Received: 31 October 2023 | Revised: 24 March 2024 | Accepted: 19 April 2024

DOI: 10.1111/ejss.13493

REVIEW ARTICLE

European Journal of Soil Science WILEY

Qualitative evaluation of nine agricultural methods for increasing soil carbon storage in Norway

Alice E. Budai[1] | **Daniel P. Rasse**[1] | **Thomas Cottis**[2] | **Erik J. Joner**[1] | **Vegard Martinsen**[3] | **Adam O'Toole**[1] | **Hugh Riley**[1] | **Synnøve Rivedal**[1] | **Ievina Sturite**[1] | **Gunnhild Søgaard**[1] | **Simon Weldon**[1] | **Samson Øpstad**[1]

[1]Norwegian Institute of Bioeconomy Research (NIBIO), Ås, Norway

[2]Faculty of Applied Ecology, Agricultural Sciences and Biotechnology, Inland Norway University of Applied Sciences (HINN), Blæstad, Norway

[3]Faculty of Environmental Sciences and Natural Resource Management, Norwegian University of Life Sciences (NMBU), Ås, Norway

Correspondence
Alice E. Budai, Norwegian Institute of Bioeconomy Research (NIBIO), Ås, Norway.
Email: alice.budai@nibio.no

Funding information
Norges Forskningsråd; European Unions' Horizon 2020 research and innovation programme, Grant/Award Number: 862695 EJP SOIL

Abstract

Carbon content is a key property of soils with importance for all ecosystem functions. Measures to increase soil carbon storage are suggested with the aim to compensate for agricultural emissions. In Norway, where soils have relatively high carbon content because of the cold climate, adapting management practices that prevent the loss of carbon to the atmosphere in response to climate change is also important. This work presents an overview of the potential for carbon sequestration in Norway from a wide range of agricultural management practices and provides recommendations based on certainty in the reported potential, availability of the technology, and likelihood for implementation by farmers. In light of the high priority assigned to increased food production and degree of self-sufficiency in Norway, the following measures were considered: (1) utilization of organic resources, (2) use of biochar, (3) crop diversification and the use of cover crops, (4) use of plants with larger and deeper root systems, (5) improved management of meadows, (6) adaptive grazing of productive grasslands (7) managing grazing in extensive grasslands, (8) altered tillage practices, and (9) inversion of cultivated peat with mineral soil. From the options assessed, the use of cover crops scored well on all criteria evaluated, with a higher sequestration potential than previously estimated (0.2 Mt CO_2-equivalents annually). Biochar has the largest potential in Norway (0.9 Mt CO_2-equivalents annually, corresponding to 20% of Norwegian agricultural emissions and 2% of total national emissions), but its readiness level is not yet achieved despite interest from industry to apply this technology at large scale. Extensive grazing and the use of deep-rooted plants also have the potential for increasing carbon storage, but there is uncertainty regarding their implementation and the quantification of effects from adapting these measures. Based on the complexities of implementation and the expected impacts within a Norwegian context, promising options with substantial payoff are few. This work sheds light on the knowledge gaps remaining before the presented measures can be implemented.

KEYWORDS

biochar, carbon farming, cover crops, deep roots, extensive grazing, inversion of cultivated peat, meadow management, organic resources, reduced tillage, rotational grazing

1 | INTRODUCTION

There is a global interest in reducing greenhouse gas (GHG) emissions from the agricultural sector, while there is an increasing need for food. While several measures to reduce agricultural emissions imply reduction of food production, or have other negative side effects, increasing carbon storage in agricultural soils can have co-benefits for livelihoods, biodiversity, water provision and food security (IPCC, 2022).

In Norway, an agreement for a common effort towards reducing emissions and increasing removals in Norwegian agriculture was signed in June of 2019 by the Norwegian farmer organizations and the Norwegian government (Norges Bondelag et al., 2019). As a part of the agreement, the parties commit to reduce the net emissions from agriculture by 5 Mt CO_2-eq. from 2021 to 2030. Norway and the EU formalized a bilateral agreement in 2019 on cooperation in their joint effort to contribute to the fulfilment of the targets of the Paris Agreement (EEA Joint Committee, 2019). This agreement includes the LULUCF sector, and thus also soil carbon (C) sequestration in agricultural soils. Thus, measures that increase C sequestration in soil—or reduce soil organic carbon (SOC) loss—is included both under national and international agreements and targets for Norway.

Many agricultural management practices can potentially have a substantial effect on carbon sequestration in soil, such as soil amendments and improved use of organic resources (Bolinder et al., 2020; Lal, 2004; Nevalainen et al., 2022), biochar (Lehmann et al., 2006; Paustian et al., 2016), management of crop residues (Bolinder et al., 2020), use of cover crops (Bolinder et al., 2020; Lal, 2004; Nevalainen et al., 2022), crop rotation and the inclusion of legumes in the rotation (Lal, 2004), cultivation of species and genotypes with larger and deeper root systems (Button et al., 2022; Paustian et al., 2016), mixed farming with leys (Lal, 2004), improved cultivation of grasses (Bai & Cotrufo, 2022), managing grazing intensity (Bai & Cotrufo, 2022; Schmitz et al., 2018; Soussana & Lemaire, 2014), adaptive grazing (Bai & Cotrufo, 2022), subsoiling (Feng et al., 2020), reduced and no tillage (Lal, 2004; Smith et al., 1998), peat management (Joosten et al., 2015), agroforestry (Mayer et al., 2022), conservation regenerative agriculture (Freibauer et al., 2004) and irrigation (Lal, 2004). The efficacy of each of these methods is expected to vary considerably depending on the regional conditions where they are implemented. Therefore, carbon farming solutions based on improved agricultural management practices need to be defined for each region of Europe (Bumbiere et al., 2022). Answering this question for Norwegian conditions is the objective of the present study.

Highlights

- Carbon farming solutions based on improved agricultural management practices are presented for Norway.
- Use of cover crops scored well for certainty of effect, degree of maturity, and acceptance level.
- Biochar has the largest potential to sequester C, but its readiness level is not yet achieved.
- Knowledge gaps prevail for most of the methods presented.

Our general method was based on a two-step approach. In the first step, a list of methods for increasing/protecting soil carbon stocks, potentially suitable for Norwegian conditions, was established. In a second step, these methods were analysed qualitatively according to four criteria, as discussed in the next paragraph. For establishing the list of potentially suitable methods, we worked under three assumptions. First, we have exclusively considered methods that keep agricultural lands for food production. We have therefore not considered methods associated with changing land use, such as taking peatland out of production for rewilding or afforestation. Second, we have only considered methods that keep the structure of the Norwegian agricultural production. Norway has only 3% of its surface under arable lands, and in addition, a small area of intensive grasslands, amounting to approximately 3.5% of agricultural land in total (Norwegian Environmental Agency et al., 2023). In addition, wasteland areas in the outfields can be utilized for grazing. Approximately 45% of the land areas can be used for extensive grazing (Rekdal & Angeloff, 2021).

Out of the 3% of arable land area, about 1/3 is dedicated to cereal production, while nearly 2/3 is dedicated to managed grasslands (Norwegian Environmental Agency et al., 2023). Because of the limited area dedicated to cereal production in Norway, we have not considered converting cereal fields into grasslands. Such a measure would also result in higher CH_4 emissions from ruminants (Rasse et al., 2019). For the same reasons, we have not considered introducing more pluriannual leys into the cereal cropping regions of Norway, although this method is seen as quite relevant for countries with larger and more geographically distributed cereal production (Freibauer et al., 2004). Third, we have not considered methods that are specifically not practiced in Nordic conditions. Agroforestry, in particular, does not appear to be practiced in Nordic conditions, although it is extensively used and promoted in many other EU countries (Mayer et al., 2022). Fourth, we have tried to eliminate overlapping concepts from our analyses. For example, the concept of crop diversification includes ley farming and introduction of legumes in the rotation. Also, the concepts of regenerative and conservation agriculture are largely defined through the co-implementation of specific measures such as cover crops, reduced tillage, and rotations (Rasse et al., 2019). Based on current practices in Norway and promising new arenas (Rasse et al., 2019), we selected the following methods for further investigation: utilization of organic resources, use of biochar, crop diversification and the use of cover crops, use of plants with larger and deeper root systems, improved management of meadows, adaptive grazing of productive grasslands, managing grazing density in extensive grasslands, altered tillage practices, and inversion of cultivated peat.

