	10.30	739.44	4.38	1766 Münster/Osnabrück	5745383.778	408576.8236
5	9.88	676.08	4.36	691 Bremen	5905259.737	458924.1372
6	9.76	781.45	4.40	1975 Hamburg-Fuhlsbüttel	5944929.404	530520.4968
7	10.17	615.16	4.65	2928 Leipzig-Holzhausen	5668108.189	726361.7216
8	9.75	483.97	4.54	198 Artern	5666391.157	654474.1205
9	10.30	592.58	4.53	5424 Weimar-Schöndorf	5643160.975	659905.5265
10	8.66	704.22	4.34	2925 Leinefelde	5686619.661	587667.7663
11	7.70	713.32	4.42	2261 Hof	5597247.988	664123.618
12	10.71	802.16	4.33	2667 Köln-Bonn	5601235.436	396621.051
13	9.07	806.80	4.60	3287 Michelstadt-Vielbrunn	5557178.958	534909.6763
14	9.49	667.96	4.79	5440 Weißenburg-Emetzheim	5447302.341	635079.6951
15	9.58	637.51	4.56	282 Bamberg	5528471.917	664595.09
16	9.10	692.00	4.83	2700 Kösching	5412865.008	680353.7853
17	11.22	822.53	4.99	2812 Lahr	5341255.83	447780.0085
18	8.68	598.28	4.44	5397 Weiden	5494562.014	744560.497
19	9.92	563.34	4.73	5629 Wittenberg	5749759.486	737313.01
20	10.13	941.71	5.02	3379 München-Stadt	5326602.602	696929.7147
21	8.94	752.34	4.82	232 Augsburg	5372124.357	675712.8542
22	9.34	781.59	4.69	73 Aldersbach-Kriestorf	5375855.28	785682.9702
23	9.38	1072.34	4.70	4261 Rosenheim	5285367.136	722233.555
24	8.85	814.57	4.86	3366 Mühldorf	5298301.368	780954.5959

Supplementary material 2: Climate models used for acquiring future climate data. {rcp} denotes the corresponding designation of the RCP scenarios (e.g.: {rcp26}, {rcp45}, {rcp85}).

- ICHEC-EC-EARTH_{rcp}_r12i1p1_CLMcom-CCLM4-8-17_v1
- ICHEC-EC-EARTH_{rcp}_r12i1p1_KNMI-RACMO22E_v1
- ICHEC-EC-EARTH_{rcp}_r12i1p1_SMHI-RCA4_v1
- ICHEC-EC-EARTH_{rcp}_r3i1p1_DMI-HIRHAM5_v1
- MPI-M-MPI-ESM-LR_{rcp}1_r1i1p1_MPI-CSC-REMO2009_v1
- MPI-M-MPI-ESM-LR_{rcp}_r2i1p1_MPI-CSC-REMO2009_v1

Supplementary material 3: Future (2020 – 2099) climate data acquired for the 24 sites as per Fig. 4.

Climate station number, name as well as site coordinates as per supplementary material 1. Climate data averaged across all 6 projections per RCP scenario.With: Mean air temperature in 2 m (Temp., °C), mean annual precipitation sum (Prec., mm) and mean global radiation (Glob., J cm^{-2}). The data was provided by the DWD in Brunswick.

Site nr.	RCP 2.6			RCP 4.5			RCP 8.5		
	Temp.	Prec.	Glob.	Temp.	Prec.	Glob.	Temp.	Prec.	Glob.
1	9.48	785.49	946.39	9.94	795.38	934.30	10.71	815.65	920.02
2	9.39	534.60	987.86	9.88	549.94	974.34	10.70	562.90	961.47
3	9.86	813.98	926.03	10.32	818.77	913.90	11.09	841.10	900.56
4	10.76	808.81	951.50	11.23	813.79	939.43	11.97	826.69	927.50
5	10.16	784.72	917.59	10.61	787.88	906.90	11.35	803.80	891.58
6	9.84	811.16	928.35	10.30	819.22	917.14	11.05	838.71	902.17
7	10.30	589.68	1030.24	10.80	602.20	1015.54	11.63	617.48	1004.66
8	9.96	552.78	1012.51	10.45	561.86	997.72	11.27	579.92	987.63
9	9.20	646.29	1009.52	9.69	659.69	994.43	10.53	678.74	985.69
10	8.34	804.40	993.68	8.81	820.17	980.01	9.61	846.81	969.06
11	8.20	856.08	999.05	8.71	878.62	985.96	9.56	895.30	976.90
12	9.49	935.61	995.20	9.97	953.01	985.14	10.75	959.00	976.01
13	8.59	1021.94	1012.47	9.09	1032.24	1003.15	9.89	1051.41	994.53
14	9.12	756.25	1063.43	9.64	764.46	1055.73	10.49	781.77	1047.62
15	8.78	913.51	1034.12	9.29	922.48	1023.05	10.12	944.67	1014.28
16	8.98	756.50	1119.20	9.51	767.92	1112.21	10.37	788.25	1102.81
17	7.92	1358.66	1065.34	8.46	1356.38	1060.72	9.33	1365.24	1055.97
18	7.66	922.67	1034.63	8.18	933.02	1025.89	9.03	955.02	1015.52
19	10.37	603.34	1006.46	10.86	610.27	992.99	11.68	633.70	981.47
20	9.27	1078.61	1131.97	9.81	1085.20	1125.41	10.70	1110.29	1118.75
21	8.94	877.76	1138.05	9.48	882.80	1131.18	10.35	907.77	1122.61
22	9.33	899.40	1128.89	9.87	909.87	1122.68	10.74	937.96	1112.90
23	6.90	1744.69	1090.67	7.46	1747.63	1084.50	8.41	1784.86	1080.10
24	7.47	1895.30	1090.31	8.03	1888.95	1083.72	8.98	1917.84	1077.48

Supplementary material 4a: Summary of total available climate data and share of missing data for the validation sites.

Availability of climate data is reported for daily values required across the entire simulation period, thus all properties share the same amount of data entries. Missing data were acquired from the DWD Climate Data Centre (CDC) (2021a, 2021b, 2021c).

Property	Precipitation	Air temperature	Sunshine duration	Total
Total data	234127	234127	234127	702381
% missing	5.2	2.0	24.7	10.6

Supplementary material 4b: Summary of total available management data and share of missing data for the validation sites.

