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Review of best practice for SOC monitoring

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Contents

1 Introduction

- The establishment of both baseline and target values for soil quality have been proposed to quantify the impacts of future land management schemes (Defra 2018). Soil quality indicators (SQIs) are often proposed to assess the delivery of soil ecosystem services, however, despite significant progress with interpreting indicator values, major scientific and practical issues remain to be addressed.
- When soil organic carbon-(SOC) stocks are reduced, it is typically coincident with other forms of soil degradation (e.g. topsoil loss or compaction, Amundson *et al*., 2015) and it is clear that policies that maintain or enhance SOC should result in multiple benefits in terms of climate change mitigation and the sustainability of agricultural systems.
- In order to reward management practices that maintain or enhance SOC it is essential that any chosen metrics should firstly be able to establish baseline levels and secondly monitor either increases or decreases in the baseline.
- Two national soil monitoring programmes (the National Soil Inventory (NSI) and the Countryside Survey (CS)) have reported conflicting evidence with regard to changes in topsoil carbon in England and Wales. The NSI reported losses in soil carbon from 1978 to 2007 whereas the CS reported no change in soil carbon. More recent soil samples taken for the Glastir Monitoring and Evaluation Programme (GMEP), between 2012 and 2016 also indicated no change in topsoil carbon for Wales (Emmett *et al*., 2017). This discrepancy in SOC levels reported by the two monitoring programmes makes it difficult to confidently set a baseline for SOC.
- A review by Kirk *et al*. (2011), eliminated several potential reasons for the differences between the samplings (i.e. statistical effects, sampling or analysis methodology, sampling design or site re-location) but found no definitive reason for the difference. This report has been commissioned to undertake a short review of the report by Kirk *et al*. (2011) and the national soil monitoring programmes. More widely it will assess potential methodology for measuring SOC (e.g. sampling programme or sample depth) at a national scale and the associated establishment of an accurate baseline for SOC.

1.1 Objectives

- The overall objective of this project is to review Defra project SP1101 (Kirk *et al.*, 2011) and the associated SOC data from previous national soil monitoring programmes (i.e. the National Soil Inventory and the Countryside Survey). The review suggests approaches for future potential SOC monitoring schemes in Wales. In detail the report has:
    - Reviewed Defra project SP1101 and associated SOC data from national monitoring programmes,
    - Assessed the methodologies for measuring SOC including sampling design, accuracy, repeatability and costs.
    - Established the evidence to determine whether changes in soil carbon can be accurately assessed at a national, regional and local scale and the certainty/uncertainty associated with this metric.
    - Considered the requirements for a monitoring framework for quantifying changes in SOC at a national scale.

2 National soil monitoring programmes

- In recent years two national soil monitoring programmes (National Soil Inventory-NSI and Countryside Survey-CS) have reported on changes in topsoil SOC for England and Wales, the first in 2005 (Bellamy *et al*., 2005) and the second in 2010 (Emmett *et al*., 2010). Both reports covered changes in SOC over a similar time period, i.e. from the late 1970s/early 1980s to the early/mid-2000s. However, the main findings of the two reports disagreed; the NSI suggested a net loss of 4 million tonnes (t) SOC/year whereas the CS showed only negligible changes in SOC over the period 1978-2007. The reasons for the difference between the two programmes remains unclear despite additional reviewing of the data by Kirk *et al*. (2011). Details of sampling strategies, depths etc. for the two monitoring programmes are given in Table 3 and discussed in more detail below.
- For Scotland, the National Soil Inventory of Scotland (NSIS) sampled 721 soils (sample stratification was based on a 10 km grid) between 1978 and 1987; approximately 25% of the original sites (195) were re-sampled between 2007 and 2010 (sample stratification for the later survey was based on a 20 km grid). Soil samples were taken to at least 75 cm (or 100 cm in peats) and carbon was measured by both elemental analyser and LOI and carbon stock was calculated to 100 cm using measured (for the second sample) or predicted (for the first sample) bulk density (Chapman *et al*., 2013). In line with the CS, the survey found no overall change in % SOC (1978-1978: 184 ± 10 g/kg; 2007-2010: 187 ± 10 g/kg) or carbon stock to 100 cm (1978-1978: 266 ± 15 t/ha; 2007-2010: 270 ± 15 t/ha) between the first and second sampling. The exception was for soils under woodland (excluding those on deep peat where organic material extended beyond 100 cm) which had a significant gain of 1.0 t/C/ha/year. For comparison with the NSI and CS, the authors also calculated the change in carbon stock over time for the 0-15 cm depth (1978-1978: 71 ±2 t/ha; 2007-2010: 74 ± 3 t/ha), which also suggested no change in carbon stock, although when deep peat sites were excluded a significant increase in overall C stock of 4.3 t C/ha was noted.

