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Long-term soil organic carbon changes after cropland conversion to grazed grassland in Southern Sweden

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Abstract

There is growing awareness of the potential value of agricultural land for climate change mitigation. In Sweden, cropland areas have decreased by approximately 30% over recent decades, creating opportunities for these former croplands to be managed for climate change mitigation by increasing soil organic carbon (SOC) stocks. One potential land-use change is conversion of cropland to grazed grasslands, but the long-term effect of such change in management is not well understood and likely varies with soil type and site-specific conditions. Through sampling of mineral and peatland soils within a 75-year chronosequence of land converted from crop production to grazed grassland, we assessed how time since conversion, catenary position, and soil depth affected SOC storage. The SOC stocks calculated at an equivalent soil or ash mass increased through time since conversion in mineral soils at all topographic positions, at a rate of ~0.65% $year^{-1}$. Soils at low topographic positions gained the most carbon. Peat SOC stock gains after conversion were large, but only marginally significant and only when calculated at an equivalent ash mass. We conclude that the conversion of mineral soil to grazed grassland promotes SOC accumulation at our sites, but climate change mitigation potential would need to be evaluated through a full greenhouse gas balance.

KEYWORDS

catenary position, chronosequence, climate change mitigation, grazed grassland, soil organic carbon, topographic wetness index

1 | INTRODUCTION

The sequestration of organic carbon (C) in soils provides a mitigation opportunity against climate change (Intergovernmental Panel on Climate Change [IPCC], 2019; Paustian et al., 2016). When photosynthetic C input to soils is higher than the C losses, soils will act as a C sink and contribute to the net removal of CO_2 from the atmosphere (Hewins et al., 2018; Minasny et al., 2017). In managed systems, the input of C is mostly because of plant turnover and added soil amendments, while the output is mainly because of respiration during the

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decomposition of organic matter, and to some extent methane emissions and losses in dissolved forms or erosion (Paustian et al., 2004). Therefore, soils can be managed to strengthen their C sink by increasing inputs and/or reducing C losses, especially in agricultural lands that have historically lost organic C (Bruni et al., 2022; Ramesh et al., 2019). Management changes could focus on decreasing C losses by reducing soil disturbance, such as ploughing and tillage, because this intensifies the decomposition of soil organic matter (SOM) (Harden et al., 2018; Smith et al., 2008). Maintaining continuous vegetation cover could also contribute, by increasing C inputs to soil. Therefore, conversion of croplands to grasslands is expected to improve SOC stocks because it both supports continuous vegetation cover and reduces soil disturbance (Minasny et al., 2017).

Converting cropland to grazed grassland has other ecological benefits. In fact, grasslands with low-intensity livestock management are recognized as biodiversity hotspots since grazing prevents secondary succession that would turn grasslands into forest and shrubland (Boch et al., 2018; Socher et al., 2012; Török et al., 2016). Especially semi-natural pastures, that is, pastures with a long history of traditional low-input grazing management, are recognized globally for their high species richness at a small spatial scale (Wilson et al., 2012). These grasslands have declined dramatically worldwide, especially in northern Europe because of local practices to increase productivity, such as drainage, fertilization, frequent harvesting, high livestock density, and decrease in landscape heterogeneity that contributes to biodiversity loss (Allan et al., 2014; Dengler et al., 2020; Lindborg & Eriksson, 2004).

In Sweden, cropland area decreased by around 30% between 1951 and 2015 (Statistiska centralbyrån [SCB], 2019), and the increased effectiveness of agricultural practices allowed croplands to be partly converted to grazed grasslands, foremost in areas with small-scale farming (Bergh et al., 2020; Slätmo et al., 2012). This conversion may help to mitigate both climate change by accumulating SOC after conversion and biodiversity, although grazers could partly offset SOC increases by releasing greenhouse gases (Rogiers et al., 2008). An added complexity when evaluating the potential benefits of conversion is that much of the low-lying cropland area in Sweden is located on artificially drained wetlands or peatlands (Naturvårdsverket, 2022). Peatlands in alpine and boreal regions might have lost historically between ~20 and ~100 kg C m^{-2} because of drainage and agricultural management (Grønlund et al., 2008; Rogiers et al., 2008). Earlier studies have shown that reducing drainage to restore the original hydrologic conditions is a much more effective climate change mitigation approach for peatlands than for mineral soils (Leifeld & Menichetti, 2018), suggesting that management strategies must account for soil type.

The variation in soil types is related to the topography, which controls the gravity-driven water movement, and to some extent transport of soil organic matter from higher to lower positions in the landscape through erosion and runoff (Hu et al., 2021). Soils in lower catenary positions are likely to be wetter, which promotes higher biomass production and thus higher litterfall (Bot & Benites, 2005). Soils in lower positions can also remain anoxic for parts of the year, further slowing down the decomposition of SOM, resulting in increases in SOC (Moyano et al., 2013) and peat soil formation—a common feature of the agricultural landscape in Sweden. Soil texture also varies with catenary position, with finer textures dominating at lower elevation. Slope processes are also important factors because they cause finer materials to slowly move downhill over time and cause accumulation of SOC associated with fine particles.

While the conversion of croplands to grasslands generally increases SOC stocks, there is no consensus on when SOC stocks in converted soils reach an equilibrium or how much C these soils can store (Paustian et al., 2016; Preger et al., 2010; Stockmann et al., 2013; Yu et al., 2019). This uncertainty raises questions on cropland to grassland conversion as a climate change mitigation strategy because the climate and biodiversity benefits trade-off with reduced land area dedicated to crop production. The potential to manage soil for climate change mitigation is also bound to a local context, where the SOC dynamics under different land uses depend on complex interactions between climate, vegetation, soil biota, soil fauna, soil properties, and topography (Wiesmeier et al., 2019). This further complicates the estimation of the mitigation potential of converted soils, and points to the need to incorporate well-founded local knowledge when designing mitigation measures (Skalský et al., 2020). This motivates studies investigating how SOC stocks vary in soils converted from croplands to grasslands with low-intensity grazing (hereafter grazed grassland, see Bengtsson et al. (2019) for definition) on a local scale and through time.

In Northern Småland, we selected a chronosequence with conversions of cropland to grazed grassland taking place in the 1940s, 1960s, and 1990s, together with croplands that have been continuously cultivated during the 20th century until the present day. The chronosequence is located in a relatively small-scale farm with homogenous climatic conditions and land management practices, whereas the local bedrock morphology has caused heterogeneity in soil types ranging from peat to mineral soils in different catenary positions (Sveriges Geologiska Undersökning [SGU], 2022). The differences in soil water content and catenary position of the converted soils along

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this chronosequence allow the investigation of combined local hydrologic and land management effects on SOC stocks in the area.