With: sowing date (sow. date), organic fertilizer application date (OF date) and amount (OF amt.), mineral fertilizer application date (MF date) and amount (MF amt.), cutting date (cut. date) and yield (cut. yld), grazing duration (grz. dur.) and stocking density (grz. dens.). Availability of management data is reported for the total amount of management operations documented as either dates or annual amount of operations. Due to a high correlation of the annual number of cuts, yield and fertilizer input the missing data could be completed using long-term means and descriptive management documentations of the validation sites as well as data provided by the German fertilization ordinance (DüV, 2017).

Property	Sow. date	OF date	OF amt.	MF date	MF amt.	cut. date	cut. yld.	grz. dur.	grz. dens.	Total
Total data	99	401	401	795	795	1073	1073	769	769	6175
% missing	29.3	18.2	14.0	33.6	26.8	43.4	76.0	25.4	12.4	35.8

Supplementary material 4c: Summary of total available soil data and share of missing data.

With: rock content (rocks), soil organic carbon (SOC) content, silt and clay content, bulk density (BD), field capacity (FC), permanent wilting point (PWP), saturated hydraulic conductivity (Ks). Availability of soil data is reported per soil layer used for simulations, thus all properties share the same amount of total data entries. Missing data were acquired form the national soil map BÜK200 (BGR Boden, 2018).

Property	**Layer depth**	**Rocks**	**SOC**	**Silt**	**Clay**	**BD**	**FC**	**PWP**	**Ks**	**Total**
Total data	125	125	125	125	125	125	125	125	125	125
% missing	0	2.4	0	30.4	30.4	2.4	41.6	41.6	20	18.8

Supplementary material 5: Soil properties of the 24 scenario sites extracted from the BÜK 200 soil map.

Standard values were used for particle density and heat capacity. Wilting point, field capacity and saturated hydraulic conductivity were calculated with pedotransfer functions of the CANDY model as described by Franko et al. (1995), Franko (1996) and Franko et al. (1997). The soil classification is provided according to the WRB (IUS Working Group WRB, 2022) and the German classification guideline KA5 (Sponagel, 2005) in parentheses.

Site nr.	Lower boundary depth [dm]	Clay content [M-%]	Silt content [M-%]	Rock content [M-%]	Bulk density [g cm^{-3}]	Soil group (topsoil)	Soil classification: WRB (KA5)
1	3	10	25	6	1.5	Sand	Luvisol
	4	10	25	6	1.5		(Parabraunerde)
	7	21	45	6	1.9		
	12	21	22.5	6	1.7		
	20	21	22.5	6	1.7		
2	3	6.5	17.5	6	1.5	Sand	Luvisol
	4	10	25	6	1.7		(Parabraunerde)
	8	21	45	6	1.9		
	12	21	35	6	1.7		
	20	21	35	17.5	1.7		
3	3	4	32.5	6	1.5	Sand	Cambisol
	6	4	32.5	6	1.5		(Braunerde)
	20	2.5	5	6	1.7		
4	3	14.5	25	6	1.7	Loam	Planosol
	7	14.5	25	6	1.7		(Pseudogley)
	12	35	22.5	6	1.9		
	20	55	42.5	75	1.9		
5	2	40	57.5	0	1.5	Clay	Gleysol
	6	40	57.5	0	1.7		(Kleimarsch)
	11	40	57.5	0	1.7		
	20	40	57.5	0	1.7		
6	2	40	57.5	0	1.5	Clay	Gleysol
	6	40	57.5	0	1.7		(Kleimarsch)
	11	40	57.5	0	1.7		
	20	40	57.5	0	1.7		
7	2	12.5	57.5	0	1.5	Silt	Fluvisol
	9	12.5	57.5	0	1.7		(Vega)
	14	22.5	57.5	0	1.7		
	20	6.5	17.5	6	1.7		
	3	21	74	0	1.5	Silt	Chernozem
8	6	21	74	0	1.5		(Tschernozem)

	7	21	74	0	1.7		
	10	12.5	57.5	0	1.7		
	15	35	22.5	17.5	1.7		
9	3	30	50	17.5	1.7	Clay	Regosol
	5	45	30	17.5	1.9		(Pararendzina)
	10	45	30	75	1.9		
10	3	35	30	17.5	1.7	Clay	Kastanozem
	6	45	30	17.5	1.9		(Rendzina)
	10	45	30	0	1.9		
11	3	12.5	57.5	62.5	1.7	Silt	Cambisol
	6	22.5	57.5	62.5	1.7		(Braunerde)
	9	22.5	57.5	62.5	1.9		
	12	12.5	45	75	1.9		
	20	10	25	75	1.9		
12	1	14.5	76.5	17.5	1.5	Silt	Cambisol
	3	14.5	76.5	17.5	1.7		(Braunerde)
	5	22.5	57.5	17.5	1.7		
	10	12.5	57.5	62.5	1.7		
13	3	10	25	6	1.5	Sand	Cambisol
	5	10	25	17.5	1.7		(Braunerde)
	10	6.5	17.5	37.5	1.7		
14	3	6.5	17.5	0	1.5	Sand	Arenosol
	6	6.5	17.5	0	1.7		(Braunerde)
	10	30	40	0	1.9		
15	3	30	50	17.5	1.7	Clay	Leptosol
	4	45	30	17.5	1.7		(Terra fusca)
	8	65	5	17.5	1.7		
	12	65	5	75	1.7		
16	1	12.5	57.5	6	1.3	Silt	Cambisol
	4	12.5	57.5	6	1.3		(Braunerde)
	8	21	45	6	1.7		
	10	35	22.5	62.5	1.7		
	15	35	22.5	75	1.7		
17	1	21	22.5	17.5	1.5	Loam	Cambisol
	2	21	22.5	17.5	1.5		(Braunerde)
	4	21	22.5	37.5	1.7		
	6	21	22.5	37.5	1.7		
	10	10	25	62.5	1.7		
18	3	6.5	17.5	17.5	1.3	Sand	Arenosol
	6	10	25	17.5	1.5		(Braunerde)
	8	2.5	17.5	37.5	1.5		
	13	2.5	17.5	37.5	1.7		
19	3	14.5	25	1	1.5	Loam	Fluvisol
	6	14.5	25	1	1.5		(Vega)