2.1 National Soil Inventory

- Bellamy *et al*. (2005) used data from the NSI obtained between 1978 and 2003 to assess changes in SOC over time. In brief, SOC was measured at *c.*5,550 sites between 1978 and 1983 and again in either 1994 (arable and rotational grass), 1995 (permanent pasture) or 2003 (non-agricultural sites, e.g. bogs, woodland etc.) to measure change. For the majority of the samples SOC was measured using the modified Walkley Black (MWB) method, with the exception of samples from non-agricultural soils with SOC >approximately 150 g/kg which were measured using Loss on Ignition. It is not clear how high SOC samples were identified. To calculate change over time, it was assumed that the process of change was linear over the sampling period. An assumed bulk density was also used to calculate the change in SOC stocks.
- Bellamy *et al*. (2005) estimated that the total SOC across England and Wales at the time of the first sampling in 1978 was 864 Teragrams-Tg (0-15 cm) and that the total rate of change was -4.44 Tg/year. This would equate to a total loss of *c*.80 Tg between the two sampling periods (assuming a loss of 4.44 Tg per year over 18 year) although the variable number of years between sampling periods make a precise estimate of change (from the published data) difficult. They calculated that the relative rate of carbon loss increased with soil carbon content and was more than 2% per year in soils in England and Wales with carbon contents greater than 100 g/kg (10% SOC). In comparison, Chapman et al. (2013) reported that they were 95% confident that the mean loss for Scotland did not exceed 0.2%/year and 99% confident that it did not exceed 0.4% per year.

Table 1. SOC concentration (g/kg) and stock (0-15 cm Tg) at initial sampling (1978-1983) and estimated rates of annual change (Tg/yr) up to second sampling (1994-2003) (Bellamy *et al*. 2005).

Original SOC (g/kg)	**0-20**	**20-30**	**30-50**	**50-100**	**100-200**	**200-300**	**>300**
Mean original SOC (g/kg)	13.9	24.5	38.5	66.8	137.6	244.5	439.7
Rate of change (g/kg/year)	0.34	0.13	-0.10	-0.68	-2.18	-4.00	-7.37
Relative rate of change (%/yr)	1.43	0.14	-0.69	-1.84	-2.71	-3.01	-2.35
Total SOC (0-15 cm) (Tg)*	66.4	111.9	214.8	220.5	92.5	36.5	121.7
Total change (0-15 cm) (Tg/yr)**	1.68	0.66	-0.65	-2.11	-1.32	-0.59	-2.10

*Original SOC x bulk density (kg dm^{-3}) x depth (m) x area (km^2) x 10^{-4} where bulk density = 1.3 – (0.275 ln(SOC/10)). **Total change = Rate of change x bulk density x area