The selected carbon farming methods were then qualitatively analysed according to four criteria, as suggested by Rasse et al. (2019). First criterium was the carbon sequestration potential following a realistic implementation of the method over the Norwegian territory. A carbon sequestration potential can be low either because the method is not well adapted to Norwegian conditions or because the method can only be implemented under very specific conditions applying to a limited cropping area. Second criterium was the probability of reaching the targeted C sequestration potential. This criterion is largely linked to the availability of data under the Nordic condition and our mechanistic and empirical understanding of the processes. Third criterium was the maturity of the technology, as some methods can be directly implemented, while others require further development. Fourth criterium was the adoptability of the method by the farming community, which is partially linked to perception and current availability of equipment at farm level. These criteria were summarized into advantages and limitations of each method in order to recommend priority measures for implementation as well as promising methods needing more research under Norwegian conditions.

2 | UTILIZATION OF ORGANIC RESOURCES

Most organic resources originate in some way from plant material produced on soil. In natural ecosystems, dead plant materials return to soil, resulting in a gradual build-up or maintenance of SOC. Within agriculture, large amounts of organic matter are removed from the fields during harvest and are often not reapplied. Practices like straw burning and prolonged winter fallows compound the negative effects of organic matter export and result in a depletion of organic matter stocks (Collins et al., 1992). This depletion is facilitated by tillage and fertilization practices that foster mineralization of organic matter in soil (Henriksen & Breland, 1999).

Restoring or maintaining soil organic carbon contents can be achieved either through enhancing plant growth or returning organic substrates to soil. The present section focuses on the latter, which is a complex issue, as organic residues represent a broad spectrum of products with varying maturity stages and properties with implications for their practical handling (storage, transport, spreading, etc.), dosage and regulatory limitations. An important aspect is also whether soil improvement with organic resources target a specific field or is used in, for example, a national scheme to enhance soil C storage. The former can freely accumulate organic matter from other areas without considering that these are further depleted, while national (or global or regional) considerations do not benefit from such shuffling of organic matter (Rasse et al., 2019). Rather the contrary: excessively high amounts of organic amendments may instead lead to higher mineralization rates, increased methane production or other undesirable effects (Shahbaz et al., 2017).

Among the major organic inputs to soil, manure is probably the most important and the most commonly used. Returning manure to soil has a long history and is still practiced in traditional ways worldwide. The last four decades have seen major changes regarding storage and spreading of manure, reducing losses of liquid and volatile fractions, as well as new uses of manure, for example, for biogas production or conversion to biochar. Apart from burning, all such practices result in organic inputs to soil which contribute to maintaining or increasing the SOC levels. Agricultural policy instruments and increased specialization in agriculture have resulted in a less even distribution of such inputs, as animal husbandry is often concentrated in specific regions of a country. This leads to overloading of the soils in these regions with manure derived nutrients, thereby causing environmental problems such as leaching, volatilization of ammonium and nitrous oxide emissions through denitrification (Menzi et al., 2010), highlighting that manure handling is not only a question of managing soil organic matter.

The extent to which manure amendments to soil maintain or enhance SOC storage is complex (Gross & Glaser, 2021), as the use of manure usually coincides with the cultivation of plants that contribute to soil C inputs. Long-term field experiments show that when cultivating crops like grain and oil seeds that provide small C inputs to soil, annual manure amendments can enhance SOC storage. Thus, Kirchmann et al. (2004) observed an increase in SOC after 42 years of manure amendments to a clay soil. Organic C content here rose from 1.50 to 2.25% in the top 20 cm after annual inputs of 2 t C ha^{-1} in crop rotations of grain crops (75%) and oil seed rape (25%). In Norway, a long-term field experiment comparing C storage in soil fertilized with two levels of either farmyard manure or NPK, showed increased C concentrations by 0.15 to 0.4% over a 56-year period, depending on the amounts of FYM supplied (110–330 kg dry matter/yr) (Riley & Kristoffersen, 2022). In the meta-analysis by Gross and Glaser (2021), SOC stocks increased most with pig and cattle manure application, followed by manure from poultry, sheep and horse. All manure types performed better than amendments with green manure or straw, both when expressed as response ratios (RRs; %) or as stock change rates (SCRs; kg C ha^{-1} $year^{-1}$). Similar conclusions were drawn by Bolinder et al. (2020).

Increased SOC storage is often obtained with the use of other soil inputs such as compost, sewage sludge, biosolids, and digestate from biogas production. These share a number of characteristics. Most relevant here is that they are commonly generated off-farm as a means of returning urban waste streams and the resources therein to agricultural land. Furthermore, they comprise a type of biomass that has been partly degraded and contain less readily degradable forms of carbon. Sewage sludge and biosolids have been extensively studied and shown to enhance SOC storage compared to soil not receiving such inputs (e.g., Rasmussen & Parton, 1994). Tian et al. (2013) observed that soils receiving biosolids over a 32-year period not only contained higher SOC stocks, but also had up to four times higher amounts of mineral-associated SOC, which are particularly stable against mineralization. Much of this is likely to apply also for digestate after biogas production based on manure and food waste. While the use of sewage sludge is regulated in Norway and limited to 2–4 Mg DM $year^{-1}$ depending on heavy metal content, the application of digestate is primarily limited by plant nutrient demands. For composts, their use frequently result in increased SOC, the magnitude of which depends on soil type, crop type, as well as amounts of compost and mineral fertilizers (Fan et al., 2014; Tautges et al., 2019). Compost and digestate from food waste are substrates for which volumes are increasing during the later years. For digestates based on manure, inputs to soil may decrease compared to direct use, while for digestate and composts based on food waste, amounts of substrate have seen an increase in later years, at the expense of combustion of domestic waste. As of 2023, all Norwegian municipalities are obliged to collect and recycle organic household wastes, potentially representing an enhanced flux of C returning to soil (up from approx. 240,000 t FW $year^{-1}$ to 450,000 t FW $year^{-1}$). Also exploiting fish sludge from aquaculture for biogas production has the potential for increasing organic substrate inputs to soil. This substrate stream represents upwards from 2 Mt $year^{-1}$, and the alternative fate of fish sludge practiced today is discarding it at sea.

The recommendations for how to manage straw after grain crop production has varied during the last decades, with burning and incorporation in soil representing the extremes. Burning clearly results in a decline in soil organic matter, while straw incorporation contributes to maintaining or slightly increasing SOC storage (Berhane et al., 2020; Powlson, Glendining, et al., 2011). Accordingly, burning of straw is thus generally discouraged or even banned (Freibauer et al., 2004). The negative impact of not returning straw to soil is worth keeping in mind in our times when the competition for feedstocks for feed, biomass and bioenergy is growing (Carvalho et al., 2017).

The potential for enhancing SOC storage based on organic substrates is limited, as much of these substrates are already used as soil amendments today. The exceptions are increased collection of food waste and treatment of fish sludge for use in agriculture. The latter is potentially a new, large source, but requires substantial technology development to cope with de-salting, de-watering and nutrient preservation. Research activities on these aspects are underway.

3 | USE OF BIOCHAR

The deployment of biochar technology for soil carbon sequestration in Norway is controlled by a series of key factors, mainly: documenting sequestration, availability of feedstock, trade-offs in reporting GHG emissions, technology readiness both for production and soil application, and eventually production costs.

Documenting biochar stability in soils is central to having it accepted as a C sequestration method in agriculture. Biochar is a stabilized form of C, mostly as condensed aromatic structures, produced from the pyrolysis of biomass. Its resistance to decomposition in soil is unique by being less dependent on ecosystem properties as compared to other forms of organic matter that are largely governed by accessibility to decomposer communities and stabilization processes such as organo-mineral complexation (Schmidt et al., 2011). Biochar stability is determined by feedstock properties and the degree of carbonization attained during pyrolysis. Pyrolysis is largely dependent on temperature but is governed by several other parameters, requiring a proxy to simply and reliably predict the stability of biochar in practice. The hydrogen-to-organic carbon molar ratio (H/C_{org}) of biochar was recommended as a proxy for estimating biochar stability only ten years ago (Budai et al., 2013), and recent work has focused on refining models using H/C_{org} for calculating the fraction of biochar C remaining in soil after 100 years (Rodrigues et al., 2023; Woolf et al., 2021). The overall C sequestration efficiency is a function of biochar carbon yield from pyrolysis and the stability of biochar in soil, and it is estimated that 25–50% of carbon in biomass remains in the soil after 100 years when converted to biochar (Rodrigues et al., 2023).

Main feedstock types available for biochar production in Norway are forest residues (branches and tops, 1.14 Mt year^{-1} total), byproducts from wood industry (0.56 Mt year^{-1}) (Tisserant et al., 2022), horse manure containing woodchips (0.5 Mt year^{-1}), and straw (0.49 Mt year^{-1}, based on projected demands from animal husbandry) (Byers et al., 2024). The technically usable fractions of these feedstocks are assumed to be around 40% of the provided values. Based on realistic extraction rates of forest residues, biochar would contribute to storing about 1 Mt CO_2 year^{-1} in Norwegian agricultural soils (Hagenbo et al., 2022; Tisserant et al., 2022). However, there are two categories of trade-offs to consider in order to translate this soil C sequestration value into negative emissions.