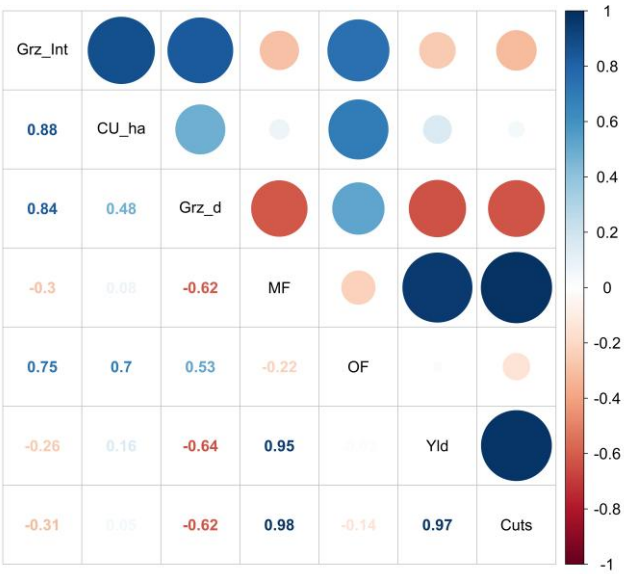
	13	21	22.5	1	1.7		
	20	21	22.5	1	1.9		
20	2	21	45	17.5	1.5	Loam	Luvisol
	4	40	40	37.5	1.7		(Parabraunerde)
	20	4	32.5	75	1.7		
21	3	21	45	6	1.7	Loam	Cambisol
	9	21	35	6	1.7		(Braunerde)
	10	14.5	25	6	1.7		
	20	10	25	6	1.7		
22	2	30	40	0	1.7	Loam	Cambisol
	4	40	40	0	1.7		(Braunerde)
	8	40	57.5	0	1.9		
	12	40	57.5	0	2.1		
	20	30	70	0	1.9		
23	1	4	32.5	37.5	1.5	Sand	Leptosol
	20	4	45	62.5	1.7		(Rendzina)
24	1	40	40	6	1.3	Clay	Cambisol
	3	40	40	6	1.5		(Braunerde)
	5	55	42.5	1	1.5		
	20	40	57.5	17.5	1.9		

***Supplementary material** 6: Summary description of management regimes developed from management data from the long-term soil survey sites and remote sensing data by Griffiths et al. (2018; 2019; 2020).*

The management operations are repeated for every year within one regime. All ten regimes were applied on all scenario sites and under all climate scenarios. Mineral fertilizer input was assumed in the form of calcium ammonium nitrate (CAN), organic fertilization in form of cattle manure with a nitrogen content of 0.46 kg N ha-1 (LWK Niedersachsen, 2018). In accordance with the DüV (2017), the annual application rate of organic fertilizers was limited to a maximum of 170 kg N ha.1, which corresponds do 370 dt FM ha-1. In order to ensure a comparability of the grazing scenarios, the grazing intensity, herein defined as the product of grazing days and number of cattle units per hectare, is kept at a constant of 50 CU ha-1 d.

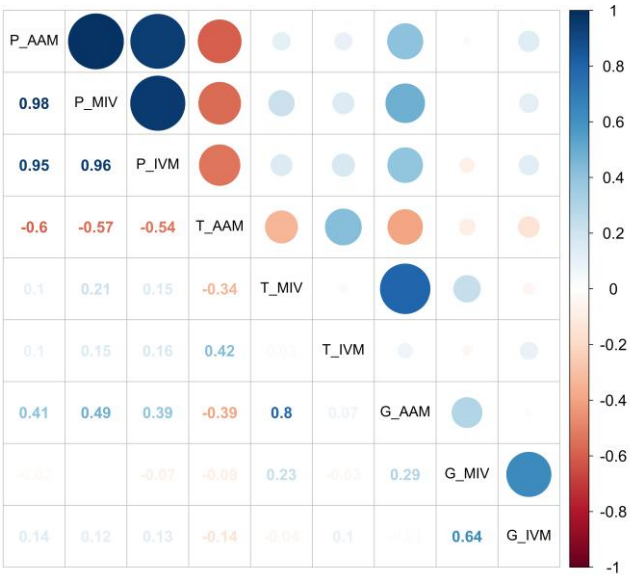
Regime	Management practice	Quantity	Date
1 pasture	Start grazing End grazing	0.2 CU ha^{-1}	01.04. 30.11.
2 meadow	Mineral fertilization Mowing	55 kg N ha^{-1} 40 dt DM ha^{-1}	01.04. 01.08.
3 mown pasture	Organic fertilization Mowing Start grazing End grazing	220 dt FM ha^{-1} 40 dt DM ha^{-1} 0.25 CU ha^{-1} 0.25 CU ha^{-1}	01.04. 01.05. 01.06. 30.11.
4 meadow	Mineral fertilization Mowing Mineral fertilization Mowing	65 kg N ha^{-1} 42 dt DM ha^{-1} 35 kg N ha^{-1} 13 dt DM ha^{-1}	01.04. 01.05. 15.06. 01.09.
5 mown pasture	Mineral fertilization Mowing Organic fertilization Mowing Start grazing End grazing	125 kg N ha^{-1} 35 dt DM ha^{-1} 140 dt FM ha^{-1} 20 dt DM ha^{-1} 0.4 CU ha^{-1} 0.4 CU ha^{-1}	01.04. 01.05. 15.05. 01.07. 01.08. 30.11.
6 meadow	Mineral fertilization Mowing Mineral fertilization Mowing Mineral fertilization Mowing	125 kg N ha^{-1} 40 dt DM ha^{-1} 32.5 kg N ha^{-1} 20 dt DM ha^{-1} 32.5 kg N ha^{-1} 20 dt DM ha^{-1}	15.03. 01.05. 15.05. 01.07. 15.07. 01.09.
7 mown pasture	Mineral fertilization Mowing Mineral fertilization Mowing Organic fertilization Mowing Organic fertilization Start grazing End grazing	165 kg N ha^{-1} 40 dt DM ha^{-1} 25 kg N ha^{-1} 20 dt DM ha^{-1} 55 dt FM ha^{-1} 20 dt DM ha^{-1} 55 dt FM ha^{-1} 0.55 CU ha^{-1} 0.55 CU ha^{-1}	01.03. 15.04. 01.05. 15.06. 01.07. 01.08. 15.08. 01.09. 30.11.
8 meadow	Mineral fertilization Mowing Mineral fertilization	165 kg N ha^{-1} 45 dt DM ha^{-1} 40 kg N ha^{-1}	01.03. 15.04. 01.05.

