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First Signs That National Cropland Organic Carbon Loss Is Reversing in British Topsoils

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ABSTRACT

High rates of soil organic carbon (SOC) loss from cropland soils are well known, contributing to climate change and compromising soil and ecosystem health. Stabilising and reversing the loss of organic matter from cropland soils is a challenge for all nations to meet the United Nations Sustainable Development Goals. Sustainable land management (SLM) has been promoted as a mechanism of achieving this, but to date, there is no evidence of positive impacts at scale. Here we show the first signs of the reversal of soil carbon loss in cultivated topsoils in Great Britain, following a period of reported SLM uptake, using 40+ years of national soil monitoring from the UKCEH Countryside Survey. Following a prolonged historic decline at rates of $-0.16\,\mathrm{t\,ha^{-1}\,year^{-1}}$, there was a significant increase in cropland topsoil SOC stocks (0–15 cm) from 2007 to 2019–22 with an accrual rate of $0.17\,\mathrm{t\,ha^{-1}\,year^{-1}}$, approximately $0.74\,\mathrm{MtC\,year^{-1}}$ nationally. We discuss reported management shifts in Great Britain in the corresponding period and identify a reduction in conventional tillage and reduced straw removal as potential drivers, but highlight additional evidence gaps worthy of consideration. This increase in topsoil SOC may represent net carbon sequestration or carbon redistribution (geographic or vertical) but nevertheless demonstrates that topsoil properties can be restored at scale and offers hope that a concerted effort by land managers can halt, and potentially reverse, SOC loss from cropland soils.

1 | Introduction

Soil organic carbon (SOC) is widely recognised as an integral part of a soil's capacity to function and support ecosystem services (Blanco-Canqui et al. 2013; Kopittke et al. 2022; Powlson and Galdos 2023) and it is estimated that a third of global soils are rapidly losing this resource (FAO and ITPS 2015). As facets of "soil health", this includes the ability of soils to support agricultural productivity, biodiversity, and flood and drought resilience, which have each been specifically associated with the presence or restoration of soil organic matter (Lal 2004; Moinet et al. 2023). Soil carbon is the largest stock of terrestrial carbon, at ~2500 PgC in the top metre of soil alone (Lal et al. 2021), and efforts to not only protect this stock but to sequester atmospheric CO_2 in this form are widely pursued as a mechanism contributing to Net-Zero emissions. Soils used for intensive cultivation are particularly susceptible to SOC loss (Sanderman et al. 2017) and have become a hotspot for efforts to mitigate SOC loss in recent years. Soils furthest from their mineralogical capacity to store carbon (i.e., the most degraded) have also been shown to

Summary

- First sign that national trends of cropland SOC loss (11% from 1978 to 2007) have reversed in GB topsoils.
- Monitored SOC stocks increased by 0.34% annually ($0.17\,t\,ha^{-1}\,year^{-1}$) from 2007 to present.
- SOC recovery coincides with a period of sustainable management uptake in the UK.
- The result suggests that management can reverse SOC loss from topsoils in practice at scale.

have some of the greatest potential for SOC recovery (Georgiou et al. 2022). Estimates from a global meta-analysis suggest conversion of pasture to cropland causes a 60% reduction in carbon stocks (Guo and Gifford 2002), demonstrating that most cropland exists well below mineralogical capacity. Updating our understanding of the current status and trajectory of cropland SOC is key to establishing a robust baseline against which future restoration efforts can be assessed. When paired with historic data, monitoring cropland SOC can also show what has or has not been achieved by recent policy and management shifts.

The restoration of SOC in managed soils has been described as a challenge of "daunting complexity" (Moinet et al. 2023), and evidence of large-scale success for SOC restoration is lacking. SOC restoration is a priority under numerous recent policy initiatives and targets implemented at both national and international scales and underpinning multiple Sustainable Development Goals (in particular Zero Hunger, Climate Action and Life on Land) (DEFRA 2018; Panagos et al. 2022; Rumpel et al. 2018; UN General Assembly 2015). The objective of protecting and restoring SOC on managed soils is proposed through the use of a suite of sustainable land management (SLM) practices that minimise soil disturbance or enhance the amount of organic matter entering the soil. Existing guidance on SLM practice is variable, and data on uptake is difficult to obtain at national scales. Estimates from 2019 suggest that 48% and 7% of cropland in England is cultivated by minimum tillage and zero tillage, respectively, having increased from approximately 35% reduced tillage in the 1970s and a low of 25% in 1988 (Alskaf et al. 2020). A recent survey of UK cropland showed 92% of respondents consider themselves to be practising sustainable soil management in some form, although the management practices themselves were variable, with the most frequently used practices (40%–50% respondents) being manure application, cover crops and diversified rotations, and not all combinations covered all regenerative agriculture principles (Jaworski et al. 2023).

Despite this apparent success in SLM uptake, there remains widespread debate and critique of proposed managements and targets for SOC restoration. Alongside the advocates of SLM's potential for soil health (Padarian et al. 2022; Smith 2004; Smith et al. 2000), justifiable concerns have been raised about the ability of SLM regimes to meet policy targets for SOC restoration, in light of practical, economic, and biological limitations on the ground (Berthelin et al. 2022; Janzen et al. 2022; Moinet et al. 2023; Smith et al. 2005). A qualitative review of the evidence underpinning different managements for below-ground carbon sequestration in England found that the supporting primary evidence was often limited or highly context-specific for croplands (Bentley, Feeney, Matthews, Evans, et al. 2023). Further, a recent review of the global evidence supporting the use of cover crops for SOC restoration concluded that the evidence was generally weak and resulting effect sizes would be much lower than previously reported (Chaplot and Smith 2023). More broadly, the value of cropland soil management for carbon sequestration has been questioned, highlighting the challenges of sequestering and preserving carbon in cropland soils and instead calling for the prioritisation of soil health restoration (Baveye et al. 2023; Berthelin et al. 2022; Moinet et al. 2023). Thus far, there has been little empirical evidence of the potential for management change to impact SOC trends at scale. However, following decades of reported policy and management shifts in favour of SOC restoration, there is now an opportunity to examine the impact this has had at a national scale, with implications for the wider feasibility of cropland SOC restoration.