The first set of trade-offs are emissions associated with the process of producing and applying biochar. Through an extensive life cycle analysis, Tisserant et al. (2022) determined that emissions associated with biochar production and application in Norwegian soils would reduce the climate benefit of biochar by about 20%, that is, to about 0.8 Mt CO_2-eq per year. The extra emissions from biochar production are mostly from feedstock collection and transportation (very little from pyrolysis), and these emissions are much smaller than the amount of CO_2-equivalents stored in soil as a result of direct biochar C sequestration. However, considering scenarios for the co-production of heat and power during pyrolysis or the geological storage of the tar would greatly increase the climate benefits, by about 50% and 100% respectively.

The second set of trade-offs is specific to using forest residues and considers the loss of C from deadwood biomass and soil C from the forest where the feedstock has been collected. This effect is especially relevant in the initial decade of the biochar deployment scenario, reducing accounted climate benefits by more than 50% (Hagenbo et al., 2022). However, in the longer term, which is relevant for C sequestration projects, the effect becomes comparatively small, as the turnover rate of deadwood and soil organic matter is substantially faster than that of biochar. Straw and rapidly decomposing waste will generate greater climate benefits in the initial years, but such feedstocks are available in smaller quantities in Norway.

Implementing biochar for soil application in Norway requires that farmers see benefits either in the form of C subsidies or in the form of improved agronomy, notably yield effects. The most important factor influencing farmers' interest in using biochar is the price of the product as related to its agronomic benefits and impact on total farm income (Kvakkestad et al., 2022). The first field trial done in Norway testing the use of pure unmodified biochar showed minimal yield and soil improvement benefits (O'Toole et al., 2018). However later experiments where biochar was used in combination with nutrient rich biogas digestate showed promise in improving nitrogen (N) efficiency and yield of vegetables (O'Toole, 2021). A recent study in Norway (Weldon et al., 2023) confirmed that biochar can improve the composting process, but the biochar enriched compost did not improve plant yields as expected or as reported in other studies (Kammann et al., 2015). Developing biochar-based fertilizers with substantial agronomic benefits remains a technological challenge (Rasse et al., 2022), and for now the greatest incentive for farmers appears to be the payment of C credits or subsidies. However, current costs of biochar are too high for large-scale C credit deployment, that is, about 700 euros per ton (Prestvik & Lilleby, 2021) and current financial incentive plans from the Norwegian agricultural directorate are limited in their ability to subsidize farmers for biochar application to agricultural soils. Pyrolyzing waste streams that need disposing or sanitizing could greatly reduce costs, but the legal framework for their application to soil is unclear at this stage (Prestvik & Lilleby, 2021).

The challenges in scaling up biochar production and implementing its application have kept C sequestration through this method at a very low level compared to its full potential. In 2023, only about 700 tons of biochar were produced in Norway, equating to <0.1% of the available biomass mentioned above being utilized for biochar production (Byers et al., 2024). Several medium sized companies are already operating successful biochar facilities in Norway or manufacturing and exporting pyrolysis reactors. Feed-quality biochar produced in Norway has been shown to improve pig health and to be commercially profitable for both biochar producer and farmer (Norsk Biokullnettverk, 2023). Although feed char applications might only result in modest amounts of carbon sequestration, they will help sustain the development of a biochar industry in the short to medium term when carbon market prices are still too low to subsidize the full production costs of biochar.

4 | CROP DIVERSIFICATION AND THE USE OF COVER CROPS

Cover crops have recently been proposed as one of the most promising soil management options for increasing soil C sequestration in Norwegian cereal production systems (Rasse et al., 2019). These plants extend the period of CO_2 fixation and C transfer to soil through active photosynthesis in autumn. Proper stand establishment is a key factor for cover crops to significantly contribute to C sequestration after harvest of the main crop. For several decades, the main objective of using cover crops in Northern countries was to reduce N and phosphorus (P) leaching from soils (Bøe et al., 2019). For this reason, cover crops are generally referred to as catch crops in Norway. Cover crops are also used to protect the soil from erosion, especially in slopy areas. The short growing season in Norway limits the use of autumn-sown cover crops, with only 14% of cover crops having been sown after the main harvest in recent years while the rest were undersown (Byers et al., 2024). Growing cover crops together with the main crop also increases plant biomass production per unit area of land and hence C transfer to soil.

The effect of cover cropping on soil organic carbon (SOC) stocks has, however, not been fully explored for Norwegian conditions. Long-term field experiments in Sweden have shown that the use of perennial ryegrass as a cover crop can result in appreciable net C sequestration, with a mean yearly soil organic C stock increase of 0.32 ± 0.28 Mg C ha^{-1} in the plough layer (Poeplau et al., 2015). This is achieved with mean annual above-ground ryegrass dry weight of 0.56 to 1.1 Mg ha^{-1} year^{-1}. For comparison, the mean annual harvested dry weight of ryegrass grown as a catch crop together with cereals in Norway was found to vary between 0.81 Mg ha^{-1} (perennial ryegrass, variety "Trani") and 1.17 Mg ha^{-1} (Italian ryegrass, variety "Macho"; Molteberg et al., 2005). This suggests that approximately similar levels of C sequestration can be achieved with ryegrass cover crops under Norwegian conditions as those found in the Swedish long-term studies. An overall positive effect of cover crops on soil C stocks was also found in a meta-analysis covering data from 30 studies at 37 sites (where 76% were situated in the temperate zone; Poeplau & Don, 2015).

Bøe et al. (2019) estimated the potential of C sequestration in Norwegian cereal production areas using cover crops. Based on results from long-term experiments carried out in Sweden, it was assumed that cover crops under Norwegian climatic conditions can bind 0.32 Mg C ha^{-1} year^{-1}. In 2019, cover crops were grown on 2240 ha or <1% of total cereal area in Norway. Carbon sequestration by cover crops on this limited surface would then be 2240 ha x 0.32 Mg C ha^{-1} year^{-1} = 717 Mg C year^{-1} or 2464 Mg CO_2 year^{-1}. The total percentage of grain production area currently cultivated with cover crops is 4.5%, indicating an increasing trend (Byers et al., 2024). Planting cover crops on 20% of the cereal acreage, that is, on 57,014 ha, would result in the sequestration of 18,245 Mg C year^{-1} in the plough layer (Bøe et al., 2019). However, this estimate does not consider diminishing C-sequestration effects with time due to the establishment of a new equilibrium between supply and loss of organic matter. In addition, perennial grasses and clover species have been found to lose much of their shoot N throughout the winter season, notably as freezing–thawing cycles occur (Sturite et al., 2007). This can result in increased off-season N_2O emissions (Sturite et al., 2021). The net effect of cover crops on N_2O emissions, whether positive or negative, has been shown to be highly variable and site specific (Guenet et al., 2021). Thus, several cover crop species need to be tested under Norwegian winter conditions for effects on N_2O emissions, which might increase and offset the positive climate change mitigation effects of cover crops.

5 | USE OF PLANTS WITH LARGER AND DEEPER ROOT SYSTEMS

Carbon from root origin is stabilized in soil more than twice as efficiently as carbon from above-ground residues (Kätterer et al., 2011; Rasse et al., 2005). This suggests that the majority of recalcitrant C in soils is of root origin, especially in agricultural systems where large proportions of the above-ground biomass are often exported from fields (Rasse et al., 2005). In recent years, this key finding has translated into the hope of increasing C sequestration in soil through increasing root C inputs (Paustian et al., 2016). Such an effect could be obtained either by introducing crops with large root systems in the rotation or by developing cultivars with increased carbon allocation to roots.

Selecting crops or cultivars for enhancing the sequestration of root C in soils requires to understand the mechanisms leading to this preferential stabilization and the plant traits that favour these mechanisms. As compared to shoot-residue inputs, root inputs have a higher concentration of recalcitrant molecules, have more intimate contact with soil particles potentially fostering physical and physico-chemical stabilization and partly happen in the deeper soil where microbial activities are restricted (Rasse et al., 2005). Higher concentration of recalcitrant molecules does not appear to play a major role in the preferential stabilization of root-C in soils (Rasse et al., 2005; Sokol et al., 2019). Root architecture appears to have little impact as well (Poirier et al., 2018). The role of exudation remains difficult to evaluate. Root exudates appear to contribute disproportionately to the mineral associated organic matter in soils (Villarino et al., 2021), but their actual net contribution to soil C remains uncertain, especially in croplands (Panchal et al., 2022). In addition to more root biomass, the main factor shown to foster root-C stabilization in soil is root depth (Poirier et al., 2018). This suggests that using deeper rooting plants and making the deep soil more hospitable to root growth are key management options for increasing C sequestration in soils (Button et al., 2022). For example, alfalfa with a deep root system has been shown to be more efficient at storing C in soils than grass species with shallower rooting systems (Saliendra et al., 2018).