	Mowing Mineral fertilization Mowing Mineral fertilization Mowing	25 dt DM ha^{-1} 20 kg N ha^{-1} 10 dt DM ha^{-1} 20 kg N ha^{-1} 10 dt DM ha^{-1}	15.06. 01.07. 01.08. 15.08. 01.10.
9 mown pasture	Mineral fertilization Mowing Mineral fertilization Mowing Mineral fertilization Mowing Organic fertilization Mowing Organic fertilization Start grazing End grazing	200 kg N ha^{-1} 75 dt DM ha^{-1} 27.5 kg N ha^{-1} 20 dt DM ha^{-1} 27.5 kg N ha^{-1} 10 dt DM ha^{-1} 60 dt FM ha^{-1} 5 dt DM ha^{-1} 60 dt FM ha^{-1} 0.65 CU ha^{-1} 0.65 CU ha^{-1}	01.03. 01.04. 15.04. 15.05. 01.06. 01.07. 15.07. 15.08. 01.09. 15.09. 30.11.
10 meadow	Mineral fertilization Mowing Mineral fertilization Mowing Mineral fertilization Mowing Mineral fertilization Mowing Mineral fertilization Mowing	210 kg N ha^{-1} 70 dt DM ha^{-1} 25 kg N ha^{-1} 20 dt DM ha^{-1} 25 kg N ha^{-1} 10 dt DM ha^{-1} 25 kg N ha^{-1} 5 dt DM ha^{-1} 25 kg N ha^{-1} 5 dt DM ha^{-1}	01.03. 01.04. 15.04. 15.05. 01.06. 01.07. 15.07. 15.08. 01.09. 15.10.



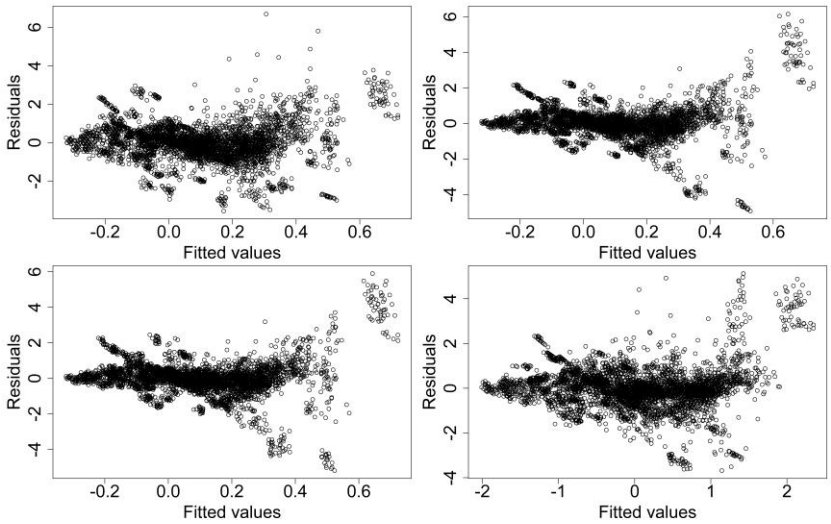
Supplementary material 7: Correlation matrix of management data (n = 70) used for the simulations with the management regimes.

With: grazing intensity (Grz_Int), stocking density (CU_ha), duration of grazing period (Grz_d), annual mineral fertilizer input (MF), annual organic fertilizer input (OF), annual yield (Yld), annual number of cuts (Cuts).

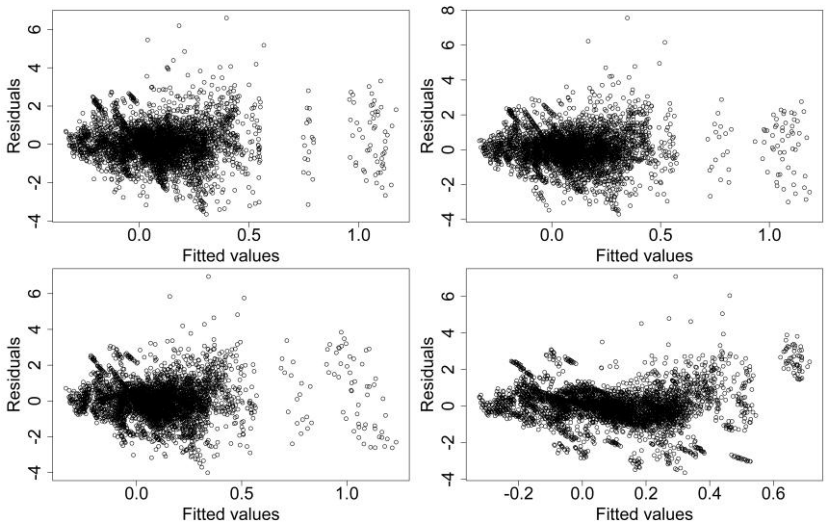


Supplementary material 8: Correlation matrix (n = 3888) of climate data.

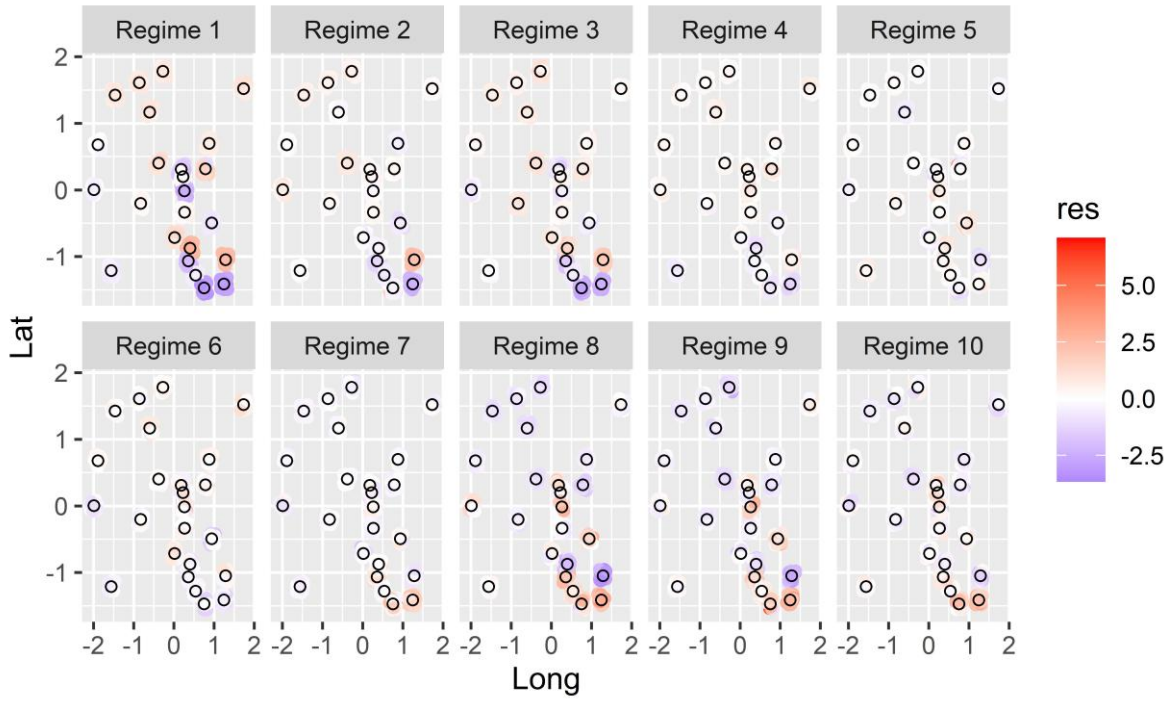
With: precipitation (P), temperature (T) and global radiation (G) aggregated by average annual means/sums (AAM), mean intra-annual variability (MIV) and inter-annual variance of means (IVM).