The UKCEH Countryside Survey (CS) is the world's longest running integrated national monitoring programme, which began in 1978 and has just completed a new monitoring cycle. It uses a stratified random design to assess the stock and change in soil health metrics at national scales. CS reported the loss of 11% of cropland topsoil SOC between 1978 and 2007 in Great Britain (GB) (Emmett et al. 2010). It has been estimated that soil degradation in England and Wales had a cost of £1.2 bn ($1.53 bn) per year in 2010, with the loss of organic matter contributing between £500 and 600 million (Environment Agency 2019). In 2019, given the importance of the SOC debate, CS evolved from decadal monitoring to a rolling monitoring program. Here we report on the latest findings from the new monitoring cycle (CS2020, conducted from 2019 to 2023) which covers a representative sample of soils in GB and includes samples from > 1600 plots to date (Figure 1). Previous monitoring cycles were conducted in 2007, 1998 and 1978 (Emmett et al. 2010) with components of the soils research published in Reynolds et al. (2013) and Keith et al. (2015). Using this unique dataset, national changes in SOC concentration and stock are reported for the last 15 and 45 years for a subset of the data relating to cropland systems and discussed in the context of shifting land management practices.

2 | Materials and Methods

The UKCEH Countryside Survey is a unique "audit" of the natural resources of the UK's countryside. The countryside is sampled and studied using scientific methods that allow us to compare current results with those from previous surveys. In this way, we can detect the gradual and subtle changes that occur over time. Monitoring campaigns in 1978, 1998, and 2007 were conducted within a single year, visiting all sampling units within the same calendar year. For current (CS2020) and future monitoring cycles, the UKCEH Countryside Survey has adopted a rolling annual programme to increase the granularity of the temporal resolution of the monitoring, making it more robust to unusual climatic conditions in a given year, which have the potential to influence soil pH and bulk density in particular. From the outset, the CS was designed to capture and integrate across multiple metrics (Carey et al. 2008; Wood et al. 2018). Currently, the scope of the monitoring programme has been reduced to focus on co-located soil and vegetation metrics, which have

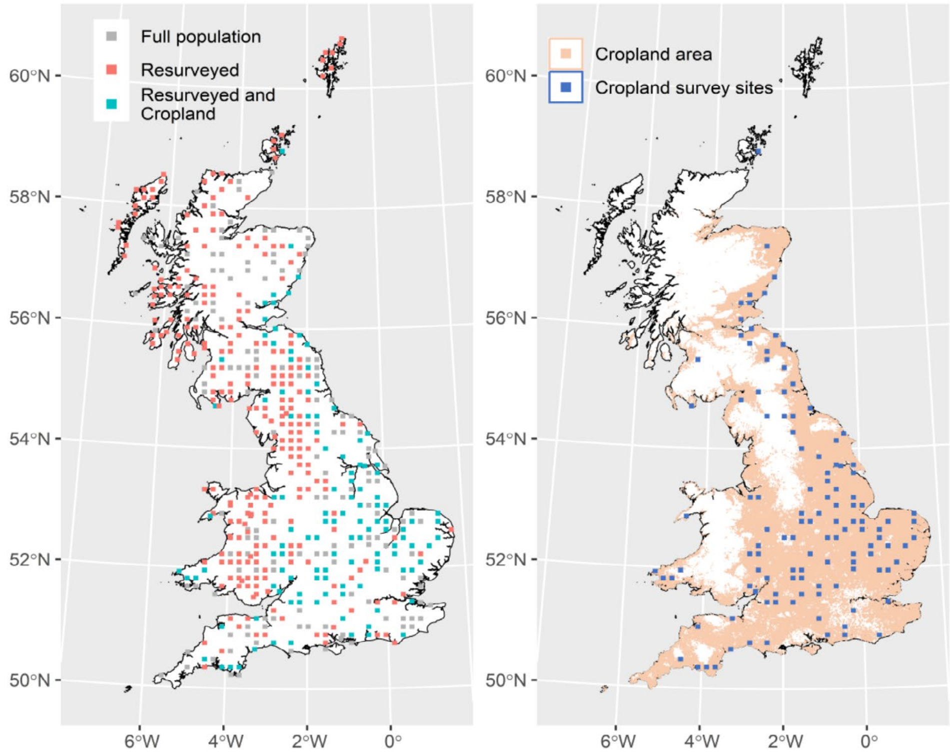

FIGURE 1 | Map of CS2020 survey squares, resurveyed locations (2019–2022) and resurveyed cropland sites (2019–2022) at 10 km resolution (left). Map of cropland sites over GB cropland area, provided by Land Cover Map 2020.

now been collected and published for the CS2020 monitoring cycle (vegetation data can be found in an accompanying data set on the Environmental Information Data Centre alongside the soils data).

2.1 | Countryside Survey Sampling Design

The Countryside Survey is designed to capture a nationally representative sample of the Great British countryside (therefore excluding marine and urban environments) and uses a stratified random sample design. The strata used capture fundamental sources of ecological variation within the UK (including soil forming factors), that are largely static and consistent over time, including parent material, relief, and climate (Bunce et al. 1996). Those strata are then sampled randomly and proportionally to strata land area, with more sampling units selected in the larger strata (although see below for exceptions to this prior to 2007 and how that is accounted for). The overall sample is considered to be representative of GB whilst ensuring that less common land classes are represented within the sample with greater sampling efficiency than a purely random sample can achieve (Robinson et al. 2024). The sampling units used in CS are 1 km×1 km squares corresponding to the British National Grid, with multiple plots established within each square to provide replication that are then revisited each monitoring cycle. Whilst replication at the square level is critical to the design, it is not with the intention of providing robust characterisation at the square level. Full replication, and therefore robust characterisation, is achieved at the strata level. The design has evolved over time with policy needs and computational capabilities, primarily to increase the total sample size of 1 km squares and increase the number of distinct strata, whilst maintaining the same core design principle and compatibility over time. A full record of changes to the survey design from CS1978 to CS2007 is provided in Barr and Wood (2011), with an overview of changes provided below. For additional information on the history and sampling design of the Countryside Survey otherwise beyond the scope of this paper, please see the Supporting Information.