2.2 *Countryside Survey*

- Emmett *et al*. (2010) used data from the Countryside Survey to assess changes in SOC concentration from 1978 and 1998 and from 1998 to 2007. In brief, soil samples were collected from randomly selected 1 km x 1 km grid squares within 32 (or 45 from 1998) land classes (derived from a statistical analysis of 40 environmental variable including climate, soils, topography and geology) in 1978, 1998 and 2007 (Table 2).
- SOC was measured by loss on ignition (LOI) on each occasion. Alongside LOI measurements, total soil C concentration was determined by elemental analyser in 1998 and 2007 but not in 1978. The elemental analyser (Elementar Vario-EL Elementaranalysensysteme GmbH, Hanau, Germany); worked on the principle of oxidative combustion followed by thermal conductivity detection. Following combustion in the presence of excess oxygen the oxides of C and N (which was analysed at the same time) flowed through a reduction column which removed excess oxygen. C was then trapped on a column whilst N was carried to a detector. C was then released from the trap and detected separately as % dry weight of soil.
- A conversion factor of 0.55 x LOI was used to convert the LOI value to SOC based on the results from the elemental analyser. For the 2007 samples, carbon stock was calculated using a co-located bulk density measurement whilst for the 1978 and 1998 samples a bulk density transfer function of $1.29e^{-0.0206x} + 2.51e^{-0.0003x} -2.057$ (calculated from bulk density samples taken in 2007) was combined with LOI/soil concentration values to estimate topsoil C stock (0-15 cm) on an area basis.
- Emmett *et al*. (2010) calculated an increase in SOC from 1978-1998 and a decrease from 1998-2007 resulting in no overall change in SOC (0-15 cm) between 1978 and 2007 (Figure 1). A range of direct and indirect explanatory variables (e.g. soil pH or soil moisture) were explored to explain the spatial and temporal variability in SOC but the results were inconclusive. However, for the arable and horticulture broad habitat/crops and weeds vegetation class there was a consistent loss of SOC (Figure 2, Figure 3, Figure 4). Due to the concomitant reduction in total soil N, Emmett *et al.* (2010) suggested that the reduction in SOC concentration/stock might be due to erosion, deep ploughing or increased decomposition.
- For Wales, no significant changes in SOC concentration was noted between 1998 to 2007; data was only reported from two vegetation classes due to the low number of samples collected from Wales in 1978 (Figure 5).
- Overall, the Countryside Survey estimated the SOC stock of Great Britain in 2007 as 1582 Tg (England: 795 Tg, Scotland: 628 Tg and Wales 159 Tg). For comparison, Bellamy *et al.* (2005) calculated 864 Tg for England and Wales (for 1978), compared to 954 Tg in the Countryside Survey (949 Tg in 1978).

Table 2. The 32 ITE land classes in 1978, 2000 and 2007 (Source: Wood, 2011)

No			
1978	***1998****	***2007***	***Brief description***
1	1	1e	Undulating country, varied agriculture, mainly grassland.
2	2	2e	Open, gentle slopes, often lowland, varied agriculture.
3	3	3e	Flat arable land, mainly cereals, little native vegetation.
4	4	4e	Flat, intensive agriculture, otherwise mainly built-up.
5	5	5e, 5w	Lowland, somewhat enclosed land, varied agriculture and vegetation.
6	6	6e, 6w	Gently rolling enclosed country, mainly fertile pastures.
7	7/7	7e, 7s, 7w	Coastal with variable morphology and vegetation.
8	8	8e	Coastal, often estuarine, mainly pasture, otherwise built-up.
9	9	9e	Fairly flat, open intensive agriculture, often built up.
10	10	10e	Flat plains with intensive farming, often arable/grass mixtures.
11	11	11e	Rich alluvial plains, mainly open with arable or pasture.
12	12	12e	Very fertile coastal plains with very productive crops.
13	13, 13	13e, 13s	Somewhat variable land forms, mainly flat, heterogeneous land use.
14			Level coastal plains with arable, otherwise often urbanised.
15	15	15e, 15w	Valley bottoms with mixed agriculture, predominantly pastoral.
16	16	16e	Undulating lowlands, variable agriculture and native vegetation.
17	17e, 17w1, 17w2, 17w3	17e, 17w1, 17w2, 17w3	Rounded intermediate slopes, mainly improvable permanent pasture.
18	18, 18	18e, 18s, 18w	Rounded hills, some steep slopes, varied moorlands.
19	19, 19	19e, 19s	Smooth hills, mainly heather moors, often afforested.
20			Mid-valley slopes, wide range of vegetation types.
21	21	21s	Upper valley slopes, mainly covered with bogs.
22	22, 22	22e, 22s	Margins of high mountains, moorlands, often afforested.
23	23, 23	23e, 23s	High mountain summits, with well drained moorlands.
24	24	24s	Upper, steep, mountain slopes, usually bog covered.
25	25, 25	25e, 25s	Lowlands with variable land use, mainly arable.
26	26	26s	Fertile lowlands with intensive agriculture.
27	27	27s	Fertile lowland margins with mixed agriculture.
28	28	28s	Varied lowland margins with heterogeneous land use.
29	29	29s	Sheltered coasts with varied land use, often crofting.
30	30	30s	Open coasts with low hills dominated by bogs.
31	31	31s	Cold exposed coasts with variable land use and crofting.
32	32	32s	Bleak undulating surfaces mainly covered with bogs.