The demand for key agricultural products such as cereals will remain high, which suggests that the largest potential for increasing C sequestration lies with the development of root-enhanced cultivars of main crops (Paustian et al., 2016). Many genes controlling root traits have been identified, such as those associated with root exudation, root elongation, root depth, synthesis of recalcitrant lignin and suberin molecules (Yang et al., 2021). In Norway, however, the use of gene editing for agricultural crops is unlikely in the foreseeable future, and advances will likely come from traditional variety testing. A key question is whether large and deep root systems can be developed without negatively impacting above-ground growth and yields (Eissenstat, 1997). Although the answer to this question is probably specific to plant types and soil environmental conditions, win-win applications certainly exist. For example, in the case of water or nutrient limitations, soybean varieties with larger root systems have been shown to produce higher yields than varieties with smaller root systems (He et al., 2021). Recently, Heinemann et al. (2023) have concluded from a meta-analysis of published data that variety selection can increase the size of root systems of wheat, rapeseed, maize and sunflower by 7% to 26% without decreasing yields.

The type of enhanced root system best adapted to Norwegian conditions remains to be determined. Deep root systems are well adapted to capturing easily leached nutrients, especially nitrates (van der Bom et al., 2020), and variety testing for this trait is currently underway, notably for wheat (Wacker et al., 2022). This might be an advantage under high-precipitation environments such as in large parts of Norway, possibly combining increased C storage with nutrient recovery. However, the deeper soil is often unhospitable to root growth, notably due to acidic conditions linked to Al and Mn toxicity, insufficient aeration, high compaction, and low temperatures during the growing season (Lynch & Wojciechowski, 2015). Although plough layers of Norwegian agricultural soils have comparatively low pH in Nordic regions as compared to many places in Europe (Reinemann et al., 2014), there is insufficient data on deeper soil conditions to assess the extent of acidity limitations to root growth. With the exception of installing drainage pipes, few management options are available for improving the deep soil, and it has been argued that breeding plants with tolerant root systems is a more viable option (van der Bom et al., 2020).

Verification of C sequestration gains is especially challenging in the deep soil, as sampling is difficult to conduct and costly. However, flux-based MRVs of C sequestration, such as eddy-covariance, are best reconciled with measured C accumulation rates when the entire soil profile is sampled (Smith et al., 2020). This underlies that deeper soil C contributions are substantial and should be fostered notably through adapted varieties. Testing the performance of wheat varieties with different size and depth of rooting systems is currently underway in Norway and in other European countries through the MaxRoot C project (https://ejpsoil.eu/soil-research/maxroot-c). More field testing will be needed to understand the response of root-driven C sequestration to interactions between genotype-driven root development and soil environmental conditions.

6 | IMPROVED MANAGEMENT OF MEADOWS

In grasslands, the high density of roots results in considerable C inputs, which is believed to maintain or even build up SOC (Katterer & Bolinder, 2022). Results from long-term experiments in Sweden and Denmark demonstrate that inclusion of leys in crop rotation increases C sequestration (Bolinder et al., 2010; Christensen et al., 2009; Jarvis et al., 2017). However, the ley period must be long enough to reach this effect. In Sweden, the inclusion of 5 years of grass-clover ley in crop rotation showed the best potential to maintain SOC storage over time (Bolinder et al., 2010). By contrast, including leys for only one or two years in the rotation was too short to build up SOC and even showed net loss on some occasions (Bolinder et al., 2010).

More than 60% of the agricultural area in Norway is used for forage production (Norwegian Environmental Agency et al., 2023). Grassland farming on arable land is often located in marginal areas with unfavourable weather conditions resulting in limiting ploughing and reseeding activities. Here, farmers tend to prolong grassland cultivation for at least 10 years (Lunnan & Todnem, 2014). In addition, permanent grasslands are found throughout the country, from Norway's southernmost agricultural territory, along the mountainous west coast and all the way to Northern Norway. In addition, the prevalence of animal production systems provides significant C inputs to the grasslands in the form of manure (Riley & Bakkegard, 2006). Therefore, the SOC content in Norwegian grassland soils is often higher than in other parts of Europe (Reimann et al., 2014). Carbon storage in grassland depends on management practices, fertilization, inter-sowing of grasses and legumes, intensification of grazing and conservation tillage can increase soil C pools (Conant et al., 2017). The extent to which this can be achieved in Norway should be determined.

Long-term grassland experiments in Norway are conducted at three NIBIO stations—Særheim (58°47′ N 5°41′ E), Fureneset (61°18′ N 5°4′ E) and Svanhovd (69°27′ N 30°3′ E)—that have been maintained for almost 50 years. The experiments include treatments without tillage and treatments that are regularly ploughed and reseeded every third year. Quantification of SOC down to 60 cm showed no significant effect on the frequency of ploughing and reseeding at any of the experimental sites (Sturite et al., 2020). In the plot without tillage, SOC was stored mainly in the upper topsoil horizon (0–5 cm) and declined gradually with increasing soil depth, whilst in regularly ploughed plots C was distributed rather evenly down to 30 cm depth. The results from the Norwegian long-term trials are in contrast with previous studies of Soussana et al. (2004), which suggested that conversion of short-term to long-term grassland increased SOC accumulation with 0.3–0.4 Mg ha^{-1}. The longer growing season and warmer climate in Central Europe than in Northern Europe could explain the difference. Our findings concur with Linsler et al. (2013), who reported that sporadically renewing grassland has a negligible effect on SOC content. In Norway, initially high SOC concentrations (>3%) might restrict our ability to further increase C stocks, as recognized for other regions (Börjesson et al., 2018; Christensen, 1990). In our field experiments, absence of tillage and stand renewal did not significantly increase the SOC concentrations in the topsoil (0–5 cm and 5–20 cm) over a 32-year period (Table 1), which is consistent with conclusions from Smith (2014) that C accumulation in grasslands may reach equilibrium over time. Therefore, to further increase SOC in grassland-dominated farms can be a challenge. However, good agronomic practices like optimal soil water regulation, minimal soil compaction, restoration by sod-seeding, use of species-rich seed mixtures including grasses and legumes and species with different root systems can be among management practices that may influence C storage positively. Sufficient availability of plant nutrients maintains good plant production and thereby maintains or even builds up SOC (Fan et al., 2019; Whitehead et al., 2018).

7 | ADAPTIVE GRAZING OF PRODUCTIVE GRASSLANDS

Management of productive grasslands on arable land, through adaptive grazing, which is also known as adaptive multipaddock grazing (AMP), has been posited to produce increased carbon storage by local research stations and practitioners around the world. AMP grazing is defined as grazing involving multiple paddocks per herd, high animal densities, short periods of grazing, long grassland recovery periods, and higher stocking rates than are traditionally considered sustainable (Teague et al., 2011). The general aim of the method is to extend the time and efficiency of photosynthesis by keeping plants in the vegetative phase for optimal leaf and root growth by reducing plant stress from overgrazing and allowing sufficient time for cold hardening.

TABLE 1 Mean concentrations of SOC (%) in upper topsoil (0–5 cm) and topsoil (5–20 cm) horizons measured in 1986, 2009 and 2018/2019 for grassland not tilled since 1968 and 1974 at Særheim and Fureneset, respectively.

		SOC (%) at respective years		
Location	**Depth**	**1986**	**2009**	**2018/2019**
Furuneset	0–5 cm	9.1 ± 0.75	11.7 ± 0.58	10.7 ± 2.27
	5–20 cm	8.0 ± 1.20	9.6 ± 0.95	9.0 ± 1.49
Særheim	0–5 cm	8.3 ± 1.07	6.6 ± 1.33	8.6 ± 1.48
	5–20 cm	5.7 ± 1.08	4.8 ± 0.34	5.6 ± 1.09

Increasing the duration of the rest period relative to grazing time has been shown to produce measurable benefits on grasslands in the form of increased plant biomass and ground cover (McDonald et al., 2019). Furthermore, Mosier et al. (2022) found on average 16% more SOC in the A-horizon in AMP grazed compared to conventionally grazed lands, based on 10 grasslands in the southeast United States. AMP grazing has been likened to low-intensity continuous grazing, differing more from high-intensity continuous grazing (Hillenbrand et al., 2019). There are also indications that this practice represents a sustainable intensification method, since twice as many animals could graze according to AMP grazing without it resulting in lower carbon sequestration in the soil compared to continuous grazing in a study by Wang et al. (2015). The results from the above studies cannot be directly translated to the expected results of the method in Norway. A meta-analysis on grazing impacts in general, covering 164 sites across different countries and climatic zones (Abdalla et al., 2018), revealed a climate-dependent impact of grazing on SOC storage. Differences in SOC stocks in the comparison were thought to be mediated by interactions between temperature, precipitation and grazing intensity, and grazing level induced impacts on plant cover and plant species occurrences, while neither soil texture nor grassland characteristics significantly explained differences seen in SOC stocks.