Here we used the topographic wetness index (TWI) to characterize the catenary position (a proxy for long-term soil hydrologic conditions) and time since conversion (TSC) to quantify SOC stock changes in soils converted from cropland to grazed grassland in the last ~75 years. We hypothesized that: (1) As a result of lower soil disturbance, SOC stocks increase through time in croplands that have been converted to grazed grasslands (Kätterer et al., 2008; Kim & Kirschbaum, 2015; Poeplau & Don, 2013). (2) Increases in SOC stocks after conversion are more pronounced in topsoil because of the change in edaphic factors and plant species composition (Don et al., 2009; Wiesmeier et al., 2019), whereas subsoil SOC stocks should vary less than in topsoil (Poeplau & Don, 2013), or decrease (Yang et al., 2022). (3) SOC stocks are higher at lower catenary positions (Li, McCarty, et al., 2018) and have potential to gain more C than upland soils after conversion thanks to wetter conditions promoting C retention. (4) Organic soils (peatlands drained prior to this 75-year chronosequence) with very high initial SOC stocks have limited capacity to accrue SOC under aerobic conditions because of intense oxidation regardless of conversion (Leifeld & Menichetti, 2018).

2 | METHODS

2.1 | Site description

The chronosequence study was conducted on a farm in Northern Småland that included 100 hectares of cropland and 50 hectares of grazed grassland (Figure 1). The grasslands are grazed without fertilization, sowing, or hay making. The livestock graze the grasslands between May and October every year, a procedure which has been continuously performed throughout the studied chronosequence and until the present day. On average, 60 cattle are released onto the 50 hectares of grazed grassland every year, which corresponds to approximately one cattle per hectare. The croplands are managed through conventional farming with pesticides and fertilizers in a 3- to 4-year crop rotation plan of lay and cereal. The fertilizers consist of YaraBela AXAN and YaraMila 21-3-10 (YARA, 2022a, 2022b), and farmyard manure containing a mixture of dung, left-over material from fodder, straw, and urine. The farmyard manure put on the cereal croplands is mixed with agricultural lime. The average rates of fertilizer applied to the cereal and lay fields are approximately 100 and 250 kg N ha^{-1}, respectively; both fields also receive 4 kg S ha^{-1}, 20 kg P ha^{-1}, and 100 kg K ha^{-1}. The plough depth in the croplands has been ~20 cm throughout the last century. Information about land management has been provided by the landowner, whose family has operated the farm during the whole chronosequence.

The geology of the studied area is characterized by a bedrock of granite, frequently exposed in rock outcrops (SGU, 2022). The bedrock is covered by coarse till deposited during late Quaternary glaciations (SGU, 2022). In elevated parts of the landscape, the till forms the near-surface parent material for (sub-)recent soil development, often covered by coniferous forest. Low-lying parts of the landscape have a coverage of post-glacial clay, silt, and sand with some development of peat, and have been drained in the past, likely before the beginning of our 75-year chronosequence. The land use in the post-glacial fine sediments is characterized by agriculture and grazing. The hydrology of the area is foremost characterized by the river Stångån, which flows through the agricultural lands, accumulates water from the artificial drainage of the land, and empties into Lake Ärlången (Figure 1). The mean annual precipitation of the area is 540 mm year^{-1} and the mean annual temperature is 6.7°C (Sveriges Meteorologiska och Hydrologiska Institut [SMHI], 2021).

2.2 | Sampling methods

Field sampling was conducted at 23 sites between June and July 2021, with a minimum of five replicates in cropland and grazed grassland for every category in the chronosequence: conversion in the 1940s (5 sites), 1960s (6 sites), 1990s (5 sites), and continuously cultivated croplands (7 sites). Sites for every category were randomly chosen with information about conversion history obtained from the landowner and maps (Lanmäteriet, 2022; Rikets Allmänna Kartverk, 1947). The sampling design was adapted from that presented by Livsey et al. (2020). Within each site, five soil cores were collected with a soil corer (Eijkelkamp), and each soil core was divided into four discrete sample depths: 0–10, 10–20, 20–40, and 40–60 cm. The five cores from each sample depth were combined to form a single sample for analysis representing each of the 23 sites. The five replicates within each site were taken at one centre point and four points 10 m around the centre with an even distribution. One bulk density (BD) sample was extracted for each site and depth at the centre point with a cylinder of 100 cm^3 being driven horizontally into an open soil profile. All samples were then analysed for SOM content, BD, and soil texture, and in selected profiles as well as total C and total nitrogen (N). The 0–10 cm samples were also used for analysing pH and plant-available nutrients. Visible roots were removed, and the samples were then stored in sealed plastic bags.

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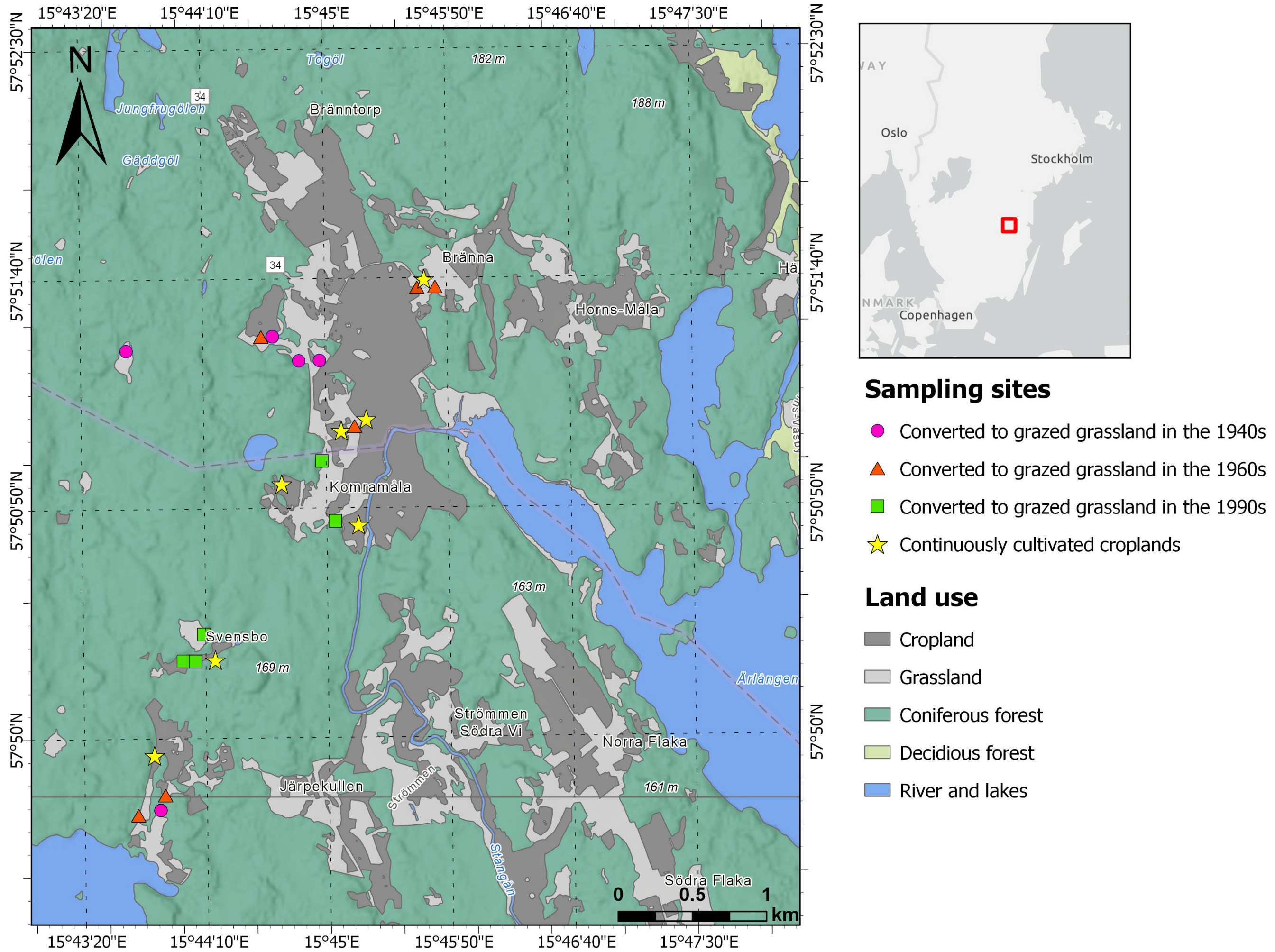