*To enable reporting at a country level classes were subsequently split into country units, e.g. 18e, 18s and 18w indicate the presence of land class 18 in England, Scotland and Wales, respectively.

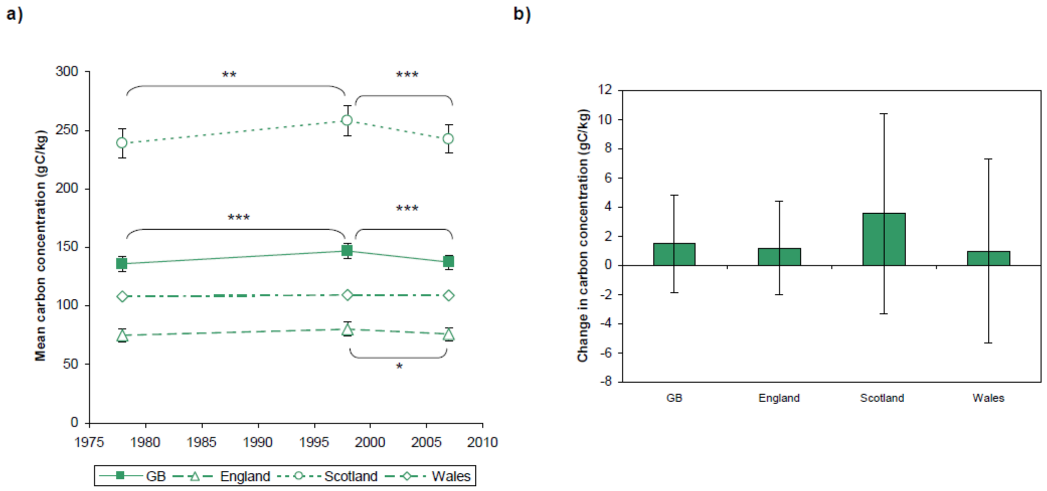

Figure 1. Change in SOC concentration (g/kg 0-15 cm) for Great Britain (GB) and individual countries a) over time and b) net change between 1978 and 2007. Significant differences: ****P*<0.001, ***P*<0.01 and **P*<0.05 (Source: Emmett *et al.*, 2010).

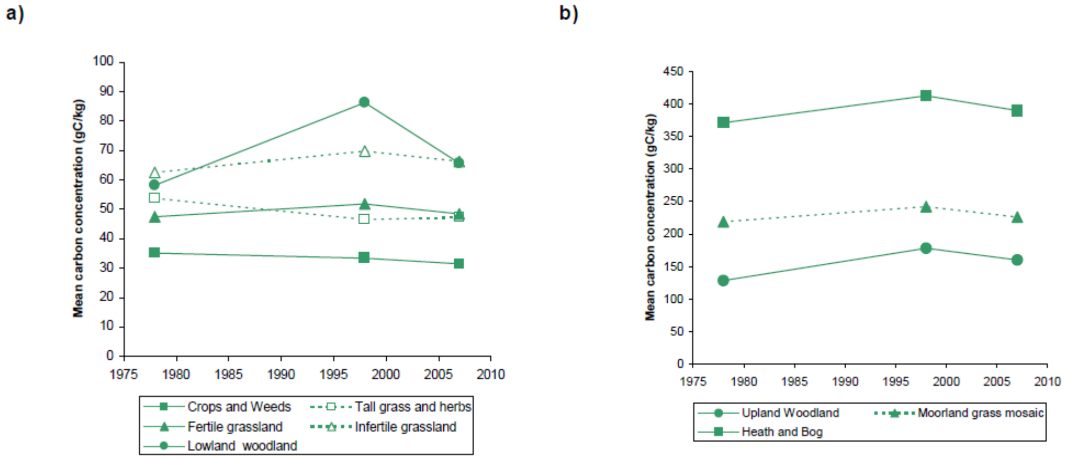

Figure 2. Change in SOC concentration (g/kg 0-15 cm) over time across land use types a) low-medium topsoil C concentrations and b) high topsoil C concentrations (Source: Emmett *et al.*, 2010).