Supplementary material 9: Residual plots of GM-SP with variance structures according to soil group (top left), RCP scenario (top right), climate projection (bottom left) and management regime (bottom right).



Supplementary material 10: Residual plots of the best candidate models: CM-AAM-1 (top left), CM-MIV-1 (top right), CM-IVM-1 (bottom left) and CM-SP-1 (bottom right).



Supplementary material 11: Residuals of CM-SP-1 (circles with colour gradient), grouped by management regimes (1 – 10).

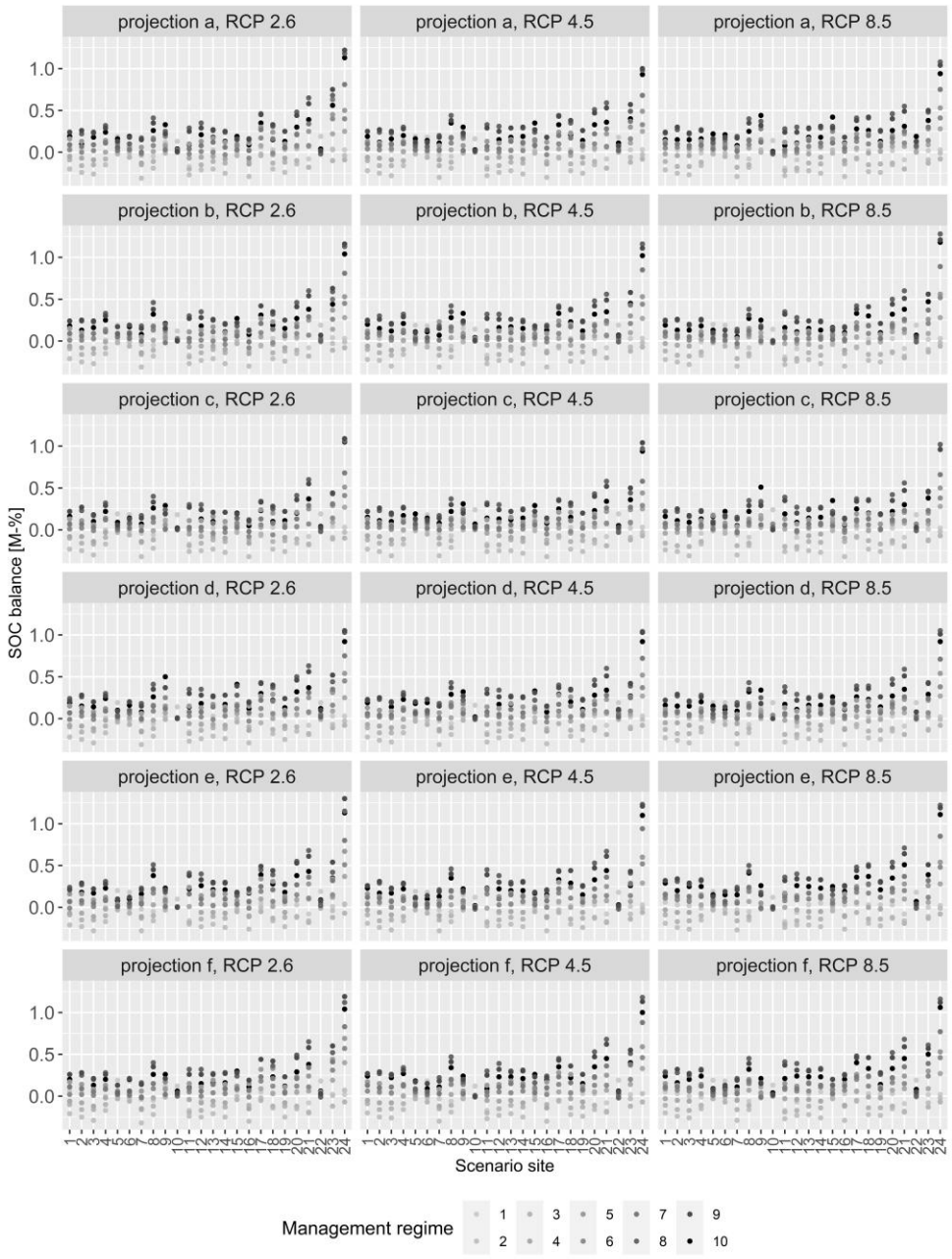
Each black-outlined circle represents one of the 24 sites with 18 simulation results (3 RCP scenarios * 6 projections). The coordinates (longitude and latitude) of the residuals were scaled and modified by a small, random number in order to avoid visual overlap as well as issues related to null-distances when calculating Moran's I at the site level. The graph shows a spatial clustering of sites with similar residuals, implying some spatial autocorrelation.

S12: Estimated change in SOC [M-%] per unit increase of the MIV of air temperature [°C], precipitation [mm], and global radiation [J cm^{-2}] for four soil types and ten management regimes (based on CM-MIV-1 model).