FIGURE 1 Soil sampling sites and location of the studied area together with land use types and topography in Northern Småland, Sweden. Source of land use data: GSD-Fastighetskartan vector, scale 1:5000–1:20,000 ©Lantmäteriet (2019). Source of basemap: Esri, Airbus, DS, USGS, NGA, NASA, CGIAR, N Robinson, NCEAS, NLS, OS, NMA, Geodatastyrelsen, Rijkswaterstaat, GSA, Geoland, TEMA, Intermap and the GIS user community (2022).

2.3 | Soil analyses

Prior to any analysis, samples were air-dried, homogenized, and passed through a 2 mm sieve. Soil texture, SOM content, and BD were analysed at the Department of Physical Geography, Stockholm University, Sweden, whereas detailed chemical analyses (total C, total N, pH, and plant-available P, K, Mg, Ca, Al, and Fe) for selected samples were performed in an external laboratory (Agrilab AB, Uppsala, Sweden; http://www.agrilab.se/gem/).

Soil texture was determined for soils containing less than 30% SOM content through the hydrometer analysis (Ashworth et al., 2001) (Figure S1), and soils with more than 30% SOM were classified as peat soils. For the analyses of SOM content and BD, the samples were additionally oven-dried at 105°C for 24 h. SOM content (SOM_C, in %) was obtained using the loss on ignition method following Heiri et al. (2001). BD was obtained according to the method by Don et al. (2007) and Rowell (1994), where the soil is categorized into a fine fraction (particles <2 mm), which is referred to as 'soil', and a coarse fraction (>2 mm) encompassing gravel and pebbles. Soil BD ($g\,cm^{-3}$) was then calculated by removing the mass and volume contributions of the coarse fraction (assumed to have a density of 2.65 $g\,cm^{-3}$).

Total C and N were determined through elemental analysis with dry combustion (LECO CN928). A Jenway with glass electrodes determined pH and an ICP Spectro Blue determined plant-available nutrient contents using the Ammonium-Lactate-method (AL-method). Because the pH was lower than 7 in most topsoil samples (Table 1) and the granitic bedrock in the area is characterized by very little inorganic C (SGU, 2022), we assumed that the SOC content was equal to the total C.

To estimate the SOC content (SOC_C, in %) in samples for which chemical analysis was not performed, a linear

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TABLE 1 Median and extremes of soil properties and plant-available nutrients for mineral and peat topsoil layers (0–10 cm) across the 23 sites of the chronosequence.

	Mineral soils			Peat soils		
	Median	Minimum	Maximum	Median	Minimum	Maximum
Sand (%)	58	36	76			
Silt (%)	37	21	56			
Clay (%)	4	2	10			
pH	5.6	5.3	7.2	5.6	4.9	6.3
BD ($g\,cm^{-3}$)	1.23	0.96	1.52	0.56	0.37	0.84
P ($g\,kg^{-1}$)	0.07	0.03	0.23	0.06	0.04	0.12
K ($g\,kg^{-1}$)	0.12	0.04	0.44	0.19	0.10	0.30
Mg ($g\,kg^{-1}$)	0.13	0.04	0.23	0.27	0.14	0.54
Ca ($g\,kg^{-1}$)	1.20	0.20	5.14	3.78	1.96	9.24
Al ($g\,kg^{-1}$)	0.32	0.20	0.78	1.17	0.94	1.63
Fe ($g\,kg^{-1}$)	0.37	0.15	0.52	0.81	0.37	1.13
SOM (%)	7.9	4.1	12.5	46.7	32.1	73.4
Total C ($g\,kg^{-1}$)	36.5	19.2	66.3	242.4	181.2	425.0
Total N ($g\,kg^{-1}$)	2.9	1.6	4.8	12.9	8.8	17.5
C:N ($g\,g^{-1}$)	11.7	10.7	15.6	19.6	16.6	24.2

least square regression of total C vs. SOM contents was calculated using data from three soil profiles in every category of the chronosequence and all topsoil samples (Figure S2),

$$SOC_C = 0.556 \times SOM_C - 0.596 \quad (1)$$

Equation 1 can be applied to our data because the lowest SOM_C value in our dataset is higher than the value below which Equation 1 yields negative SOC_C estimates. However, Equation 1 would need to be recalibrated for soils with lower SOM content.

The complete dataset is available from the Bolin Centre Database (https://bolin.su.se/data/johansson-2023-soil-1).

2.4 | Topographic wetness index

To analyse the influence of catenary position on the SOC stocks, the TWI of each sampling site's centre point was used as a soil moisture proxy (Kopecký et al., 2021). Higher TWI indicates higher soil moisture content, which would be found in lower elevations and at low slope angles. TWI was derived from calculations based on the methods from Beven and Kirkby (1979) and Sørensen et al. (2006), and the values for each sampling point were assumed to be representative for all depths in the soil profile. A Digital Terrain Model (DTM) with a spatial resolution of 2 m was used in ArcGIS Pro 2.8.0 (Environmental Systems Research Institute [ESRI], 2021) and the hydrological toolset in the spatial analyst toolbox.

2.5 | Estimation of soil organic carbon stocks

For each sampling depth, SOC stocks (SOC_S, in kg m^{-2}) were calculated from the SOC contents (in %) at each depth (Livsey et al., 2021),

$$SOC_{S,j} = SOC_{C,j} \times BD_j \times \Delta D_j / 10 \quad (2)$$

where j is the sampling depth, BD_j is the dry soil BD ($g\,cm^{-3}$), and ΔD_j is the thickness of the sampled layer (either 10 or 20 cm). For all statistical analyses, BD and SOC contents and stocks in the two top layers were aggregated to have a topsoil layer thickness of 20 cm consistent with the thickness of the deeper layers. Aggregation was done by averaging the BD and SOC content values (BD weighted average for the latter), and by summing the SOC stock values in the two top layers.