Herbivore grazing in general has direct and indirect effects on ecosystem C storage (Taboada et al., 2011). Direct effects include changing the quality and quantity of organic matter inputs entering soils and the physical effects of trampling that alters soil physical properties such as temperature, water content and aggregation. Indirect effects are mediated by changes in plant species composition and net primary production which are important for the distribution of biomass and the fraction of C allocated above- and belowground (Schmitz et al., 2018; Taboada et al., 2011). Feedback between herbivore-induced changes in soil properties and plant community structure may alter nutrient cycling, important for C assimilation via photosynthesis, C losses via plant and microbial respiration, and stabilization or decomposition of soil organic matter (SOM) (Bai & Cotrufo, 2022; Bardgett & Wardle, 2003; Soussana & Lemaire, 2014). The effects of grazing on biogeochemical processes altering SOC storage also depend on ecosystem productivity, soil type and climate as well as herbivore feed selection and herbivore density (Bai & Cotrufo, 2022; McSherry & Ritchie, 2013; Schmitz et al., 2018; Taboada et al., 2011).

In Norway, there are about 75 farmers practicing various forms of AMP grazing, motivated by an interest to increase forage quality and production (Regenerative Agriculture Norway, personal communication with Anders Lerberg Kopstad). Despite it being practiced and conveyed among farmers, there are no studies documenting soil carbon content in response to AMP grazing in Norway. Furthermore, it will take years before a carbon sequestration potential can be estimated for this practice because long-term experiments are required to demonstrate the length of time it would take for soil carbon stocks to reach a new equilibrium under the new management. AMP relies on frequent intervention by the farmer and varies greatly among individual operators, with grazing periods of paddocks in Canada ranging in length from hours to a few days, and resting periods generally lasting a little over two months (Bork et al., 2021). In a Norwegian context, recommended grazing times vary from one day up to a week, while recovery spans two weeks to two months, depending on factors such as temperature and moisture (Regenerative Agriculture Norway, personal communication with Anders Lerberg Kopstad). Due to the posited importance of grazing and recovery times, traditional experimentation methods that rely on pre-determined timeframes for rotating animals may not successfully demonstrate the potentials of AMP grazing (Teague et al., 2013). Therefore, scientific monitoring protocols (Xu et al., 2019) have been developed to enable a multi-level and farm-adapted approach for measuring change in ecosystem health outcomes of AMP-grazing on farm level over time. The use of such protocols may help evaluate the impacts of AMP grazing by taking farmer choices and heterogeneity of the climate and landscape across Norway into consideration.

8 | MANAGING GRAZING PRESSURE IN EXTENSIVE GRASSLANDS

In extensive grazing systems, animals are primarily fed on food from rangelands as compared to intensive grazing systems where animal feed mainly comes from seeded pastures (FAO, 1991). In Norway, where sheep is the most dominant livestock herbivore, animals generally graze outside on pastures from May to October and are kept inside the rest of the year (Austrheim et al., 2008). Since extensive grasslands and forest cover around 72% of the land (Norwegian Environmental Agency et al., 2023), farmers depend on rangelands during the summer as pastures (Austrheim et al., 2008). Due to the large areal extent, land use affecting C sequestration in these areas could have a significant impact on the national carbon budget (Bartlett et al., 2020).

Studies of different densities of sheep (80, 25 and 0 sheep km^{-2}) in Hol, Norway, suggest a density-dependent (non-linear) response of grazing on SOC and nutrient dynamics (Martinsen et al., 2011; Martinsen et al., 2012) as well as a strong impact on vegetation (Speed et al., 2010; Speed et al., 2015). Seven years of grazing resulted in less SOC in the soil O-horizon at sites with high density as compared to ungrazed sites, whereas a moderate density increased the SOC content in the O-horizon. The proportion of particulate organic matter (POM) associated with greater biomass production also increased in the moderate density (Austrheim et al., 2014), compared to both non-grazed and intensively grazed. POM is a labile pool of SOM consisting of partly decomposed plant and microbial residues and is considered as an indicator of recent changes in land use (Bai & Cotrufo, 2022; Leifeld & Fuhrer, 2009). In Hol, average stocks of SOC in grassland habitats were ~ 55 Mg ha^{-1} (8.5 cm depth) for O-horizons and ~ 24 Mg ha^{-1} (17.8 cm depth) for the mineral layer (Martinsen, 2011), comparable to those reported for grasslands in European mountain systems (80–100 Mg ha^{-1}, 0–30 cm depth at 600–1400 m a.s.l.) (Sjögersten et al., 2011). In Setesdal, Speed et al. (2014) studied effects of long-term exclusion of grazers by comparing areas with high grazing intensities (between 44 and 88 sheep km^{-2}) with areas not grazed for >50 years. Long-term exclusion of sheep resulted in establishment of birch forests and a significant increase in above-ground C storage. SOC stocks were greater in areas with long-term absence of sheep (~210 Mg ha^{-1}) than in those with high grazing intensities (~130 Mg ha^{-1}). The main differences in SOC storage, however, were attributed to intrinsic differences in the thickness of O-horizons (21.5 and 15 cm, respectively) as well as the presence of Histosols at the sites with long-term absence of sheep, rather than changes in grazing patterns and changed vegetation composition (Speed et al., 2014). In a study from Trøndelag, Kolstad et al. (2017) reported effects of 8 years of exclusion of moose browsing on vegetation composition and various soil parameters. The study showed that the cessation of moose browsing led to an increase in deciduous trees in relation to conifers. There was a significant increase in the thickness of the O-layer and a significant reduction in soil density and temperature when moose browsing ceased. However, they found no change in nutrient availability or SOC stocks between browsed and non-browsed areas. Total carbon stock (0–30 cm) was on average 87 Mg ha^{-1} (Kolstad et al., 2017).

Sheep grazing has an important impact on biodiversity and ecosystem services in Norwegian mountains and is a significant driver in sustaining cultural landscapes, for example, by preventing woody encroachment (Speed et al., 2010; Speed et al., 2014). Speed et al. (2015) looked at changes in ecosystem C stocks along an elevation gradient from the mountain birch forest (950 m a.s.l.) to above the tree line (1300 m a.s.l.) in Hol. The results showed a continuous reduction of C in the vegetation and a continuous increase in SOC in the O-horizon along the elevation gradient, with no significant changes in SOC stocks of the mineral soil. Reduction and increase in C stocks of the vegetation and O-horizon, respectively, led to decreasing ecosystem C storage with elevation below the forest line (dominated by aboveground C), but increasing ecosystem C stocks above the tree line (dominated by organic carbon in the O-layer), so that the total C stocks showed a minimum at the transition between the forest line and tree line (Speed et al., 2015). This suggests that woody encroachment (raising the tree line) because of the cessation of grazing does not change total ecosystem C stocks, but that C storage is moved from soil to vegetation. Results from Sørensen et al. (2018) near Hjerkinn in Dovrefjell assessing differences in C pools and fluxes between an *Empetrum*-dominated heath, herb- and cryptogam dominated meadow and a *Salix*-shrub community revealed greater ecosystem C stocks in the meadow and heath communities compared to the shrub community, primarily due to lower SOC stocks under shrub (~76, 107 and 50 ton ha^{-1} under heath, meadow, and shrub, respectively). Although aboveground biomass was highest under shrub vegetation, it was not sufficient to compensate for less SOC and the authors suggest that shrub expansion into alpine meadow and heath will reduce SOC content of alpine soils due to high rates of decomposition (Sørensen et al., 2018). Loss of SOC due to the expansion of shrubs into tundra heath associated with increased turnover of SOC was also reported by Parker et al. (2015) near Abisko in Sweden. By contrast, a study from Trøndelag (115 m a.s.l.) showed no significant differences in SOC stocks in the upper 30 cm between an actively grazed pasture (~85 ton ha^{-1}) and an adjacent afforested pasture (~71 ton ha^{-1}) 50 years after afforestation with Norway spruce (Strand et al., 2021). Together, these studies illustrate the great variation in expected impacts on SOC due to grazing cessation leading to woody encroachment of grasslands and heathlands.