The first CS was carried out in 1978 using 256 × 1 km squares across 32 ITE land classes (Table S1, Bunce et al. 1996, 1990). In 1978, 8 squares per strata were sampled, which was determined as the minimum viable sampling intensity for reporting. However, this resulted in an over-representation of smaller land classes, which was accounted for in design-based analyses by incorporating weights relating to inclusion probabilities. Soil organic matter was sampled via loss on ignition for those same 256 squares (8 × 32 strata) in both CS1978 and CS1998. In CS2007, a superior approach could be adopted with increased resource availability and increased reporting requirements, where each strata was sampled according to its size. As a consequence, the number of 1 km squares increased to 591. This larger sample included all pre-existing squares but added additional squares so that each land class is sampled relative to its area and to ensure compatibility with more resolved strata. At the same time, the original 32 land classes were refined to produce 45 classes in the current ITE Land Classification of Great Britain (Bunce et al. 2007) that are compatible with reporting for Great Britain and the devolved nations. These 45 classes, which can be mapped easily to the original 32, have remained in use from CS2007 onwards (Barr and Wood 2011). In CS2020, 500 1 km^2 squares were sampled, selected from within the 591 surveyed in 2007 to preserve the relative sampling efforts per land class. All of the original squares from 1978 were prioritised for resurvey to give a repeat sampling over the longest period to identify change in subsequent analysis. Consistent with the definition of the sampling domain since the inception of CS (i.e., the exclusion of urban and marine environments), any square that now contained more than 75% of urban land or that was more than 90% sea (defined by UKCEH Land Cover Map 2007 and the UK Census mean high tide data) was excluded from the sample. Although the sampling coverage of the strata was not directly representative in the 1978 and 1998 datasets, the analytical approach described below mitigates the effect of this on contemporary analyses (Scott 2008; Black et al. 2008).

To provide replication for the sampled strata, five plots were established within each 1 km square using a random sample from within five equal-area segments within the square (Wood et al. 2018). These plots are revisited in each subsequent monitoring cycle, subject to changing access permissions, for the purpose of co-located soil and vegetation sampling. These are referred to as plots in this paper, but are called 'X plots' or Walley plots in other publications from the CS (Emmett et al. 2010; Maskell et al. 2008; Wood et al. 2018). Plot relocation is achieved through the use of detailed field sketch maps, photographs and in modern monitoring cycles, GPS (from 1998 onwards) (Smart et al. 2023). Plots also contain a metal plate to facilitate accurate relocation, using a metal detector. Additional quality assurance is carried out after the field survey to confirm plot relocation by inspecting site photos, field maps and GPS data for consistency between monitoring cycles. If it is suspected that a plot has not been accurately relocated in a given monitoring cycle, then associated records are assigned a new identity for analysis and not considered a repeat of previous sampling. Data from CS2007 showed 85% of plots were relocated over all habitat types (Wood et al. 2018). A further description of the plot relocation procedures is published in Wood et al. (2018). A summary of the number of 1 km squares and plots in each monitoring cycle of the Countryside Survey is provided in the supplement (Table S1) along with further information about the Countryside Survey methodology in Section 2.3.

2.1.1 | Soil Sampling

Soil samples (0–15 cm) were collected from the five randomly located plots within each survey square, subject to access permission from landowners, using a 15 cm long × 5 cm diameter cylindrical soil core (Robinson et al. 2023). In practice, an average of 4.4 plots were visited per 1 km square across all habitats in CS2020. At each plot, soil cores are sampled from 15 cm outside of a central 2 m × 2 m quadrat used for vegetation monitoring and within the bounds of a larger quadrat which extends up to 14 m × 14 m (Wood et al. 2017). In each monitoring cycle, the corner of the 2 m × 2 m quadrat from which the soil samples are taken is rotated relative to the previous monitoring cycle. Whilst this introduces short range variation (2–3 m) between resamples, it was considered a higher priority to minimise the chance that samples were affected by disturbance in prior monitoring visits. All measurements used in this analysis are collected from a single 15 cm core per plot and are thus directly comparable.

Pilot studies of the sampling method have concluded a single core is sufficient in the context of the monitoring design and objectives, providing the required consistency in measurement across a five different soil types whilst being feasible for the field teams to implement in practice when compared to five different core sampling approaches and a soil pit extraction method (Emmett et al. 2008). We acknowledge that the use of a single core will introduce additional variability to repeat samples for a given plot (Lark 2012), as a result of short-range variation in soil properties, but (a) this is not expected to introduce any systematic bias to the dataset and (b) this variation is taken into account when determining the sampling effort (squares) across the survey to ensure we retain sufficient analytical power. Whilst additional cores are collected at each plot for the purposes of archiving dried and frozen soil material in the UKCEH soil bank, those cores are protected for future use by the wider scientific community. The number of additional cores taken for archiving has varied over monitoring cycles (up to 4 in CS2020) and these are taken an additional 15 cm away from the corner of the 2 × 2 quadrat. All soil cores were collected between the months of April and September in each year, with the timing of each sample‘s collection designed to coincide with the collection of that same sample in the previous monitoring cycle where possible.

2.1.2 | Habitat Allocation

In addition to the soil sampling and vegetation monitoring, habitats are also assigned by surveyors in the field following the Joint Nature Conservation Committee UK BAP Broad Habitat classification (Jackson 2000). Habitats were last mapped across each 1 km square in CS2007 where surveyors assigned habitats at a minimum spatial resolution of 400 m^2 (Wood et al. 2018). In CS2020, surveyors updated the habitat assignment of plots using the same method, without mapping the entire 1 km square. Prior to CS2007, habitats were mapped following a pre-existing classification which was subsequently mapped to the broad habitat classification as described in Wood et al. (2012).

To understand the potential impact of fine scale spatial variation in land management on data in subsequent analyses, in CS2020 surveyors also assigned a broad habitat based on the immediate sample sites, corresponding to the area inside the paired vegetation plots. A comparison of these two classification scales demonstrated a low impact of spatial heterogeneity on the attributions of broad habitat in the data, with 98% of arable and horticultural plots assigned to the same habitat at a smaller spatial resolution. Plots in an arable and horticultural broad habitat are hereafter referred to as cropland plots and soils.