Comparison of SOC stocks over a fixed soil depth across sites that have undergone surface displacement can be problematic, as it does not allow accounting for SOC lost because of subsidence. This is often the case in peatlands where oxidation of organic soil may cause a lowering of the physical soil surface (Plaza et al., 2019). Similar issues arise when comparing sites with decreasing BD as TSC from cropland progresses. In this case, some SOC remaining at the bottom

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of the profile would not be counted in the lower-density soils (Wendt & Hauser, 2013). To reduce these confounding effects, SOC stocks can be calculated at a fixed soil or ash mass (ash is the mineral soil particle residues that remain after loss on ignition at 550°C). Here we modified the procedure described by Plaza et al. (2019) as follows. First, we calculated cumulative soil and ash mass ($\mathrm{kg\,m^{-2}}$) from the surface to the layer z of the sampled profiles,

$$M_z = \sum_{j=1}^{z} \mathrm{BD}_j \times \Delta D_j \times 10, \tag{3}$$

$$A_z = \sum_{j=1}^{z} \left(1 - \frac{\mathrm{SOM}_{\mathrm{C},j}}{100}\right) \times \mathrm{BD}_j \times \Delta D_j \times 10, \tag{4}$$

where M_z and A_z indicate the soil mass and ash mass to depth z, $1 - \frac{\mathrm{SOM}_{\mathrm{C},j}}{100}$ is the ash mass fraction, and j is an index representing soil layers above z. Second, the cumulative SOC stocks to depth z were calculated as

$$\mathrm{SOC}_{\mathrm{S},z} = \sum_{j=1}^{z} \mathrm{SOC}_{\mathrm{C},j} \times \mathrm{BD}_j \times \Delta D_j / 10, \tag{5}$$

and were plotted against M_z or A_z. A hyperbolic function was fitted to the data to describe the relation between $\mathrm{SOC}_{\mathrm{S},z}$ and M_z or A_z,

$$\mathrm{SOC}_{\mathrm{S},z} = \frac{ax}{b+x}, \tag{6}$$

where x represents either M_z or A_z. Finally, this function was used to calculate $\mathrm{SOC}_{\mathrm{S},z}$ at fixed values of M_z or A_z for all sites. This approach is less sensitive to uncertainties in the data than using splines (as in Wendt & Hauser, 2013) or linear interpolation (as in Plaza et al., 2019). As reference soil and ash mass, we set M and $A = 200\,\mathrm{kg\,m^{-2}}$, approximately corresponding to the top 20 cm of the mineral soils (but a thicker soil layer in the peat soils). The chosen value of A required extrapolating Equation 6 for the peat soils where the mineral fraction was very low, but lower reference values that required minimal or no extrapolation did not change the final results.

2.6 | Statistical analysis

The effects of TSC, depth along soil profile, and TWI on BD and SOC contents and stocks were analysed with a linear mixed effect (LME) model (Davison, 2003),

$$\begin{aligned} Y_{i,j} &= \alpha_0 + \beta_{\mathrm{TSC}}\mathrm{TSC_i} + \beta_{\mathrm{D}}D_{\mathrm{j}} + \beta_{\mathrm{TWI}}\mathrm{TWI_i} + \beta_{\mathrm{Peat}}\mathrm{Peat}_{i,j} \\ &+ \beta_{\mathrm{TSC,D}}\mathrm{TSC}_i D_j + \beta_{\mathrm{TSC,TWI}}\mathrm{TSC}_i\mathrm{TWI}_i + \beta_{\mathrm{TSC,Peat}}\mathrm{TSC}_i\mathrm{Peat}_{i,j} \\ &+ \beta_{\mathrm{D,TWI}}D_j\mathrm{TWI}_i + \beta_{\mathrm{D,Peat}}D_j\mathrm{Peat}_{i,j} + \beta_{\mathrm{TWI,Peat}}\mathrm{TWI}_i\mathrm{Peat}_{i,j} \\ &+ \delta_i + \epsilon_{i,j} \end{aligned} \tag{7}$$

where Y is the SOC stocks, SOC content, or BD at site i (1–23) and sampling depth j (10, 30, 50 cm); α_0 represents the intercept; TSC_i indicates the time since conversion for each site (75, 55, 25, and 0 years for the category of continuously cultivated croplands); D_j represents the centre of each sampling depth (the first two sampling depths are aggregated to have equal layer thicknesses in the model, so $D_j = 10$, 30, and 50 cm); TWI_i is the topographic wetness index for each site; $\mathrm{Peat}_{i,j}$ is a categorical variable indicating mineral (Peat = 0) or peat soil (Peat = 1). The subsequent terms represent the interactions between the fixed factors. For each independent variable, β is the regression coefficient. The site is considered as a random effect (δ_i represents the site number, 1 to 23) to account for the correlation of soil properties in the soil profile at each sampling site. Finally, ϵ_{ij} is the error term. SOC contents and stocks were log-transformed before fitting the LME model to ensure the normality of the residuals, which was checked visually using quantile-quantile plots. The LME model was fitted using the function *fitlme* in Matlab (R2018b).

Total SOC at fixed depth, soil mass and ash mass (Y_i) were predicted using a linear model,

$$\begin{aligned} Y_i &= \alpha_0 + \beta_{\mathrm{TSC}}\mathrm{TSC}_i + \beta_{\mathrm{TWI}}\mathrm{TWI}_i + \beta_{\mathrm{Peat}}\mathrm{Peat_i} \\ &+ \beta_{\mathrm{TSC,TWI}}\mathrm{TSC}_i\mathrm{TWI}_i + \beta_{\mathrm{TSC,Peat}}\mathrm{TSC}_i\mathrm{Peat}_i \\ &+ \beta_{\mathrm{TWI,Peat}}\mathrm{TWI}_i\mathrm{Peat}_i + \epsilon_i \end{aligned} \tag{8}$$

without site as a random factor because only one total SOC stock value was calculated for each site. The meaning of the model coefficients is analogous to that of coefficients in Equation 7, except that here depth is not included as a fixed factor. A simpler version of Equation 8 was also tested, where only mineral soils were considered (i.e., $\beta_{\mathrm{Peat}} = \beta_{\mathrm{TSC\cdot Peat}} = \beta_{\mathrm{TWI\cdot Peat}} = 0$). Total SOC stocks were log-transformed before fitting the linear model and normality of the residuals was checked visually. The linear model was fitted using the function *fitlm* in Matlab (R2018b).

In both models, we initially retained all interaction terms, but then systematically removed interactions that were not significant to increase model robustness, as long as the Akaike Information Criterion continued to decrease. Effects are regarded as significant when $p < 0.05$ and marginally significant when $0.05 < p < 0.1$.