Management regime	Soil group	SOC change [M-%] per one unit increase of MIV of		
		Air temperature	Precipitation	Global radiation
1	Clays	0.02721	-0.05756	0.00024
	Loams	0.10145	-0.03281	-0.00012
	Sands	-0.01342	-0.02823	0.00010
	Silts	0.09119	0.02104	0.00093
2	Clays	0.09157	-0.04202	0.00027
	Loams	0.05735	0.00294	0.00009
	Sands	-0.01594	0.01068	0.00032
	Silts	-0.05617	0.02320	0.00012
3	Clays	0.09980	-0.04961	0.00020
	Loams	0.04483	-0.02245	-0.00016
	Sands	-0.03338	-0.01914	0.00011
	Silts	0.05579	0.02308	0.00052
4	Clays	0.10018	0.00159	0.00023
	Loams	0.03934	-0.01112	0.00020
	Sands	-0.00447	0.00831	0.00019
	Silts	-0.06032	0.02016	-0.00017
5	Clays	0.07983	0.04954	0.00019
	Loams	-0.00875	0.00781	0.00013
	Sands	0.04469	0.01128	0.00051
	Silts	0.02222	0.04469	0.00046
6	Clays	0.14227	0.01636	-0.00019
	Loams	-0.00461	-0.02108	0.00020
	Sands	-0.01249	0.00534	0.00049
	Silts	-0.03755	0.01005	0.00030
7	Clays	0.06116	0.09877	0.00024
	Loams	0.01673	-0.00324	0.00059
	Sands	0.02216	0.02951	0.00060
	Silts	-0.02249	0.01318	0.00046
8	Clays	0.16949	0.13374	-0.00014
	Loams	-0.04017	0.01882	0.00074
	Sands	0.05814	0.02508	0.00076
	Silts	-0.15721	0.01318	-0.00008
9	Clays	0.09879	0.13454	0.00019
	Loams	-0.02125	0.01108	0.00064
	Sands	0.07005	0.02786	0.00068
	Silts	-0.08815	0.02637	0.00012
10	Clays	0.17561	**0.10450**	-0.00025
	Loams	-0.03205	**0.00899**	0.00067
	Sands	-0.02412	**0.02360**	0.00073
	Silts	-0.10161	**-0.02166**	0.00040

Supplementary material 13: Simulated change in SOC [M-%] acquired from the CANDY model in dependence of climate projection (model designations according to supplementary material S2) and RCP scenario.

With: management regimes as greyscale. Climate projections change row-wise, RCP scenarios column-wise. The 24 scenario sites are spread along the x-axis, numbered from left to right in accordance to Fig. 4. The graph shows a) visible differences in the SOC balance between different sites, b) small differences between different climate projections and c) large differences between management regimes, thereby substantiating a low uncertainty due to climate data.



Supplementary material to chapter 4

The supplementary material contains information on climate models used for future climate scenario data (supplementary material 1), parametrisation of fertilisers used for simulation (supplementary material 2), additional background information on the management data used to develop the management regimes (supplementary material 3, 4, 5), summary of climate metric of future climate scenarios (supplementary material 6), estimated changes in the SOC balance per one-unit-increases of selected properties (supplementary material 7, 8, 11), visualisations of effects of mean air temperature and precipitation on the SOC balance (supplementary material 9, 10), and a visualisation of the simulate SOC balance in dependence of clay content, climate scenarios and climate projection (supplementary material 12).

Supplementary material 1: Climate models used for acquiring future climate data.

The {rcp} denotation corresponds to the RCP scenarios (e.g.: {rcp26}, {rcp45}, {rcp85}). Each model provided a unique climate projection under each RCP scenario.

a) ICHEC-EC-EARTH_{rcp}_r12i1p1_CLMcom-CCLM4-8-17_v1

b) ICHEC-EC-EARTH_{rcp}_r12i1p1_KNMI-RACMO22E_v1

c) ICHEC-EC-EARTH_{rcp}_r12i1p1_SMHI-RCA4_v1

d) ICHEC-EC-EARTH_{rcp}_r3i1p1_DMI-HIRHAM5_v1

e) MPI-M-MPI-ESM-LR_{rcp}1_r1i1p1_MPI-CSC-REMO2009_v1

f) MPI-M-MPI-ESM-LR_{rcp}_r2i1p1_MPI-CSC-REMO2009_v1

Supplementary material 2: Properties of the organic fertilisers as parametrised in the CDYOPSPA table in the CANDY database.

Fertiliser type	Dry matter content [M-%]	C content in dry matter [M-%]	CN ratio in organic matter	Total CN ratio	Reference
cattle slurry	8	34	16	7.77	[63]
pig slurry	8	31	13	4.66	
mixed slurry (pig + cattle)	8	33	14.64	5.89	
cattle manure	25	31	18	14.07	
sheep manure	30	38	17.67	13.62	
digestate	5.8	39.2	7.26	4.97	[64]

Supplementary material 3: Selected descriptive statistics for the annual yield across all Hainich sites.

Property	Original management data					DüV expected yield	Scenario yield	Scenario N input
	Minimum	Maximum	Average	n				
Yield <1 cut [dt DM ha-1]	10	51.85	28.19	20		40	30	41
Yield <2 cuts [dt DM ha-1]	55.67	72	63.78	24		55	n. a.	n. a.
Yield <3 cuts [dt DM ha-1]	61.96	81.96	68.61	6		80	70	166
Yield <5 cuts [dt DM ha-1]	n. a.	n. a.	n. a.	0		110	100	282

The sites were grouped into those with up to one cut per year on average, one to up to two cuts and two to up to three cuts on average. No data is available for five cuts per year in the original data. The table also provides the expected yield according to the German Fertilisation Ordinance [65] corresponding to the respective number of cuts. It is evident that for one and three cuts per year on average, the original yield in the Hainich sites is approximately 10 dt ha-1 lower than typical according to [65]. The scenario yield, which correspond to the developed management regimes M1, M2 and M3 respectively, is therefore also 10 dt lower than the expected yield. The last column provides the N requirement according to [65] corresponding to the scenario yield.

Supplementary material 4: Selected descriptive statistics for grazing properties for all 50 Hainich grassland sites, determined per year.

Property	Minimum	Maximum	Average
Start grazing	April	November	July
End grazing	May	November	September
Grazing duration [d]	5	273	93
Stocking rate [LU ha-1]	0.01	4.35	1.11

Supplementary material 5: Properties of sites with less than one cut per year on average (n = 20).

Property	Minimum	Maximum	Average
Start grazing	April	August	June
End grazing	July	October	September
Grazing duration [d]	80	273	146
Stocking rate [LU ha-1]	0.14	4.35	1.70

Supplementary material 6: Summary of future climate metrics for the Hainich Biodiversity Exploratory.