2.2 | Soil Processing

2.2.1 | Loss on Ignition

For each soil core, loss on ignition (LOI) was measured to quantify soil organic matter content among other measurements which can be found with the data submission on the Environmental Information Data Centre (see below). A detailed explanation of laboratory methods used, and subsequent calculation of soil organic carbon concentration and stock are provided in Emmett et al. (2008). All methods were identical to those described in Emmett et al. (2008), with the exception that a Thermogravimetric Analyser (LECO TGA701, henceforth TGA) was used in CS2020, in place of the muffler furnace used in CS2007, to measure LOI via the procedure described below following in-depth comparison work.

To measure LOI, samples were loaded into a TGA and heated from 25°C to 1000°C in four steps (Lebron et al. 2024). In the first step the furnace reaches 105°C and temperature is then kept constant for 3 h. In the second step, samples are heated to 375°C and sustained at a constant temperature for a total of 15.45 h (Ball 1964). This procedure was used for almost 50 years in the CS to maintain methodological continuity and comparability in long-term results. The mass lost between 105°C and 375°C gives the soil organic matter content as LOI:

$$\text{LOI (g per 100 g)} = 100 \times (\text{TGA\_105} - \text{TGA\_375}) / (\text{TGA\_105})$$

where TGA_105 is the mass of dried soil after heating to 105°C for 3 h and TGA_375 is the mass of soil after organic carbon ignition and being kept at 375°C for 15.45 h.

2.2.2 | Comparing TGA and Muffler Furnace Measurements of LOI

To determine whether this change in instrument had a significant impact on the values of LOI, and thus SOC, we compared measurements of LOI using both methods for 500 samples collected in 2019, 103 of which were cropland soils. The error between the two datasets was non-normal (Shapiro–Wilk $W = 0.96$, $p < 0.01$), a disagreement between the two datasets was assessed using a paired Wilcoxon signed rank test.

We found no significant difference in the measurements of LOI for the cropland samples ($p = 0.55$) but acknowledge that the lack of a detected significant difference does not imply no difference. The mean difference was less than 0.004 g per 100 g of soil, compared to a mean LOI (TGA) for arable of 5.2 g per 100 g soil. However, there was a significant difference across all habitats ($p = 0.02$), where the muffler furnace reported on average larger values than the TGA and the mean difference was 0.114 g per 100 g soil. This could be explained by greater rates of water resorption by oven-dried organic matter during weighing when using a muffler furnace, erroneously increasing the subsequent measurement of organic matter loss. When using a thermo-balance TGA, the samples are not removed from the furnace for weighing, whereas this is required when using a muffler furnace, although sample mass was measured in a desiccator to reduce this effect in both previous CS cycles and this analysis.

In support of this, visual inspection of the data suggested that the error was driven by samples with high absolute LOI (above 75% OM). We found no significant bias in samples where LOI (muffler furnace) was less than 75 g per 100 g ($n = 397$, $p = 0.21$), whereas there was a significant difference when LOI was greater than 75 g per 100 g ($n = 78$, $p < 0.0001$). Where LOI was > 75 g per 100 g, LOI from the muffler furnace was, on average, 0.61 g per 100 g greater than that measured by the TGA, representing an average percentage bias of 0.66%, relative to the muffler furnace measurement. The habitats represented in this subset were primarily bog, coniferous woodland, and dwarf shrub heath. This suggests that in future analyses with the CS, a correction may be needed to facilitate comparisons over time for habitats where mean LOI is high, although this will only impact the detection of very small trends and has no impact on the present analysis of cropland soils.

2.2.3 | Bulk Density

Prior to 2007 (CS1978 and CS1998) bulk density of the fine earth fraction was not measured from cores but was instead predicted from LOI (Emmett et al. 2010). From 2007 onwards, the decision was taken to improve the reliability of SOC stock estimates by measuring bulk density independently of LOI.

For soil samples collected in CS2007 and CS2020, bulk density was measured from the same soil cores that are subsequently used for LOI measurements. Prior to the introduction of this change in methodology, a variety of core designs and sampling methods were tested with replication across 5 different soil types (clayey soil, sandy soil, peaty soil, stony soil and a woodland loam) to identify a reliable and practically feasible method of sampling bulk density in a consistent manner at the scale required by the survey described in Emmett et al. (2008). From this work, it was concluded that using a single 15 cm long and 5 cm diameter core per plot provided a feasible and sufficiently representative sample of bulk density with suitable surveyor training. Bulk density of the fine earth was then calculated as the mass of oven dry soil per unit volume, corrected for stone mass greater than 2 mm, extracted through sieving:

$$\textit{Bulk density}\ \left(g\ cm^{-3}\right) = \frac{\textit{Dry soil mass} - \textit{stone mass}}{\textit{fresh soil core volume} - \textit{stone volume}}$$

where dry mass of soil is the oven-dry mass (g) of soil after air drying in the laboratory to constant mass, then drying at 105°C for 3 h, and fresh soil core volume (cm^3) is the soil core volume prior to drying. Core volume is calculated using the formula for the volume of a cylinder with a diameter of 5 cm and a length of the sampled core. The volume of stones is calculated from their mass and an assumed density of 2.65 g cm^{-3}.

2.2.4 | Calculating SOC From LOI

A conversion factor of 0.55 was used to convert LOI (g per 100 g) to carbon concentration (g per 100 g), as described in (Emmett et al. 2008). This conversion was derived using the relationship between total carbon (g per 100 g) and LOI (g per 100 g) observed in CS 1998 and CS 2007, as described in Emmett et al. (2010). We have confirmed that the same relationship holds for CS2020, using a linear model with a zero intercept ($r^2 = 0.99$), where:

$$\textit{Carbon Concentration} = 0 + 0.55\ \textit{LOI}$$

Carbon stock (t ha^{-1}) of the fine earth fraction is subsequently calculated from carbon concentration (g kg^{-1}) and bulk density (g cm^{-3}) of the fine earth fraction via the following equations:

$$Carbon\ Stock = 1.5 \times Bulk\ Density \times Carbon\ Concentration$$

2.3 | Data Used in This Analysis

The CS2020 monitoring cycle spans from 2019 to 2023; however, at the time of writing, only data from 2019 to 2022 (382 1 km squares, > 1600 soil samples) is available for analysis, along with the corresponding data from previous monitoring cycles. The squares for each annual survey are selected to be a representative subset with some constraints (e.g., pairing two otherwise remote and isolated sites together within a single monitoring year) due to the practical limitations of the field survey campaign. However, the schedule for CS2020 was adapted in the face of the COVID-19 pandemic to include a reduced population in 2020 (in addition to the introduction of additional safeguards in line with government guidance at the time). As a result, more than 100 squares were surveyed in some subsequent year of the monitoring cycle. Because of the statistical approach taken to selecting each annual population, we do not anticipate the omission of data from 2023 will have biased the trends detected below. Data on the differences between the resurveyed populations and the sites yet to be resurveyed is provided in the Supporting Information.