3 | RESULTS

3.1 | Site characteristics

Soil texture was generally dominated by sand or silt in the top 0–10 cm of soil (Table 1). The BD was lower and SOC contents and the organic C to total N (C:N) ratios were higher in the peat than in the mineral soils (as expected).

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In the topsoil, BD and pH tended to decrease, while SOC contents increased with TSC (Table S1; detailed analyses are shown in Sections 3.2 and 3.3). In contrast, the C:N ratio remained stable after conversion, suggesting that SOM quality did not markedly change along the chronosequence (Table S1). Despite differences between mineral and peat soils, there were no clear patterns in plant-available nutrients along the chronosequence that could indicate more or less favourable conditions for vegetation, which could have driven SOC changes.

3.2 | Vertical profiles of soil organic carbon and bulk density

The SOC contents generally decreased with depth, regardless of TWI (Figure 2). The decrease was less pronounced in the three upper sampling depths in the continuously cultivated croplands compared to the converted soils (Figure 2d). In the peat soil profiles, a slight increase in SOC contents with depth was apparent (Figure 2b,d). BD generally increased with depth along soil profiles in the mineral soils, whereas in the peat profiles, it decreased with depth (Figure S3). SOC concentrations (g C cm^{-3}), calculated as SOC content multiplied by BD, overall decreased with depth along the soil profile (Figure S4).

The average SOC stocks to 60 cm depth in the mineral soil profiles were between 12 and 18 kg m^{-2}, while peat profiles had much higher SOC stocks between 32 and 73 kg m^{-2} (Figure 3). There were no immediately visible trends in SOC stocks of either mineral or peat soils along the chronosequence, but TSC, topographic position, and site specificities are all confounded, requiring the statistical models of Equations 7 and 8 to separate these effects, as described next.

When using Equation 7 to predict BD and SOC contents and stocks (calculated at fixed depths) along the soil profiles, the only retained interactions were between peat and depth or TWI. As a result, the LME model was simplified by setting $\beta_{\mathrm{TSC,D}} = \beta_{\mathrm{TSC,TWI}} = \beta_{\mathrm{TSC,Peat}} = \beta_{\mathrm{D,TWI}} = 0$ in Equation 7. This simplified LME model confirmed that SOC contents and stocks decreased with depth and peat layers had significantly more C than mineral soils (Figure 4a). BD increased with depth in mineral soils, while it decreased with depth in peat soils (Figure 4a). SOC contents and stocks were higher at low topographic positions (high TWI) in mineral soils, whereas the effect of TWI on SOC contents was reversed in peat soils. TSC had no effect on SOC contents or stocks.

3.3 | Total soil organic carbon stocks

When using Equation 8 to predict total SOC stocks, only the interactions involving peat and TSC or TWI were retained, so the LME model was simplified by setting $\beta_{\mathrm{TSC,TWI}} = 0$ in Equation 8. The total SOC stocks calculated at a fixed depth were higher in peat soils than in mineral soils, and at higher TWI in mineral soils only (TWI had a negative effect on peat SOC stocks). When calculated at fixed soil mass, total SOC stocks increased with TSC and TWI in mineral soils ($0.05 < p < 0.1$) (Figure 4b).

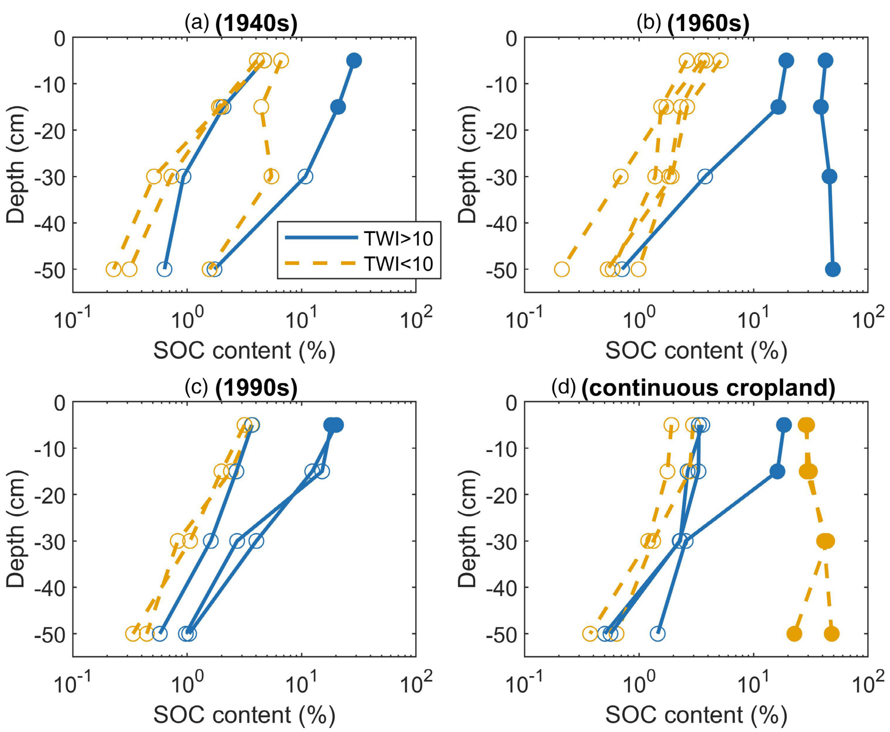

FIGURE 2 Variation in soil organic carbon (SOC) contents in vertical profiles for each site and time since conversion along the chronosequence. Filled symbols indicate peat layers (SOM content > 30%). The depth for each of the sampled soil layers corresponds to the centre of that layer: 0–10 cm (−5), 10–20 cm (−15), 20–40 cm (−30), and 40–60 cm (−50). TWI: topographic wetness index.

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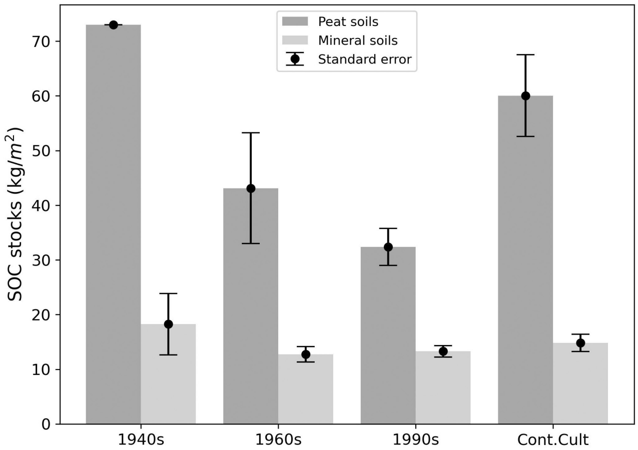

FIGURE 3 Average soil organic carbon stocks (SOC) in the 0–60 cm depth range per category for soil profiles containing at least one peat layer (dark grey) and mineral soil profiles (light grey) with corresponding error bars indicating the standard error (see number of replicates in Figure 2; note that there was only one peat soil in the 1940s category).