RCP scenario	Projection	AAM			MIV			IVM		
		tmp	prc	glob	tmp	prc	glob	tmp	prc	glob
2.6	a	9.67	606.62	1009.08	7.36	3.43	753.96	0.73	86.52	40.61
	b	9.72	678.02	1036.53	7.17	3.77	781.52	0.76	104.61	49.14
	c	9.93	624.48	1016.12	7.34	3.65	692.23	0.75	107.37	31.34
	d	9.54	632.20	1015.52	7.24	3.64	735.42	0.67	112.04	40.43
	e	9.75	673.29	1002.52	6.99	3.84	803.48	0.68	101.84	44.88
	f	9.76	643.60	1017.12	7.16	3.70	815.75	0.65	99.72	47.37
4.5	a	10.24	628.36	994.27	7.30	3.64	750.62	0.82	123.07	45.13
	b	10.35	691.29	1030.86	7.15	3.92	783.37	0.88	100.97	46.32
	c	10.53	639.75	1010.98	7.23	3.73	691.63	0.91	105.24	28.63
	d	10.08	666.29	1004.03	7.15	3.78	734.16	0.72	104.16	40.03
	e	9.86	686.99	981.91	7.14	3.93	792.26	0.87	101.97	49.76
	f	10.09	657.97	987.29	6.93	3.78	805.96	0.84	101.88	46.13
8.5	a	11.00	647.56	983.15	7.18	3.95	745.59	1.28	112.27	44.92
	b	11.18	709.14	1016.97	7.00	4.11	774.52	1.40	132.60	54.24
	c	11.37	669.70	1001.74	7.15	3.97	688.89	1.39	109.67	33.23
	d	10.93	694.43	1006.25	7.12	4.21	740.03	1.14	125.89	47.53
	e	10.70	692.49	968.18	6.94	3.98	788.54	1.19	107.29	56.15
	f	10.80	709.52	969.75	7.02	4.13	800.88	1.18	107.69	50.00

With: average annual mean values (AAM), mean intra-annual variability (MIV) representing seasonality, and inter-annual variability (IVM) representing the long-term change for air temperature (tmp, °C), precipitation (prc, mm) and global radiation (glob, J cm-2). Projection designations correspond to supplementary material 1.

Supplementary material 7: Estimated change in the ambient SOC balance in dependence of the LUI mean and slope.

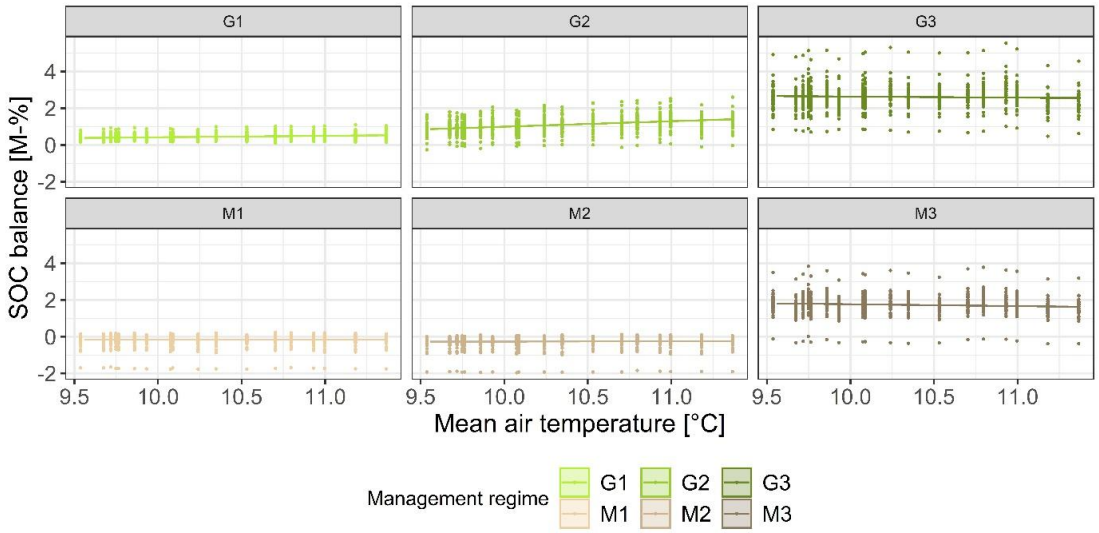
Level of LUI mean	Estimated change in SOC balance per one unit increase of the LUI slope
0.2	0.24
0.4	0.20
0.6	0.16
0.8	0.11
1.0	0.07
1.2	0.03
1.4	-0.01
1.6	-0.05
1.8	-0.09
2.0	-0.13
2.2	-0.17
2.4	-0.21
2.6	-0.25
2.8	-0.30
3.0	-0.34

Results presented as per one unit increases of the LUI slope (i.e. long-term intensification) for different levels of LUI mean across the 50 Hainich grassland sites in the ambient period (2008 – 2017, based on the best ambient CM).

Supplementary material 8: Estimated change in the future SOC balance in dependence of clay content and bulk density.

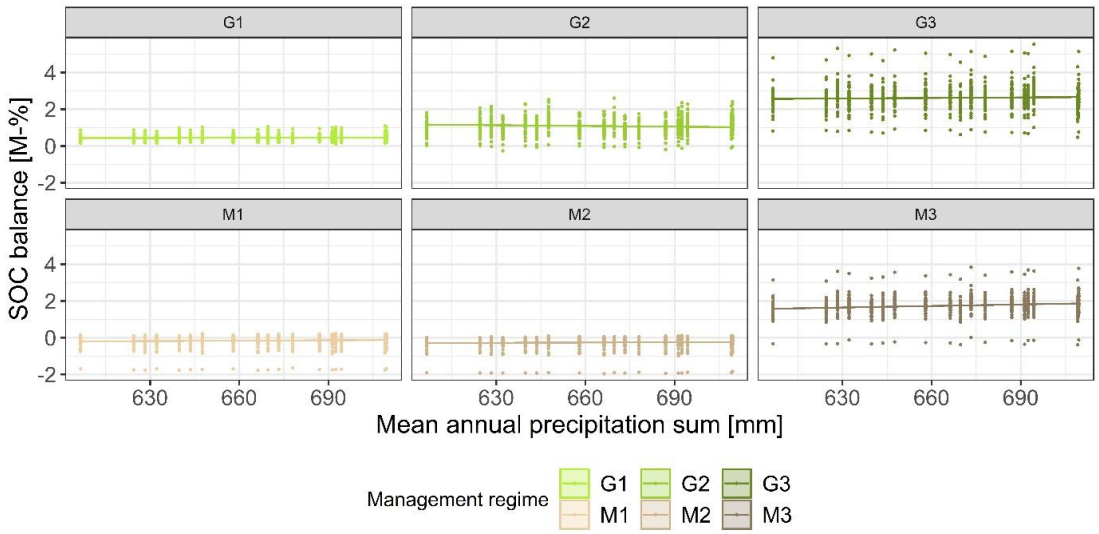
Level of soil clay content [M-%]	Estimated change in SOC balance [M-%] per one unit increase of soil bulk density [g cm-3]
5	0.25
10	0.14
15	0.03
20	-0.08
25	-0.19
30	-0.30
35	-0.41
40	-0.52
45	-0.63
50	-0.73
55	-0.84
60	-0.95
65	-1.06
70	-1.17

Results presented as per one unit increases of the soil bulk density for different levels of soil clay content across the 50 Hainich grassland sites in the ambient period (2008 – 2017, based on the best ambient CM).