Of the 1600 soil sample plots across 382 squares visited from 2019 to 2022, 316 (19.1%) were on cropland and 226 (13.7%) were on cropland in all visits made to that site (i.e., there has been no record of land use change within CS), and these latter plots are henceforth referred to as stable cropland soils. We also restricted the available data to squares that had been revisited within 2019–2022 (Table S2). This means that reported mean estimates for CS1978–CS2007 may differ from previous reports but will reflect the population surveyed in 2019–2022 and thus be directly comparable. In this analysis, we are making the assumption that the cropland sites within this nationally representative data set are also representative of cropland in GB, with the potential implications of this assumption discussed further. Sample sizes for cropland soils and squares in each monitoring cycle used in the analysis are provided in Table 1, with additional information on sample numbers from the CS more generally provided in Tables S1 and S2.

2.4 | National Change in SOC

Change in SOC concentration and stock were modelled as a function of monitoring cycle (CS1978, CS1998, CS2007 and CS2020) for (1) all cropland plots and (2) stable cropland plots (as previously defined). To estimate magnitudes of change in SOC concentration and stock, respectively, we used linear mixed effects models with monitoring cycle (CS1978, CS1998, CS2007, CS2020) as a fixed effect factor. Random intercepts were normally distributed, including an adjustment for square and plot, using the R package nlme (Pinheiro et al. 2023). This approach is robust to an incomplete resurvey of all plots in all years, due to variable landowner permissions or habitat change (Scott 2008). An AR1 correlation structure was used to correct for temporal autocorrelation at the site level, thus assuming that the covariance between repeat measures across successive monitoring cycles is constant. This approach assumes squares to be independent of one another as the random effects are included as unstructured, independent and identically distributed (i.i.d) draws rather than as structured effects. The same modelling approach was used to estimate the change in measured bulk density from CS2007 to CS2020. With the inclusion of square level i.i.d random effects, it is unlikely that there would be any remaining spatial structure present in the data. However, to check this and hence determine whether spatial autocorrelation was present in the residuals of each model, we used Moran's I test for spatial autocorrelation for 20 evenly spaced distance lags between 10 and 100 km using R packages spdep (Bivand and Wong 2018). The incorporation of the square level random effects structure and a temporal auto-correlation element also enables the models to effectively correct for the "missing squares" in the 1978 and 1998 sample (as described above) and for the impact of any plots missing in a given monitoring cycle, using the structure of the data in other survey years to provide consistent and comparable estimates over the multiple surveys. This approach has been adopted since 2007 to maximise the use of all available data in producing estimates of status and change in different metrics, following a period of extensive testing and comparison with previous design-based estimates (Scott 2008; Black et al. 2008). All analysis was conducted using R version 4.4.0 (R Core Team 2023). The generic model structure used is provided in the

TABLE 1 | Numbers of 1 km squares and plots in each monitoring cycle that contained soil sampling plots on cropland soil, subsequently used in the analysis for cropland soil organic carbon stends in Great Britain from CS1978–CS2020.

		Number of cropland soils		Number of stable cropland soils	
Monitoring cycle	**Sampling year**	**1 km squares**	**Samples**	**1 km squares**	**Samples**
CS1978	1978	48	89	29	49
CS1998	1998	52	141	42	101
CS2007	2007	114	308	90	245
CS2020	2019–2022	118	316	86	226
CS2020	2019	38	103	29	76
	2020	17	38	12	24
	2021	30	79	22	60
	2022	33	96	22	66

Note: For additional information on full sample sizes for CS monitoring cycles see Table S1.

following equation, where y_{ijk} is the response variable for square i, plot j and monitoring cycle k, $Cycle_k$ is the effect of monitoring cycle k, α is the random intercept error structure which varies by square i and plot j, β is the AR1 temporal autocorrelations structure, which varies with square i, plot j and cycle k, and ε is the residual error:

$$y_{ijk} = Cycle_k + \alpha_{ij} + \beta_{ijk} + \varepsilon$$

Both SOC concentration and stock were analysed following a logarithmic transformation to satisfy the assumptions of normality within the models, following an initial inspection of the model residuals and random effect distributions using the qqnorm function (Pinheiro et al. 2023). When using a log transformation, the reverse transformation to the original scale is known to introduce sub-estimation (Baskerville 1972), thus we use a correction term for linear mixed effect models when reporting on the original scale (Ramírez-Aldana and Naranjo 2021) (Equation S1). The correction reduced the root mean squared error (rmse) between model predicted and observed values from 189.5 to 11.99 g kg^{-1} and the mean bias from −132.7% to 33.3% for SOC concentration and from 6.32 to 6.16 t ha^{-1} (rmse) and −82.3% to 11.1% (bias) for SOC stock. The correction had no impact on the significance of any change detected between monitoring cycles. The correction factor for the fixed effects was 1.04 for SOC concentration and 1.01 for SOC stock (Ramírez-Aldana and Naranjo 2021).

We first tested the null hypothesis that mean response values were equal across all monitoring cycles using ANOVA and, having refuted the null hypothesis, subsequently used the Tukey Post Hoc test via the R package multcomp to determine whether there was a significant difference between each successive monitoring cycle and over the entire monitoring period (CS1978–CS2020) (Hothorn et al. 2008). Use of the Tukey Post Hoc test accounts for the implicit multiple comparisons, hence multiple statistical tests, conducted when looking at the difference in means across the monitoring cycles. In addition to this, the differences between monitoring cycles were then verified using non-parametric bootstrap resampling over 1000 iterations to generate 95% confidence intervals for the change between monitoring cycles using the R package boot (Canty and Ripley 2022). The addition of the bootstrap approach, free from the distributional assumptions made in the post hoc tests, provided robustness and a conclusive assessment of the hypotheses of interest. In all cases, bootstrap resampling and post hoc testing reported the same significant changes between monitoring cycles, as reported in Table 2. Confidence intervals reported in Table 2, Figures 2 and 3 were produced via bootstrap resampling, with the p values derived from the Tukey multiple comparison tests. Subsequent results are, in part, informed by assumptions made in the modelling approach taken, including that there is no spatial-autocorrelation at scales larger than our 1 km squares, which is supported through the previous use of the Moran's I statistic. A full discussion of the development of this approach to modelling Countryside Survey data for national condition and change and assumptions therein is provided in Scott (2008).