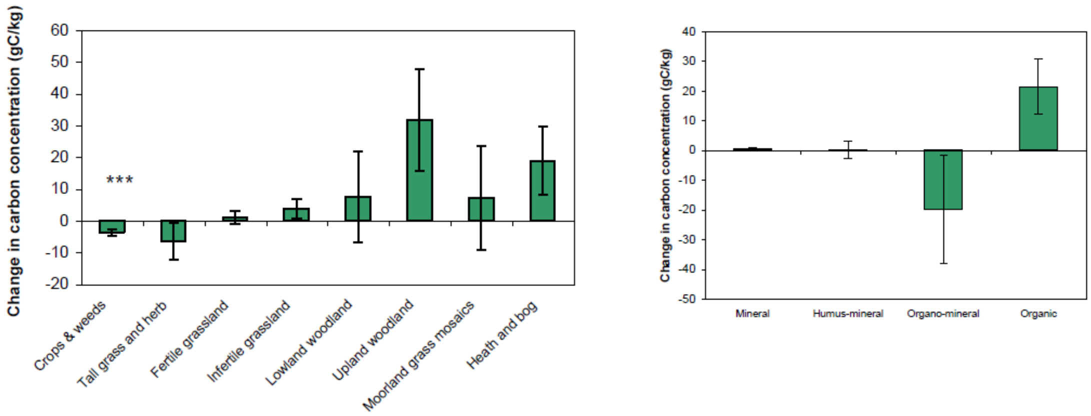

Figure 3. Overall change in SOC (g/kg 0-15 cm) between 1978 and 2007 by a) land use type and b) soil type. Significant differences: ****P*<0.001 (Source: Emmett *et al.*, 2010).

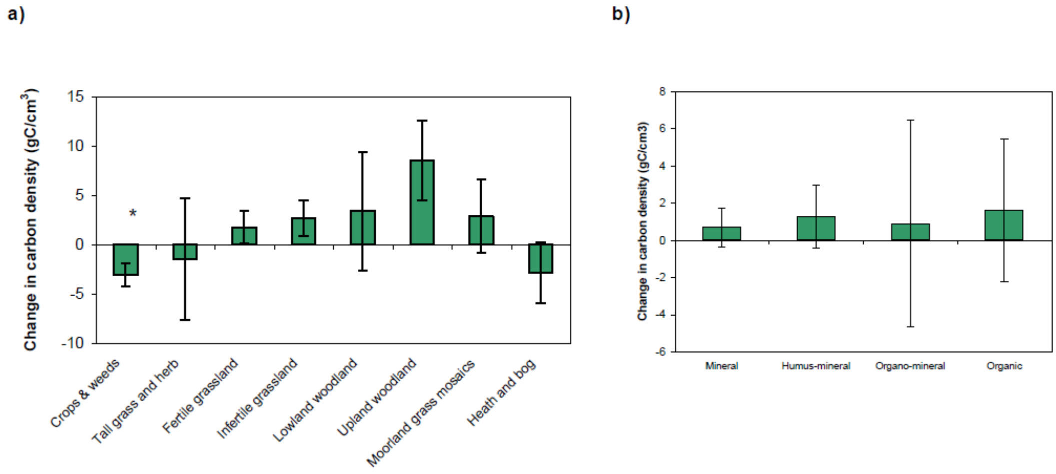

Figure 4. Overall change in SOC density (g/cm^3 0-15 cm) between 1978 and 2007 by a) land use type and b) soil type. Significant differences: **P*<0.05. (Source: Emmett *et al.*, 2010).