SOC stocks calculated at a fixed ash mass increased with TSC only in peat soils ($0.05 < p < 0.1$), but were not affected by TWI.

When considering only mineral soils in the regression, TSC and TWI effects on SOC stocks were statistically more significant and became significant for stocks calculated at fixed ash mass. Moreover, their effects were additive; i.e., the interaction between TSC and TWI was not significant. Based on results from this simple LME model, mineral SOC stocks calculated at fixed soil or ash mass gained approximately 50% of the initial SOC after conversion (or ~ 0.65% year^{-1}) (Figure 5b,c).

Taken together, these analyses show that mineral soil SOC stocks (calculated at fixed soil or ash mass) increased with TSC at all topographic positions. Peat SOC stocks calculated at fixed ash mass increased markedly after conversion, but this trend was only marginally significant (because of a low number of peat profiles and high variability among them).

4 | DISCUSSION

4.1 | Effect of time since conversion, catenary position and depth on mineral soil organic carbon storage

The expected increase in SOC in croplands that have been converted to grazed grasslands (Kätterer et al., 2008; Kim & Kirschbaum, 2015) was apparent in mineral soils in the studied chronosequence when calculating SOC stocks at a fixed soil or ash mass (Figures 4 and 5). However, the expectation of this increase being more pronounced in the topsoil (Don et al., 2009; Poeplau & Don, 2013; Wiesmeier et al., 2019; Yang et al., 2022) was not met. Our results also confirmed the hypothesis of finding higher SOC stocks in soils located in lower catenary positions (Li, McCarty, et al., 2018) (Figures 4 and 5). The soils in low catenary positions also gained more SOC after conversion than in upper positions (same relative change, but higher absolute stock change), highlighting the importance of catenary position when estimating the climate change mitigation potential of land use conversion.

The increase in SOC up to 75 years since conversion in mineral soils showed that these soils can accumulate C over a longer period than previous studies in Sweden have shown. Poeplau et al. (2015) showed that over 25 years (between 1988 and 2013) there was a general increase of 0.38% year^{-1} in SOC concentration in Swedish agricultural topsoil (0–20 cm) because of an overall increase in the ley area. This increase is approximately half as large as our estimate, but converted fields in our chronosequence were left undisturbed, whereas the data in Poeplau et al. (2015) were spatially aggregated and thus represent averages of converted and currently cultivated cropland. Changes in SOC during 31 years after cropland to grazed grassland conversion were studied by Kätterer et al. (2008) at a site approximately 250 km north of the studied sites. They estimated an average increase of 0.04 kg m^{-2}year^{-1} in the top 20 cm of soil, similar to the predicted increase between 0.03 and 0.05 kg C m^{-2}year^{-1} in this study (Figure 5). Our estimates are for SOC stocks calculated at fixed soil or ash mass, but in mineral soils, the chosen mass thresholds (200 kg m^{-2}) correspond to a depth of about 20 cm, so values in our study and reported by Kätterer et al. (2008) are comparable. However, our estimates are lower than the increase of 0.08 kg m^{-2}year^{-1} over 50 years that this type of conversion is expected to have on the SOC stocks according to IPCC (2000). Overall, these results suggest that conversion of croplands has the potential to sustain SOC accumulation over a 75-year period.

The effect of depth on SOC stocks throughout the chronosequence is negative in the linear mixed effect model (Figure 4), and consistent with the known patterns of decreases in SOC content with depth in mineral soils (Jobbágy & Jackson, 2000), despite the limited increase in BD (Figure 2 and Figure S3). The decrease in SOC contents with depth was less pronounced in the continuously cultivated mineral soils compared to soils converted to grazed grassland (Figure 2d). This could be explained by the higher incorporation of plant residues in the first two sampling layers, which encompass the typical plough depth of 20 cm. A high-density plough layer (plough-pan) is also expected in these soils beneath the plough depth because of compaction by agricultural machinery (Podder et al., 2012). After conversion, soil compaction would be ameliorated and BD would slowly decrease, as seen in our topsoil data (Table S1). BD

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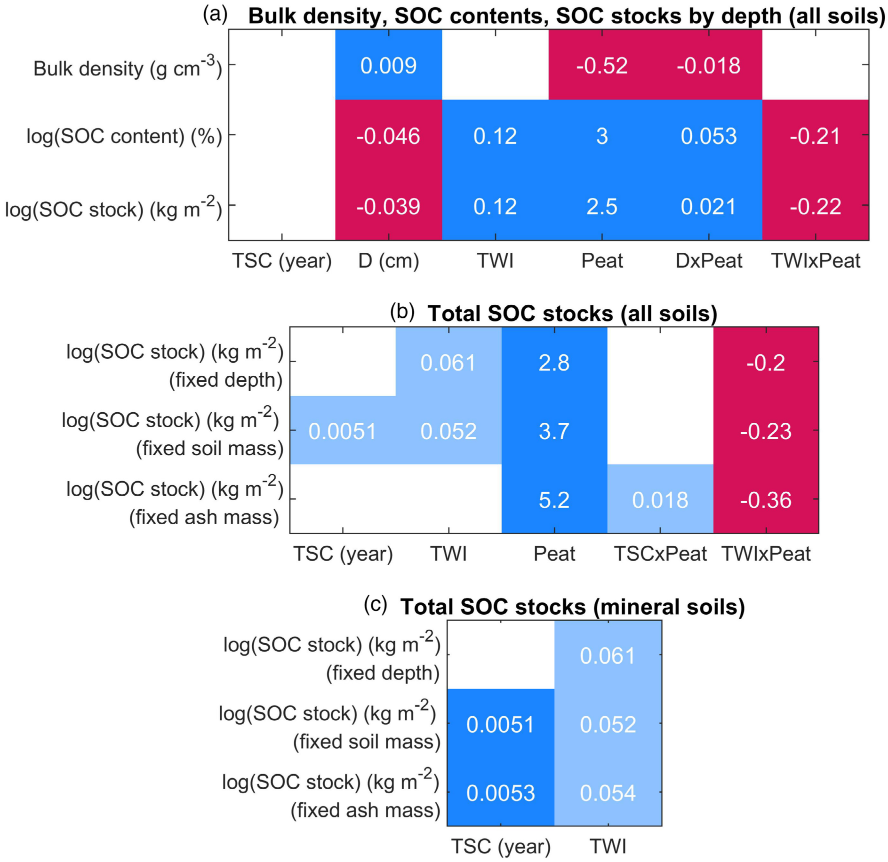