TABLE 2 | Mean estimates, 95% confidence intervals (bootstrapped) and Tukey post hoc test p values for change in topsoil (0–15 cm) organic carbon concentration (g kg^{-1}), organic carbon stock (t ha^{-1}) and bulk density (g cm^{-3}) between repeat visits of the Countryside Survey for all cropland soils and stable cropland soils without any reported change in broad habitat.

Change in organic carbon concentration (g kg^{-1})						
	All cropland			**Stable cropland**		
Change	**Mean**	**95% CI**	***p***	**Mean**	**95% CI**	***p***
1978 to 1998	−1.04	−2.77 to 0.75	0.417	−1.16	−2.76 to 0.37	0.244
1998 to 2007	−3.18	−4.20 to −2.16	< 0.001	−3.00	−4.04 to −1.90	< 0.001
2007 to 2020	+2.10	1.24 to 2.90	< 0.001	+1.86	1.19 to 2.56	< 0.001
1978 to 2020	−2.12	−4.09 to 0.15	0.035	−2.30	−4.03 to −0.55	0.008
Change in organic carbon stock (t ha^{-1})						
	All cropland			**Stable cropland**		
Change	**Mean**	**95% CI**	***p***	**Mean**	**95% CI**	***p***
1978 to 1998	−1.22	−3.06 to 0.72	0.667	−1.05	−2.91 to 0.83	0.801
1998 to 2007	−3.47	−5.67 to −1.26	0.001	−3.32	−5.83 to −0.90	0.004
2007 to 2020	+2.40	0.56 to 4.24	0.002	+2.40	0.74 to 4.20	0.003
1978 to 2020	−2.29	−4.93 to 0.45	0.107	−1.96	−4.52 to 0.65	0.239
Changes in bulk density (g cm^{-3})						
	All cropland			**Stable cropland**		
Change	**Mean**	**95% CI**	***p***	**Mean**	**95% CI**	***p***
2007 to 2020	−0.06	−0.09 to −0.03	< 0.001	−0.04	−0.07 to −0.01	0.025

2.5 | Estimating National SOC Change

Estimates of the impact of the observed topsoil SOC stock change ($t\,ha^{-1}$) were scaled to the area of all national cropland in GB using the area of cropland in the UKCEH Land Cover Map 2020 (Morton et al. 2021).

3 | Results

Significant changes over time have occurred in cropland SOC concentration, SOC stock, and bulk density in GB, with the monitoring cycle found to have a significant effect on all response variables considered, both for all cropland sites and stable cropland sites (all p values < 0.001, Table S4). No spatial autocorrelation was detected in any model for any response variable considered (all p values > 0.05, Table S5). In agreement with previous reporting on cropland topsoil SOC trends from the Countryside Survey in Emmett et al. (2010), a non-significant decrease in SOC is observed from CS1978 to CS1998, followed by a significant decline from CS1998 to CS2007 for both carbon concentration ($g\,kg^{-1}$) and carbon stock ($t\,ha^{-1}$) in both cropland and stable cropland (Table 2, Figure 2). The change in carbon stock from CS1978 to CS2007 is $-4.69\,t\,ha^{-1}$ for cropland and $-4.37\,t\,ha^{-1}$ for stable cropland.

SOC concentration ($g\,kg^{-1}$) and stock ($t\,ha^{-1}$) were found to have significantly increased in British cropland topsoils (0–15 cm) from 2007 to present (Table 2, Figure 2). The increase is significant when considering both all cropland soils and stable cropland (Table 2, Tables S6 and S7). Mean SOC concentrations and stocks are consistently lower for stable cropland than for the full cropland dataset, for each monitoring cycle, by an average of $3.2\,g\,kg^{-1}$ and $2.4\,t\,ha^{-1}$, respectively. For example, mean SOC