Studies indicate that both quantity and chemical stability of SOC change during woody encroachment, because of reduction or cessation of grazing. Impacts of grazing on SOC depend on grazing intensity, where overgrazing causes loss of SOC but light or moderate grazing may maintain or increase SOC stocks (Garnett et al., 2017; Martinsen et al., 2011). However, it is difficult to assess whether a change in grazing management will produce positive or negative changes in the system's total C storage due to a great spatial variability in productivity, climate and SOC stocks that determine the net effect of grazing on SOC (Garnett et al., 2017). In addition, a potential increase in C sequestration in the soil will have to be weighed against the climate effect of woody encroachment on C sequestration in vegetation and change in albedo (de Wit et al., 2014). Caution must be exercised in scaling up results from studies at a small-scale level to national levels by only using information based on areal distribution of different land covers. As described by Speed et al. (2014), main differences in SOC stocks between sites with long-term exclusion of grazers compared to sites with high grazing intensities in Norway were mainly attributed to different thicknesses of O-horizons as well as different soil types and not grazing per se.

Intermediate grazing densities lead to high biodiversity in semi-natural pastures and continued grazing is essential to preserve this biodiversity (Austrheim et al., 2016). Despite evidence for positive effects of grazing on biodiversity and ecosystem services, studies on synergies and trade-offs between aboveground and belowground processes, including the role of N availability controlling ecosystem C storage are lacking from Norwegian outfields. Due to the great variation in estimated C stocks (both above- and belowground) and predicted effects of grazing on ecosystem C storage, more research on processes altering C dynamics is needed. Soil models as used today may help in estimating SOC pools, but unless properly parameterized and without accurate input data, they may fail in predicting changes in SOC and corresponding C emissions (Bolinder et al., 2006). The impacts of grazing on both C inputs into the soil and the stability of SOC are poorly captured in current C models, especially in colder climates at higher latitudes. Given that these regions are likely to see relatively higher future warming, it is essential to understand the mechanisms that underpin grazing impacts on SOC to project future climate scenarios. Current knowledge is too limited to give specific recommendations, and there is a need for more controlled trials with a selection of management practices such as different grazing densities or rotational grazing across climatic gradients (EIP-AGRI-Focus-Group, 2018). Long-term studies with a mechanistic focus on biogeochemical processes controlling C dynamics are needed. More knowledge about region- and site-specific impacts of grazing will give a better management of grazing animals in the Norwegian outfields.

9 | ALTERED TILLAGE PRACTICES

It is often considered that the avoidance of plough tillage mitigates SOC loss. Optimistically, Smith et al. (1998) estimated that 100% adoption of no-tillage (NT) would equal all carbon emissions from fossil fuel in European agriculture. However, the potential of reduced tillage for carbon storage has more recently been questioned. Baker et al. (2007) concluded that though there are good arguments for reduced tillage, evidence for increased carbon storage is not compelling. Luo et al. (2010) found no significant difference in carbon storage between NT and conventional plough tillage (PT) in a global analysis of 69 trials. In a review seeking to 'identify the true and the false' concerning SOC sequestration, Powlson, Whitmore, and Goulding (2011) concluded that SOC-increases from reduced tillage are much smaller than claimed, at least in temperate regions.

Climate factors may be important in relation to the effect of tillage on SOC storage. Contrasting effects of NT on carbon storage were found by Janzen et al. (1998) under prairie conditions in western Canada to those found by Gregorich et al. (2005) on wetter soils in eastern Canada. These studies concluded that NT had greater potential for carbon storage in the west, on account of differences in climate, tillage depth and the amount and quality of crop residues. In eastern Canada, Angers et al. (1997) found no significant differences in the total mass of SOC between NT, shallow tillage (ST) and PT, despite differences in its depth distribution. Under cool and moist climatic conditions in Switzerland, Hermle et al. (2008) found greater SOC mass at 0–20 cm after NT and ST than after PT, whilst the reverse was true at 20–40 cm, but there was no overall difference in SOC storage.

Changes in soil bulk density (BD) must be considered when calculating SOC. In a study on loam soil in southeast Norway, Riley and Ekeberg (1998) found effects on both SOC% and BD after various tillage depths, but the same total SOC mass. Similarly, Riley et al. (2005) found little difference in SOC mass between ST and PT on clay loam and sandy loam soils in central Norway. Several field trials with different tillage regimes (PT, ST and in some cases NT) have been conducted in southeast Norway for periods of up to 40 years. Effects of ST versus PT on SOC% and BD are presented here after 20–40 years, as well as total SOC mass expressed on equivalent mass basis.

Mean data are presented from four trials on loam soil and two trials on clay loam soil. Normal annual temperature and precipitation are 3.6°C and 585 mm at the loam site and 6.1°C and 850 mm at the clay loam site. PT to ~25 cm was compared with ST to ~10 cm in all trials. Details of trials on loam are given in Riley (2014) and of those on clay loam in Riley et al. (2009). Seventeen soil profile pairs were sampled at three 10 cm depth intervals on loam after ~28 trial years, and twelve pairs were sampled on clay loam after 20 and 40 trial years.

The mass of SOC in each 10 cm soil layer to 30 cm was calculated by multiplying SOC% with BD. The topsoil depth lay in all trials at 25–30 cm. No difference was found in subsoil SOC% between ploughed and unploughed plots. Constant values of SOC% and BD below 30 cm were thus used to calculate SOC in equivalent soil mass, as by Hermle et al. (2008).

Mean SOC% in the upper topsoil horizon increased in the absence of ploughing but differed little in the middle

Depth	Ploughed	Unploughed	Difference	SE of diff.	*p*-level[a]
SOC concentration (%)					
0–10 cm	2.40	2.65	+0.25	0.09	0.010**
10–20 cm	2.38	2.24	−0.14	0.10	0.146 ns
20–30 cm	1.96	1.51	−0.44	0.15	0.007**
Bulk density (kg/litre)					
0–10 cm	1.31	1.25	−0.06	0.02	0.015**
10–20 cm	1.35	1.41	+0.06	0.02	0.009**
20–30 cm	1.46	1.51	+0.06	0.03	0.067+
SOC mass (kg/m^2)					
0–10 cm	3.13	3.31	+0.18	0.10	0.083+
10–20 cm	3.18	3.13	−0.05	0.11	0.646 ns
20–30 cm	2.80	2.20	−0.60	0.19	0.004**
Sum 0–30 cm	9.11	8.64	−0.47	0.32	0.156 ns
SOC mass (equivalent mass basis)					
0–30 cm	9.71	9.05	−0.66	0.41	0.116 ns
0–40 cm	10.21	9.70	−0.51	0.35	0.115 ns

[a]Probability of two-tailed paired *t*-test (** = significant, + = 'trend', ns = non-significant).

TABLE 2 Effects of many years without ploughing versus annual ploughing on soil organic carbon (SOC) and soil bulk density (BD) measured in 29 soil profile pairs from six long-term tillage trials conducted on loam and clay soils in south-east Norway.

horizon and decreased in the lowest horizon (Table 2). The opposite was found for mean BD, which showed high negative correlation with SOC% ($r^2 = 0.92$). Without ploughing the SOC mass was slightly higher in the upper topsoil than with ploughing, whilst in the lower topsoil it was significantly less than with ploughing. There was no significant difference between ST and PT in SOC mass summed to 30 cm, nor in the total amounts calculated on equivalent mass basis. Our results apparently contrast with those of Singh and Lal (2005), who reported a small but significant increase of SOC stocks for one reduced-tillage experiment in Norway. However, this latter experiment considered only SOC stock changes in the 0–20 soil layer, while our present study shows that such positive effects are not found when considering the 0–30 cm layer.

The results of these trials suggest that reduced tillage on its own appears to have very limited potential for increasing carbon sequestration in soil, and this measure does not fulfil expectations for a management practice that increases soil carbon pools at depth. This conclusion accords with that of Kätterer et al. (2012), with respect to the role of reduced tillage for carbon storage under Nordic conditions. Reduced tillage in combination with the use of cover crops may have some potential, as this may increase the amount of carbon returned to the soil. Reduced tillage also provides benefits due to greater soil stability, which reduces erosion risk, improved soil structure in surface layers and lower fuel energy consumption and machinery/labour costs. Disadvantages include the necessity for greater use of herbicides for weed control, increased risk of grain infection by mycotoxins under moist growing conditions and the possibility that reduced tillage may lead to greater N_2O emissions when practiced on poorly drained soils.

10 | INVERSION OF CULTIVATED ORGANIC SOILS

Cultivated peatland in Norway is estimated to be about 60,000 ha (Norwegian Environmental Agency et al., 2023), mostly used for long-term grassland, with some limited area for cereal and vegetable production. Drainage and cultivation of organic soils generate high GHG emissions, in particular CO_2 and N_2O (IPCC, 2014), and therefore many countries are currently trying to curtail their cultivation. The area with drained organic cropland soils in Norway constitutes an annual emission of more than 2 Mt CO_2-eq. (Norwegian Environmental Agency et al., 2023). In Norway, organic soils represent only about 7% of arable lands (Lågbu et al., 2018), but their cultivation appears crucial to the local economy of some districts, which highlights the need for mitigation methods compatible with continued agricultural use.