Supplementary material 9: Linear regression (lines) of the effect of mean air temperature on the future SOC balance.

Effects were extracted from the best candidate model and grouped by the management regimes (G: grazing; M: mowing; 1: extensive; 2: moderate; 3: intensive). The SOC balance simulated with CANDY is additionally plotted (point features) for reference.



Supplementary material 10: Linear regression (lines) of the effect of annual precipitation sum on the future SOC balance.

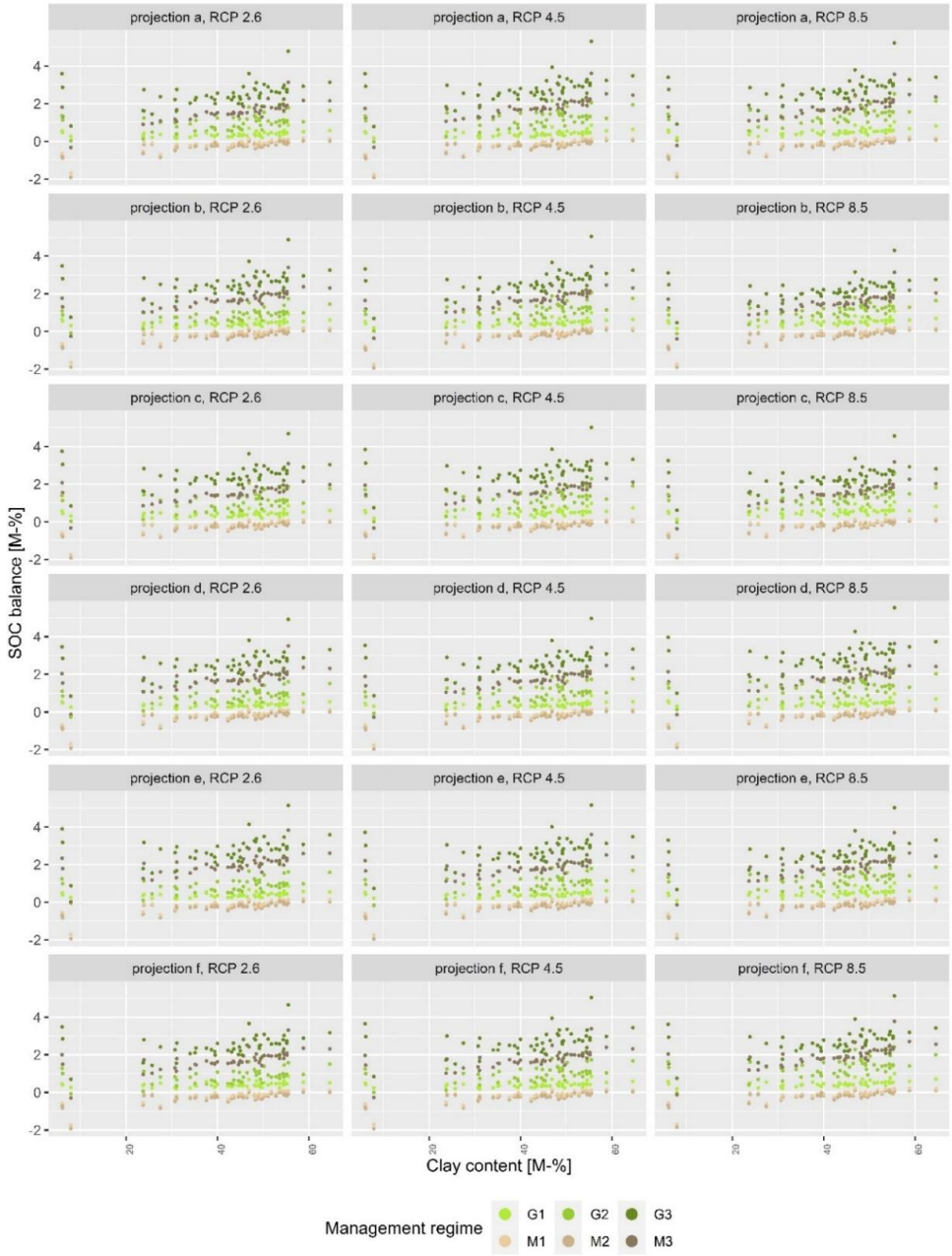
Effects were extracted from the best candidate model and grouped by the management regimes (G: grazing; M: mowing; 1: extensive; 2: moderate; 3: intensive). The SOC balance simulated with CANDY is additionally plotted (point features) for reference.

Supplementary material 11: Estimated change in the future SOC balance in dependence of initial SOC and precipitation sum.

Level of initial SOC content [M-%]	Estimated change in SOC balance [M-%] per 100 units increase of annual precipitation sum [mm]
3.53	0.0584
3.73	0.0595
3.93	0.0605
4.13	0.0616
4.33	0.0626
4.53	0.0637
4.73	0.0648
4.93	0.0658
5.13	0.0669
5.33	0.0679
5.53	0.0690

Results presented per one unit increase of the annual precipitation sum for different levels of the initial SOC content across the 50 Hainich grassland sites in the future period (2017 – 2100, based on the best future CM).

Supplementary material 12: Simulated change in future SOC contents in dependence of climate



Climate model designations according to supplementary material 1. Climate projections change row-wise, RCP scenarios column-wise. The topsoil clay content of the 50 Hainich grassland sites is spread along the x-axis. The graph shows 1) increasing SOC contents with increasing clay content, 2) visible differences in the SOC balance between different sites, 3) small differences between different climate projections as well as scenarios, and 3) large differences between management regimes, particularly grazing and mowing, thereby substantiating a low uncertainty due to climate data.