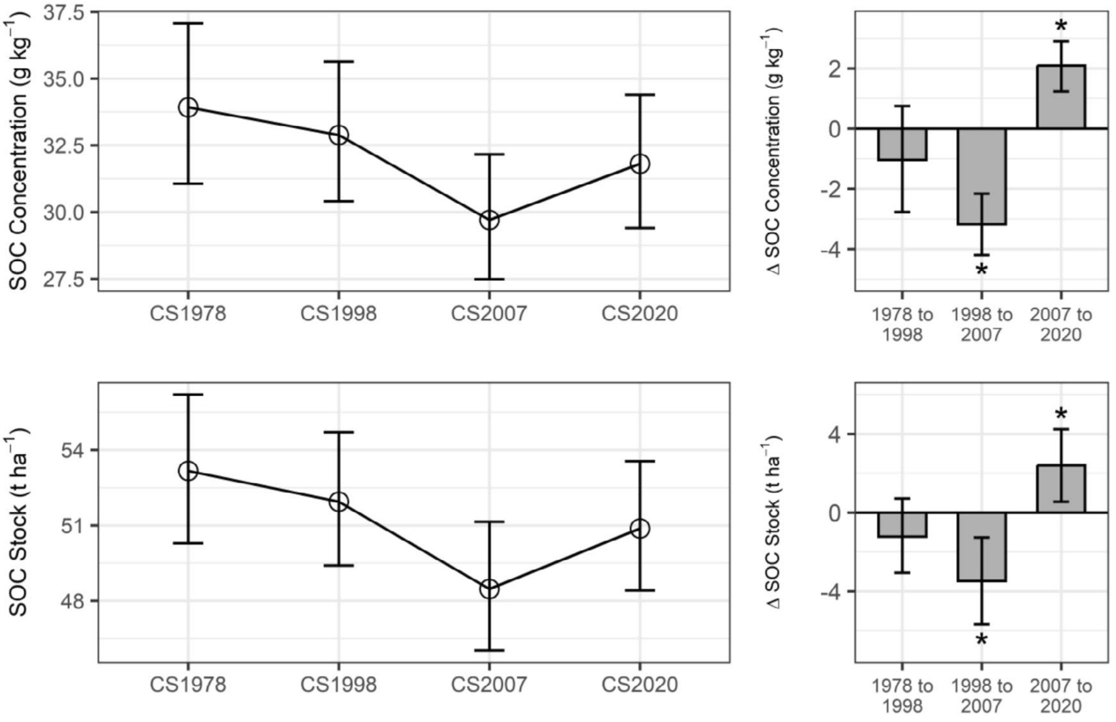

FIGURE 2 | Trends in mean topsoil organic carbon (SOC) concentration ($g\,kg^{-1}$) and stock ($t\,ha^{-1}$) with bootstrapped 95% confidence intervals for the mean in each monitoring cycle for all cropland sites (left) and mean change in topsoil SOC concentration ($g\,kg^{-1}$) and stock ($t\,ha^{-1}$) between monitoring cycles with bootstrapped 95% confidence intervals for all cropland sites (right).

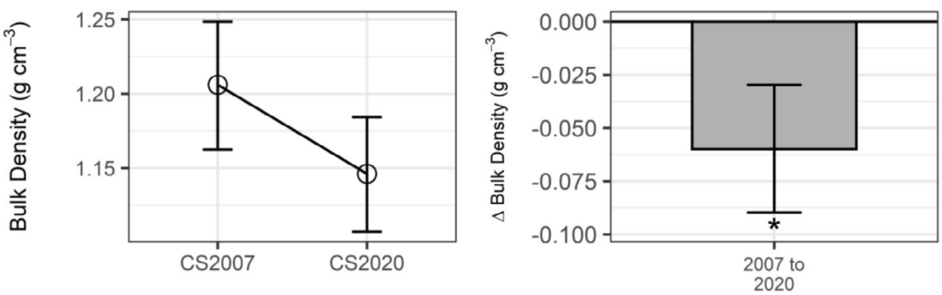

FIGURE 3 | Trends in mean topsoil bulk density ($g\,cm^{-3}$) with bootstrapped 95% confidence intervals for the mean in each monitoring cycle for all cropland sites (left) and mean change in topsoil bulk density ($g\,cm^{-3}$) between monitoring cycles with bootstrapped 95% confidence intervals for all cropland sites (right).

in CS2020 was estimated as 50.9 $t\,ha^{-1}$ across all cropland sites, but 48.6 $t\,ha^{-1}$ across stable cropland sites. This is likely due to the exclusion of sites converted from other land uses with higher baseline SOC (Emmett et al. 2010). At the same time as SOC concentration and stock have increased, bulk density was observed to significantly decrease from CS2007 to CS2020 by 0.06 $g\,cm^{-3}$ across all cropland sites, and by 0.04 $g\,cm^{-3}$ for stable cropland sites. Mean estimates of SOC concentration, SOC stock, and bulk density for each time period are provided in the supplement (Table S8).

Estimated rates of change in SOC concentration and stock from CS2007 to CS2020 (Table 2) correspond to accrual rates of +0.15 $g\,kg^{-1}\,year^{-1}$ and +0.17 $t\,ha^{-1}\,year^{-1}$, respectively. At a national scale, using the area of cropland in Britain as estimated by the UKCEH LCM 2020 (Morton et al. 2021), these findings indicate a potential increase in cropland topsoil SOC stocks of approximately 0.74 $Mt\,year^{-1}$.

4 | Discussion

The observed increase in SOC is a positive sign for the functionality of British cropland soils and marks a stark departure from the significant losses reported in the previous three decades, as identified by multiple national assessments (Bellamy et al. 2005; Emmett et al. 2010). To our knowledge, this is the first evidence that the loss of cropland SOC can be reversed at a nationally significant scale. However, caution should be taken in the interpretation of these results and their implications for cropland in GB, considering the context of the sampling design (e.g., for topsoil only and representing a national average). The annual rate of SOC stock increase we observed from CS2007 to CS2020 was approximately 0.34%, and thus approaches the 0.4% increase per year advocated for by the "4 per 1000" initiative (Lal et al. 2018; Smith 2004), despite being a small change relative to the overall topsoil SOC stock in cropland. The observed increase in SOC contrasts with data for cropland SOC for Europe + UK from the LUCAS database, which reported an EU wide decline in cropland SOC of −0.04 $gC\,kg^{-1}year^{-1}$ from 2009 to 2018 with high rates of SOC loss estimated for Southeast England using an EU + UK wide spatial modelling approach, informed by land use, clay content and climate (De Rosa et al. 2023). This disagreement between our results is likely due to the greater sampling density for GB in CS, whereas the UK is a comparatively small component of the LUCAS monitoring. Additionally, the model used by De Rosa et al. (2023) does not capture the impact of management on SOC trends within a land use, which will vary substantially across the EU and UK. Although Bellamy et al. (2005) previously reported a small significant decline in English and Welsh cropland SOC concentration per year using data collected through the National Soil Inventory, they also report a significant increase in mean carbon concentration for soils with < 30 $g\,kg^{-1}$, although the distribution of these soils by habitat was not described.

Based on the available evidence of changing cropland management practices, we consider that there is a strong possibility that SLM uptake is the driver of topsoil SOC recovery. National estimates for the uptake of these practices are difficult to obtain, but evidence suggests that over 40% of UK farmers now practice some form of reduced tillage, up from 25% in the 1980s (Alskaf et al. 2020), and data from Eurostat (2020) show a shorter-term reduction in conventional tillage in place of conservation and no-till practices from 2010 to 2016. Straw removal on wheat and barley also peaked (as a % of straw removed) in the early 1990s, increasing rapidly from the 1980s and slowly declining thereafter, broadly the inverse of rates of reported SOC loss for the same period (DEFRA 2020). The retention of straw in field will have also increased nationally as a result of the UK shifting from a barley to wheat dominated system over this period (DEFRA 2022), with wheat crops consistently associated with greater rates of straw retention (DEFRA 2020). The sown area of Britain receiving manure is unlikely to be a driver of SOC recovery since this has undergone a small decrease since 2007 (DEFRA 2020). Some additional, well-supported methods for increasing cropland soil carbon content include the use of temporary grassland and switching to the production of perennial crops (Bentley, Feeney, Matthews, Evans, et al. 2023); however, changes to the frequency of temporary grassland cover for the study period are as yet unknown (although going forwards estimates could be derived from remote sensing, Upcott et al. 2023), and the total area of perennial crops in the UK remains low (DEFRA 2022). As we define stable cropland using habitat type during surveyor visits, it is likely some of these sites have included leys as part of their rotational management, and a change in the frequency or duration of leys could be contributing to the reported change in SOC since 2007.