In addition to the negative climate impact, drainage of organic soils is technically challenging because humified peat has poor infiltration properties (Myhr, 1984), and poorly drained organic soils are vulnerable to compaction, poor plant growth and low yields (Haraldsen et al., 1995;

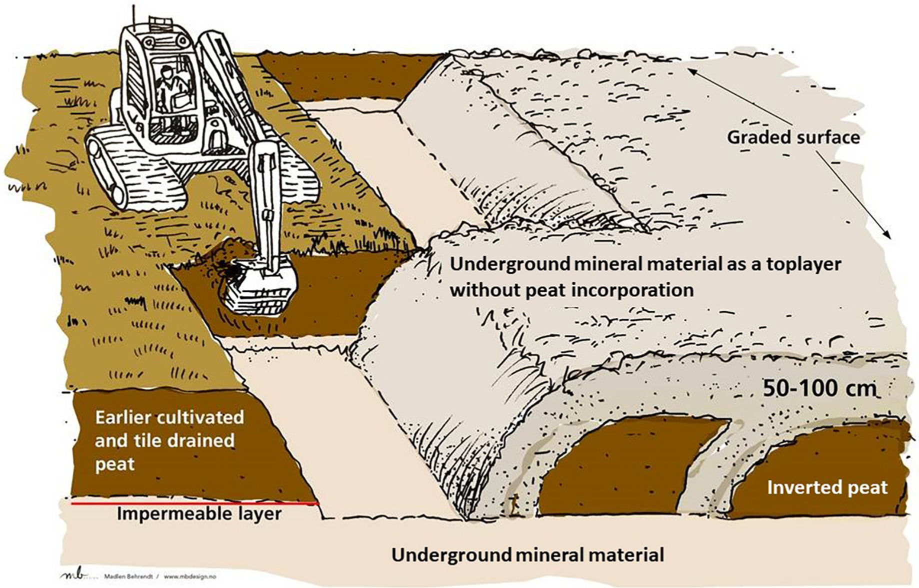

FIGURE 1 Illustration of the peat inversion method. Mineral soil from below the peat layer is placed as a top layer, burying the peat. Vertical layers of mineral material increase water infiltration. Modified illustration by J. A. Heir, appearing in Aamodt (1990).

Myhr & Njøs, 1983; Øpstad, 1991). In addition, the rapid mineralization of the previously protected organic matter results in peat subsidence over time, which can require redraining at greater depth after a number of years. For these reasons, alternate solutions to draining the peat surface have been explored even before climate effects were considered an issue.

Inverting peatland was first proposed in Norway as a method for improving the drainage properties of the cultivated peatland and reducing the effects of subsidence. The practice was combined with profiling to improve surface runoff and allow more rapid drainage early in the season following snow melt. The method of inverting cultivated peat is to excavate the underlying mineral subsoil and place this as a topsoil covering the organic material (50–100 cm thick layer). The mineral soil is placed in inclined layers where the water can drain to the subsoil (Figure 1). The layer of mineral mass brought to the surface should be thicker than the plough layer. Using this method, the buried peat is expected to maintain a high moisture content, which protects the organic material from oxic decomposition. The use of oxygen sensors in a field experiment in 2017 and 2018 in Western Norway, confirmed that buried peat material experienced mainly anaerobic conditions following inversion (Dörsch et al., 2017). Although peat inversion has been applied in Norway since the 1980's, relatively little is known about the impact on the decomposition of the peat material. One study in Switzerland applied ~40 cm of mineral soil on top of a drained organic soil and showed a reduction in CO_2 emissions, suggesting that burying organic material reduces mineralization rate (Wang et al., 2021).

Inversion has also been suggested as a method for reducing non-CO_2 GHG emissions from cultivated peat. N_2O is a significant GHG produced in fertilized organic soils, largely because of the relatively high and fluctuating moisture content of these soils, the low pH and the presence of high quantities of organic matter (Kasimir-Klemedtsson et al., 1997). Peat inversion can mitigate these factors by improving the drainage of surface soil, altering the pH and acid buffering capacity, and reducing availability of organic carbon within the rooting zone where much of the nitrogen cycling occurs. CH_4 is often not considered an issue for cultivated peat soils due to its high potential for oxidation in the drained topsoil. However, in poorly drained soils and in open field drains, cultivated peatland can still be a net source for CH_4 (Tiemeyer et al., 2016). Burying the peat offers the potential to improve drainage and thereby increase the volume of oxygenated soil that can harbour methane consuming bacteria. Emissions of CH_4 and N_2O have been compared between a traditionally pipe-drained cultivated peatland and an inverted peat over a 2-year period in Norway (Dörsch et al., 2017; Hansen et al., 2016; Rivedal et al., 2017). Emissions of CH_4 were 200 and 140 kg CH_4–C ha^{-1} from the traditionally-drained site in 2015 and 2016 respectively, while the inverted site emitted 1.0 kg CH_4–C ha^{-1} in 2015 and had an uptake of 1.0 kg CH_4–C ha^{-1} in 2016. Similarly, N_2O emissions were reduced from 4.25 and 9.50 kg N_2O–N ha^{-1} in the traditionally drained site, to approx. 3.60 kg N_2O–N ha^{-1} in the inverted site in 2015 and 2016 respectively (Rivedal et al., 2021). At an experimental site in Switzerland, authors measured 20.5 ± 2.7 kg N ha^{-1} $year^{-1}$ N_2O–N in a traditionally drained cultivated peatland, compared with 2.3 ± 0.4 kg N ha^{-1} $year^{-1}$ in a similar site buried under ~40 cm of mineral soil (Wang et al., 2022). Therefore, inversion appears to drastically reduce fertilization-induced N_2O emissions from cultivated peat soils.

TABLE 3 Loss on ignition (%) in soil samples taken in 1993 and 2017 from the mineral layer of sand (89%) with gravel covering inverted peat, and increase of carbon content per ha and year during the period.

	1993	2017	Increase in C-content during the period	Yearly increase
0–5 cm	5.30	6.60	Approx. 5.0 t	Approx. 200 kg
5–20 cm	3.30	3.51	Approx. 2.3 t	Approx. 80 kg

Note: Ley was established at the site in Vikeid in Sortland in 1985 as long-term grassland, and was each year only fertilized with mineral fertilizer (NPK).

TABLE 4 Statistic about new draining on earlier drained cultivated area in Norway in the period 2013–2022.

Year	Systematic pipe drained (ha)	Inversion (ha)*	Graded (ha)
2013	4603	130	173
2014	4857	159	296
2015	2756	125	117
2016	2352	70	151
2017	5168	66	265
2018	3836	100	228
2019	3077	89	448
2020	3316	105	201
2021	2401	74	184
2022	2672	42	122
Total for 10 years	35,038	960	2225

Source: Norwegian Agriculture Agency (direct correspondence).
*All inversion is done on peat soil. Largest drained area by inversion is in: Møre og Romsdal, Trøndelag Innlandet, Norland.

Alongside protecting native organic matter and reducing the GHG emission potential of cultivation on peatland, there is a potential to actively store organic carbon in the mineral subsoil that is brought to the surface in the process of peat inversion. Deep mineral soils may lack organic carbon and are therefore a potential "blank slate" for the building of organic material. In a long-term monitoring of grassland cultivation over inverted peat conducted in Norway between 1993 and 2017, a pilot study carried out by NIBIO showed an increase in soil carbon of approximately 0.25–0.30 Mg C ha^{-1} $year^{-1}$ in the top 20 cm (Table 3).

Despite the potential of peat inversion for storing organic carbon and reducing GHG emissions from peatland cultivation, the method is poorly studied. The high degree of field disturbance together with the cost of the excavation works may be prohibitive to the wide scale adoption. The practice of applying peat inversion is utilized in Norway, but inversion has only been used on a small proportion of the total area of cultivated peatlands (Table 4). Internationally, peat inversion appears little used and is mostly limited to some pilot studies (Wang et al., 2021).