FIGURE 4 Coefficient estimates from the linear mixed effect model for: (a) soil properties along the soil profiles (Equation 7) and (b, c) total soil organic carbon (SOC) stocks at fixed depth, soil mass, or ash mass (Equation 8), when including all soils (b) or mineral soils only (c). In all panels, the predicted variables are on the y-axis (rows), and the fixed effects on the x-axis (columns; *D*, depth at the centre of each sampling interval; TSC, time since conversion; TWI, topographic wetness index; Peat, categorical variable indicating mineral (=0) or peat soil (=1), and their interactions retained after model simplification). Red and blue colours indicate respectively negative and positive effects (numbers are the coefficient estimates in Equations 7 and 8), whereas white indicates no significant effect on the predicted variables ($p > 0.1$); bright colours indicate significant effects ($p < 0.05$) and light colours marginally significant effects ($0.05 < p < 0.1$). In a, conditional $R^2 = 0.87$, 0.97 and 0.92 for bulk density, SOC contents, and SOC stocks, respectively; in b, $R^2 = 0.86$, 0.93 and 0.92 for total SOC stocks at fixed depth, soil mass, and ash mass, respectively; in c, $R^2 = 0.81$, 0.91 and 0.90 for total SOC stocks at fixed depth, soil mass and ash mass, respectively.

increases are mostly because of reduced soil macro-porosity in cultivated fields compared to grassland (Hu et al., 2022). After conversion, macro-pores can form again and BD decreases towards values close to natural conditions. While in other studies macro-porosity was mostly reduced by cultivation at 20–30 cm depth (Hu et al., 2022), in our study we found that BD decreased with TSC only when focusing at the topsoil (0–10 cm, Table S1).

4.2 | Effect of time since conversion, catenary position, and depth on peat organic carbon storage

In contrast to mineral soils, more SOC is found at depth in the peat soils, which also have much higher SOC stocks compared to the mineral soils (Figure 4). This is also in line with broad-scale patterns of peat accumulation, where compaction of the lower peat column over time increases the SOC stocks at depth (Hugelius et al., 2020). The statistical analysis suggests C gains in peat soils after conversion, but the trends are only marginally significant when stocks are calculated at fixed ash mass. However, this result should be interpreted with caution, because of the small number of peat samples and the extrapolations required to calculate SOC stocks at a fixed ash mass of 200 kg m^{-2}.

From a methodological perspective, calculating SOC stocks at fixed soil mass (e.g., Wendt & Hauser, 2013) or ash mass (e.g., Plaza et al., 2019) allows accounting for changes in BD and surface displacement that occur when peat is oxidized. In fact, peat oxidation caused by aerobic conditions because of drainage may result in changes to the soil depth, so that the absolute position of the soil surface may have shifted through time in the peat sites. The continuous cultivation of

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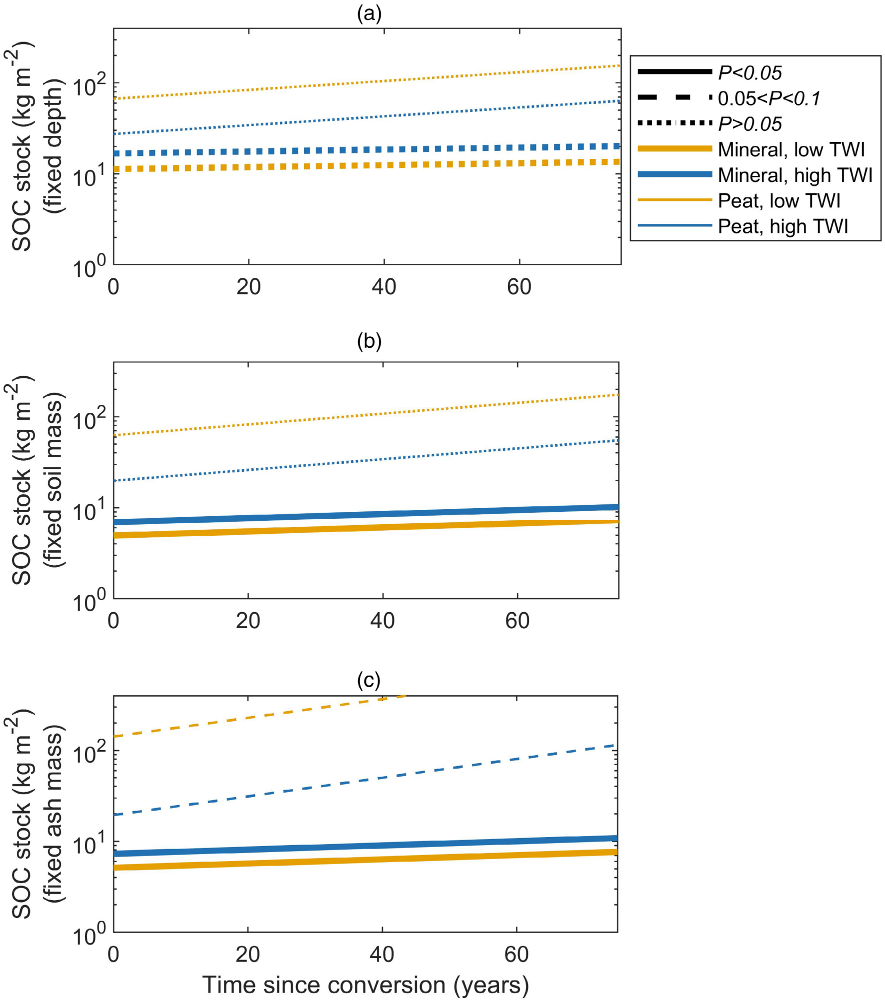

FIGURE 5 Predictions from Equation 8 of changes in total soil organic carbon (SOC) stocks in mineral soil (thick lines, coefficients and *p* values from Figure 4c) and peat soil (thin lines, coefficients and *p*-values from Figure 4b) along the chronosequence, at sites with low (orange lines) or high (solid blue lines) topographic wetness index (TWI). Total SOC was calculated at: (a) fixed depth of 60 cm, (b) fixed soil mass of 200 kg m^{-2}, and (c) fixed ash mass of 200 kg m^{-2}. Low and high TWI correspond to the first and third quartile of all the TWI values (5.23 and 11.73, respectively). Dotted, dashed and solid lines respectively indicate that the TSC effect was not significant ($p > 0.1$), marginally significant ($0.05 < p < 0.1$), or significant ($p < 0.05$).

the peat soils has likely caused both a net SOC loss (causing a lowering of the soil surface) and a compaction of the soil. Previous studies reported SOC losses after peatland drainage between 0.5 and 0.9 kg C m^{-2}year^{-1} in alpine and boreal climates (Grønlund et al., 2008; Rogiers et al., 2008). Using ash content accumulation in topsoils (following Rogiers et al., 2008), we estimated SOC losses ~40 kg C m^{-2} from peat soils in our chronosequence. These are net overall C losses since drainage started till today, as lack of data prevents us from distinguishing net C losses at different times since conversion. Assuming that drainage was initiated between 100 and 200 years ago, this amounts to a net SOC loss between ~0.2 and 0.4 kg C m^{-2}year^{-1}—lower than in the cited studies, possibly due in part to SOC recovery after conversion. While the results from peat soils must be interpreted with caution, we can at least conclude that there is no evidence of strong C losses after the conversion of peat soils.