Globally, the maximum technical potential for SLM practices to sequester carbon in cropland has been suggested to range from 0.1 to 1 $t\,ha^{-1}\,year^{-1}$ (Lal et al. 2018), whilst estimates for European cropland include 0.69 $t\,ha^{-1}\,year^{-1}$ for cereal straw inclusion, 0.38 $t\,ha^{-1}\,year^{-1}$ for animal manure application, and 0.38 $t\,ha^{-1}\,year^{-1}$ for zero tillage, but emphasise that biologically and practically achievable rates may be substantially lower (Freibauer et al. 2004; Smith 2004). In the UK, Powlson et al. (2012) found a change from conventional to reduced tillage increased carbon storage by 0.31 $t\,ha^{-1}\,year^{-1}$ for 0–30 cm. Reduced tillage has been associated with increases in bulk density (Soane et al. 2012), which we do not observe (Figure 3), but can also result in reduced bulk density and reduced compaction risk long-term (Bentley, Feeney, Matthews, Evans, et al. 2023; Blanco-Canqui and Ruis 2018; Newell Price et al. 2023). Increased topsoil carbon is likely to result in lower bulk density, in the absence of a change in compaction, given the close relationship between the two variables (Robinson et al. 2022). Land use is well established as the primary driver of variation in topsoil SOC within the UK and the driver of historic topsoil SOC loss (Emmett et al. 2010), and whilst relationships between climate and SOC are well known (Thomas et al. 2020) we know of no step-change in climatic conditions to explain the observed reversal in cropland SOC. The potential for other drivers to be interacting with climate change warrants further exploration.

The observed increase in topsoil SOC is expected to improve topsoil health and functionality as one of the most widely used indicators of soil health (Kopittke et al. 2022). However, in the context of Net Zero, it is of great importance whether this additional topsoil carbon is additional carbon fixed from the atmosphere, or whether it reflects geographic or vertical carbon redistribution

(Baveye et al. 2023; Moinet et al. 2023). Carbon concentration is variable with soil depth and observed trends for increased carbon concentration may not hold when considering the full soil profile. In particular, increased uptake of reduced till practices could be driving a greater stratification of the soil profile and enhancement of topsoil SOC, as typical rotational tillage depth exceeds the sampling depth in this study (20 vs. 15 cm). The limitations of fixed depth sampling for estimation of SOC stocks have been widely discussed, with a risk of compaction elevating measured carbon stocks despite stable carbon concentration (Lee et al. 2009; Wendt and Hauser 2013). However, we see a significant increase in both SOC concentrations and stocks from CS2007 to CS2020 and evidence of reduced compaction during the study period, indicating the change in SOC stocks may have been underestimated for equivalent soil mass. Furthermore, there is evidence for agricultural soils that changes from 0 to 15 cm are indicative of changes to depth and are where the majority of change in SOC stocks occur (Fornara et al. 2016; Guo and Gifford 2002). We know of no source of bias between monitoring cycles that could give rise to the observed SOC increase from CS2007 to CS2020. One of the limitations of this work is the lack of corresponding information on changes in management and the sometimes decade long intervals between monitoring cycles, making the identification of drivers challenging. The results are also subject to uncertainties integral to a long-term national monitoring approach which has many logistical and practical challenges that must be met, as described in the supplement. Going forward, we hope the 5-year rolling monitoring cycle and synthesis with additional data streams (e.g., remote sensing) will address some of these knowledge gaps, with the next CS monitoring cycle planned from 2024 to 2028. Increasing the number of 1 km squares sampled and sampling depth could also provide additional insights into the reported trends. Ultimately, whilst the observed increase in SOC remains a positive sign for cropland soil health restoration, we emphasise that caution is needed in interpreting topsoil SOC trends with regards to Net Zero.

5 | Conclusion

The observed increase in cropland topsoil SOC demonstrates the potential for SOC recovery in cropland soils and the potential for recovery in degraded soils more generally at scale. Recovery is not seen at all sites however, and there are locations where SOC has continued to decrease. Understanding this variation will help reveal "what works where" for sustainable cropland soil management in a UK context and more broadly for temperate systems. National changes to tillage practices and straw removal are identified as recent management shifts with the potential to drive this trend, however substantial evidence gaps remain around other additions to cropland and changes to rotational management. This finding offers a new perspective on the potential for SOC recovery in cropland soils and provides the first hope that SLM could be effective at restoring, to some extent, cropland SOC stocks at national scale.

Author Contributions

L.B., D.A.R., S.M.S. and B.A.E. led the work and L.B., D.A.R., P.A.H., S.M.S. and B.A.E. designed the study. L.B. performed the data analysis. Field team management and data collection coordinated by A.G. and B.W. Laboratory analysis was performed by I.L., M.B. and P.K. Data curation was performed by L.B., S.R., B.J.C. and C.W. L.B., A.T., S.R., P.A.H. and D.A.R. wrote the manuscript.

Conflicts of Interest

The authors declare no conflicts of interest.

Data Availability Statement

All data analysed in this publication is available via the EIDC and can be accessed at the following links. Soil used in this study has been archived at the UKCEH Soil Bank. (Bunce et al. 2016; Black et al. 2016; Emmett et al. 2016; Bentley, Reinsch, Alison, Andrews, et al. 2023; Bentley, Reinsch, Alison, Brentegani, et al. 2023; Bentley, Reinsch, Brentegani, Chetiu, et al. 2023; Reinsch et al. 2023).

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Supporting Information

Additional supporting information can be found online in the Supporting Information section.