More research is required to understand how vulnerable the buried peat is to decomposition processes in an actively managed inverted peat. In terms of potential implementation, we need to identify and map regions where these methods can be applied, starting by focusing

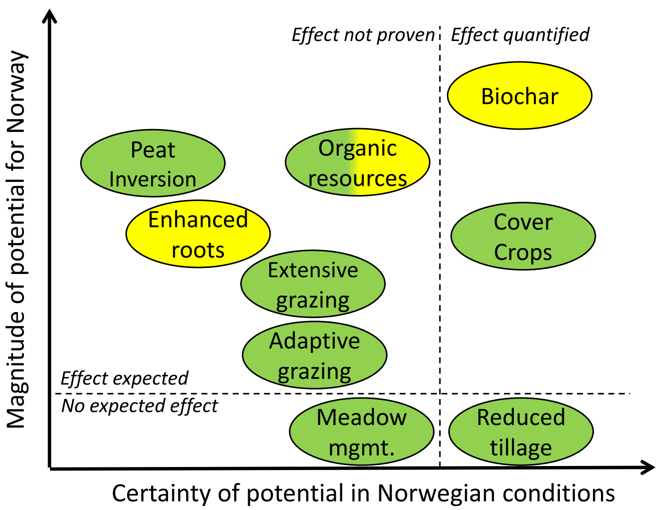

FIGURE 2 Summary diagram presenting a qualitative evaluation of the magnitude (y-axis) and certainty (x-axis) of the nine most promising methods to increase SOC storage in Norway. The availability of the technology and the likelihood of its implementation are presented by the implementation readiness level (green for high and yellow for middle).

on cultivated peatland areas over loose sediment. It will also be important to identify mineral subsoils that are suitable for cultivation following inversion in different climatic conditions. Additionally, it will be important to

TABLE 5 Summary of the advantages and limitations of the nine management methods.

Management method	Advantages	Limitations
Utilization of organic resources	Technology and farm equipment are readily available in most cases. Easily implementable by most farmers.	Available resources are already largely exploited locally. Improved use might require extensive shipping of resources across the country. Net mitigation at the national level is difficult to demonstrate as application at one location may compromise quantities available for other locations. Resources (animal manures, straw, etc.) are limited and compete with other uses, such as bioenergy.
Use of biochar	Has the largest potential of all methods for sequestering carbon: 0.9 Mt CO_2 per year. Permanent storage (>100 years). Synergy with reduction in N_2O emissions.	High production costs, currently exceeding the value of C credits. Competes with other uses of clean feedstock (straw, forest residues, etc.), such as bioenergy.
Crop diversification and the use of cover crops	Well-accepted technology easily implementable by most farmers. Sustainable C sequestration potential of 0.2 Mt CO_2 per year.	More knowledge is needed about adapted agronomic practices to Norwegian conditions with short growing season. Interactions with N_2O emissions uncertain.
Use of plants with larger and deeper root systems	A potentially promising method. Farmers can easily cultivate new varieties without having to change their management practices.	There is not enough data and knowledge about the effects of cultivating plants with deeper roots, both in general and in Norway. Verification of C storage in deeper soil is challenging.
Improved management of meadows	Technology is mature and acceptance by farmers is high. Continuing good management practices already in place will prevent loss of carbon.	Effect has not been demonstrated in Norway. There is little additional carbon to store in meadows as current meadows already contain high carbon levels.
Adaptive grazing of productive grasslands	A potentially promising method. Able to feed more animals on less land.	Requires closer follow-up from the farmer and more frequent herding of animals. Effect has not been demonstrated in Norway.
Managing grazing pressure in extensive grasslands	Large areas of land are available for extensive grazing in Norway.	Knowledge is too limiting to estimate carbon sequestration potential and to make recommendations.
Altered tillage practices	Technology is available and the method is generally accepted by farmers. Reduction in tillage results in fuel savings.	Norwegian studies show no increase in soil C after 20–40 years. Relies largely on use of broad-spectrum herbicides. Increased risk of fungal diseases and associated mycotoxins.
Inversion of cultivated organic soils	Technology is available. Large reduction of soil C losses and GHG emissions per unit surface area.	Requires costly investments, which might be prohibitive. Few reliable data currently available. Applies to a limited area: not all cultivated peatlands are suitable for inversion, and potential areas have not been fully mapped.

establish best practice methodologies to reduce the costs of applying the method and maximize the benefits for the agronomic system and the associated climate benefits.

11 | CONCLUSION

In the present study, we have looked at measures available to farmers in Norway to maintain and potentially increase carbon storage in soil in order to influence the Earth's climate for the next 100 years and beyond. For most of the measures identified here, based on a Norwegian context, the knowledge gaps on the potential effect are large. It is not straightforward whether studies carried out in other parts of the world translate to similar effects in Norway. Furthermore, it is unknown how much, and specifically for how long the increase in C stocks will last, as it is dependent on soil conditions and climate.

A summary diagram presenting the estimated magnitude and certainty of the nine methods to increase SOC storage is presented in Figure 2. The use of cover crops performs well in terms of certainty in the reported potential, availability of the technology, and likelihood for implementation by farmers. Based on data from Sweden, the implementation of cover crops on 60% of grain fields can achieve a carbon sequestration of approximately 0.2 Mt CO_2-eq, corresponding to 4.4% of Norwegian agricultural emissions and 0.4% of total national emissions (Miljødirektoratet, 2024), provided that it is adapted to Norwegian conditions, further developed and disseminated correctly. This potential is twice as much as previously estimated. Biochar has the greatest potential and has generated great interest in the industry. Nevertheless, biochar requires further development before it can be implemented on a large scale. Today, large amounts of organic resources are added to Norwegian soil in a sensible way, with a positive effect on carbon storage. This needs to be continued in the future so that current carbon stocks are not depleted. The continuation of good practices is also important for the management of meadows and grazing pressure in extensive grasslands so that losses of already high levels of SOC are not induced. Cultivation of plants with deep roots has a potential for increasing carbon storage and is easy for farmers to implement, but it is still very uncertain how the effect can be achieved and quantified. For these two methods, research is mostly recommended for better documentation. The same applies to inversion of cultivated peatlands, which could be used in suitable areas in several parts of the country, but the carbon storage effect needs to be documented and adoption will depend on it being economically justifiable. A summary of the main advantages and disadvantages of the nine measures evaluated is presented in Table 5.

This work emphasizes that it is important to take local conditions into account. Some methods that are internationally accepted have a more limited application in Norway, such as reduced tillage. Improved management of meadows and adaptive grazing of productive grasslands should be further investigated for both effects on carbon in the soil and applicability in Norway. The synergies of combining methods, such as the use of cover crops together with reduced tillage and/or application of biochar have potential but are not presented here because of limited documentation thus far.

AUTHOR CONTRIBUTIONS

Alice E. Budai: Writing – review and editing; writing – original draft; methodology; conceptualization. **Daniel P. Rasse:** Conceptualization; funding acquisition; writing – original draft; writing – review and editing; methodology; project administration. **Thomas Cottis:** Writing – review and editing. **Erik J. Joner:** Writing – original draft; writing – review and editing. **Vegard Martinsen:** Writing – original draft; writing – review and editing. **Adam O'Toole:** Writing – original draft. **Hugh Riley:** Writing – original draft; writing – review and editing. **Synnøve Rivedal:** Writing – review and editing. **Ievina Sturite:** Writing – original draft; writing – review and editing. **Gunnhild Søgaard:** Writing – review and editing. **Simon Weldon:** Writing – review and editing; writing – original draft; conceptualization. **Samson Øpstad:** Writing – review and editing; writing – original draft.

ACKNOWLEDGEMENTS

This research was mainly developed in the framework of the European Joint Program for SOIL "Towards climate-smart sustainable management of agricultural soils" (EJP SOIL) funded by the European Unions' Horizon 2020 research and innovation programme (Grant Agreement No 862695 EJP SOIL). Partial support was also provided by the NRC-funded projects SUSCOW (# 320699) and PLATON—a PLATform for Open and Nationally accessible climate policy knowledge project (# 295789).

DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available from the corresponding author upon reasonable request.

ORCID

Alice E. Budai https://orcid.org/0000-0002-6675-4548
Daniel P. Rasse https://orcid.org/0000-0002-5977-3863
Thomas Cottis https://orcid.org/0009-0004-1824-3390
Erik J. Joner https://orcid.org/0000-0002-5033-7811
Vegard Martinsen https://orcid.org/0000-0002-7096-1806
Adam O'Toole https://orcid.org/0000-0003-4791-6975
Ievina Sturite https://orcid.org/0000-0001-5662-7695
Gunnhild Søgaard https://orcid.org/0000-0003-4798-3932
Simon Weldon https://orcid.org/0000-0001-9851-9182

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How to cite this article: Budai, A. E., Rasse, D. P., Cottis, T., Joner, E. J., Martinsen, V., O'Toole, A., Riley, H., Rivedal, S., Sturite, I., Søgaard, G., Weldon, S., & Øpstad, S. (2024). Qualitative evaluation of nine agricultural methods for increasing soil carbon storage in Norway. *European Journal of Soil Science*, *75*(3), e13493. https://doi.org/10.1111/ejss.13493