4.3 | Implications for management

All former and present croplands in the studied area experience artificial drainage, which is more intense in the low-lying areas in the landscape where most peat soils are found. Our results show that mineral soils in low catenary positions could accumulate more SOC after conversion than in higher positions, making them suitable targets for conversion to grazed grassland. Converting peat soils to grazed grasslands could also increase SOC storage, but restoring hydrological conditions by closing ditches and tile drainages may have a much stronger effect than land conversion, as it would avoid continued C losses. However, any evaluation of climate change mitigation potential of wetland restoration should also include a full greenhouse gas balance, which we did not include in our study as we focus on changes in SOC stocks.

Earlier research has shown that it may be preferable to protect peat soils from aerobic conditions while under cultivation (Andrén et al., 2008; Kätterer et al., 2012). An alternative practice is seen in East Anglia, southeast UK, where former cropland on peat soil was converted to seasonally-inundated grazed grassland and despite this type of management, ~0.13 kg C m^{-2}year^{-1} were lost (Peacock et al., 2019). In contrast, a nearby fen maintained under natural hydrological conditions accumulated ~0.1 kg C m^{-2}year^{-1}. Thus, the combination of

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seasonally restored hydrological conditions and grazing practices could represent a good compromise to decrease C losses compared to only managing the peatlands with grazing livestock.

The conversion of mineral soils to grazed grasslands tends to increase SOC stocks, but without grazing, this increase might not occur (Li, Ciais, et al., 2018). These findings highlight the importance of grassland grazing for climate change mitigation. However, this conversion reduces the area available for crop production, which is important for food security (Bouma & McBratney, 2013), and grazing animals can contribute to greenhouse gas emissions, partly offsetting SOC gains (e.g., Rogiers et al., 2008)—a contribution that we could not quantify here. The conversion to grazed grassland is one of several possible management practices that can increase SOC in soils, but other practices such as cover cropping, optimized crop rotation, the addition of organic amendments and fertilizers, and agroforestry can be applied to croplands to possibly maintain or increase SOC storage (Bolinder et al., 2020; Chenu et al., 2019; Soussana et al., 2019). Soil flipping (mechanically burying topsoil and exposing subsoil) has also been adopted to reduce waterlogging and seems to promote SOC retention at depth and SOC storage in the new topsoil (Schiedung et al., 2019), but this practice is particularly invasive.

In our study, the croplands were converted because of the development of the farm machinery that made cultivation of large and uniform fields more profitable and to the increase in crop yields in the continuously cultivated fields, not because the converted soils were classified as less productive. This suggests that the local socio-economic context is important when analysing SOC storage, which is in line with the findings of Poeplau et al. (2015), and that in some contexts land use conversion can be implemented without compromising food security or the economy of the farms. Lastly, the great biodiversity values of grazed grasslands should not be neglected when planning on implementing this conversion. In Sweden, the grazed semi-natural grasslands have declined by approximately 90% during the last century (Eriksson, 2021), highlighting the need to expand these grasslands to conserve biodiversity in farmland landscapes.

4.4 | Sampling approach and interpretation of SOC predictors

The sampling strategy was based on randomly selecting 5–7 replicate sites for each category of TSC. The selected sites included both mineral and peat soils, which exhibit large differences in SOC storage because of their different SOC accumulation processes (Moyano et al., 2013; Singh, 2018). By studying the different soil types separately, a better insight into their SOC trends could be provided (Andrén et al., 2008)—indeed, when considering only mineral soils, SOC accumulation after land conversion was statistically more significant. A site selection approach targeting specifically peat or mineral soils could allow a more accurate quantification of SOC trends without the uncertainties introduced by an unbalanced sampling. Moreover, to fully quantify changes to peat soils, the whole peat column should ideally be sampled. Calculating SOC stocks at fixed ash mass allows using shallower sampling depths to compare soils with varying peat contents (Plaza et al., 2019), but this method still does not allow quantifying the effects of land use below the sampling depth.

While many studies only cover the top 30 cm of soil, which could lead to over- or underestimations of changes in whole-profile SOC stocks (Don et al., 2009; Yang et al., 2022), this study includes SOC changes further down in the soil profile with a sampling depth of 0–60 cm that covers the most active soil layers and allows both top- and subsoil analyses. One important reason for sampling subsoil is that the root biomass in soils converted to grassland is higher and reaches deeper layers compared to continuously cultivated croplands (Von Haden & Dornbush, 2019). Changes in the root biomass after conversion can lead to changes in both soil microbial activity and soil properties in the subsoil, which could result in both increases and decreases in SOC storage (Dijkstra et al., 2021; Lu et al., 2019). Also, by including measurements of the root C pool, potential differences between C inputs could be revealed that are not visible when only looking at the SOC stocks (Dietzel et al., 2017). Therefore, the inclusion of root biomass analyses and sampling below 60 cm to capture the role of deep roots (Rolando et al., 2021) could provide additional insights into SOC changes after the conversion of cropland to grazed grassland.

The sampling method could be improved by sampling transects to capture the intra-field topographic variability. Also, to account for the effect of artificial drainage of the fields, measuring the distance from the nearest ditch or stream to the sampling point could complement the TWI. Additionally, an extended analysis of the TWI based on bedrock topography would help capture variations in the subsurface water flows affecting SOC processes (Beven, 2012). Furthermore, TWI is not the best indicator of soil moisture conditions in flat areas because other variables such as groundwater table, vegetation, soil properties, and land use have a greater influence in these areas (Western et al., 1999). Therefore, future research could complement our findings by focusing on these factors, or

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others that could affect the SOC storage after conversions, such as livestock patterns, grazing intensity, manure input from livestock, slope and aspect, or microbial communities and their activity (Bot & Benites, 2005; Hewins et al., 2018; Maillard & Angers, 2014; Wiesmeier et al., 2019).

5 | CONCLUSIONS

This study investigated variation in SOC storage along a chronosequence of croplands that have been converted to grazed grasslands since the 1940s. SOC accumulated over decades within mineral soils, consistent with previous estimates within Sweden. This confirms the potential of land conversion to grazed grasslands within Sweden for climate change mitigation. When calculated at a fixed ash mass (instead of fixed depth), SOC stocks in peat soils also increased after conversion to grazed grassland, but uncertainties in the data prevent us from quantifying accurate changes. Despite some SOC gains in the peat soils, we speculate that the greatest benefit for these soils may come from the removal of drainage systems, rather than conversion to grazed grassland per se. When considering the demands placed on agricultural production, the conversion of arable land to grazed grasslands highlights the complexities and dependencies on local conditions (such as topography and soil type) in balancing the most appropriate management practice to achieve both food security and climate and biodiversity goals.

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CONFLICT OF INTEREST STATEMENT

The authors declare no conflict of interest.

DATA AVAILABILITY STATEMENT

The data that support the findings of this study are openly available in Bolin Database at https://bolin.su.se/data/johansson-2023-soil-1.

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