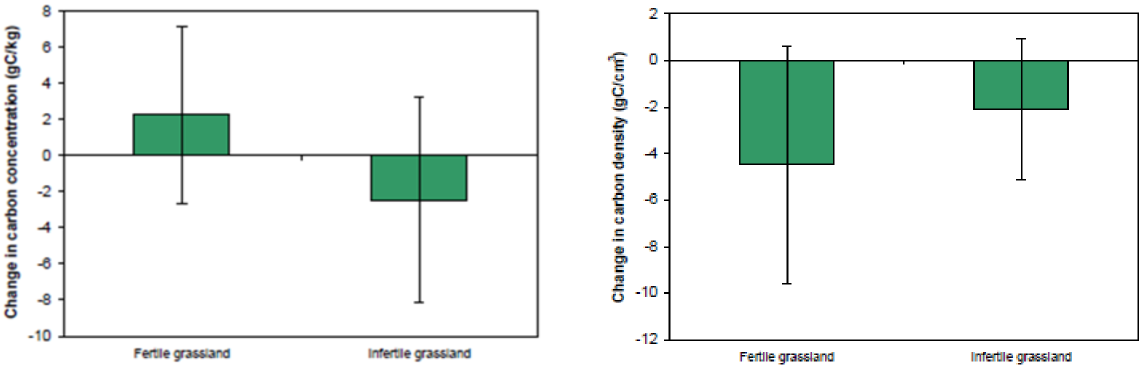

Figure 5. Change in a) SOC concentration (g/kg 0-15 cm) and b) SOC (g/cm^3 0-15 cm) content for Wales from 1978-2007.

Table 3. Comparison of methodology for National Soil Inventory and Countryside Survey soil organic carbon (SOC) monitoring[1]

	National Soil Inventory	**Countryside Survey**
Site location	Intersection of 5 km grid The NSI was designed to be an unbiased sample of agricultural land. The geographical distribution of sites to be resampled was based on the proportion of the original NSI sites under each land use in each MAFF region. Statistical analysis was used to determine the number of sites that should be resampled to allow small changes in SOC to be measured. For example in Wales, 7% of re-sampled sites were arable, 23% grass ley and 70% permanent grassland.	Stratified random sample of 1 km squares based on the Institute of Terrestrial Ecology Land Classification. Sample squares were random representation of 32 (or 45 from 1990) land classes. Soils were sampled from the CS X plots of which there were 5 randomly spaced per 1 km square; they are not replicates as they may be in different land uses.
Site number	6127 sites. 5589 topsoil samples	
First sampling	1978-1983	1978 – 256 sites/*c*.1200 soil samples (32 land classes x 8 per class x 5 plots)
Second	*1994-1995* Arable/ley - 842 sites *1995-1996* Permanent grass - 745 sites *2003* Other land uses (e.g. bogs, woodland etc.) – 562 sites For arable/ley and permanent grass land uses, resampling, numbers were calculated based on the number of samples required to detect a 2 g/kg change in SOC. For other land uses sampling was designed to detect a 2 g/kg change for topsoils with OC <150 g/kg, a 5 g/kg change where OC was 150-300 g/kg and a 10 g/kg change where OC >300 g/kg.	*1998.* Soils taken 1.15 m to the north of the centre of the X plot. As for 1978, one soil sample was taken from each X plot giving a total of *c*.1,200 samples.
Third		*2007* – 591 sites (2,614 soil samples) Soils also taken from plots not previously sampled.
Sample depth/number	*0-15 cm.* At least 25 cores at the nodes of a 4 m grid within a 20 m x 20 m square centred on the OS 5 km grid point. Screw-type mild-steel auger (2.5 cm diameter).	*0-15 cm* Each square was divided into 5 segments of approximately equal area. An 'X' plot (14.4m x 14.4m or 200m^2) was

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[1] The implications (if any) for SOC stock calculations of variations in methodology are discussed in more detail in subsequent sections of the report.

	Cores bulked in the field to give a sample of c.1 kg of moist soil and double bagged in polythene bags	located using random co-ordinates within each segment of the square. Soil was sampled from all 5 X plots. *1978.* One soil sample taken from a pit in the centre of each X plot with a trowel after removing fresh plant litter on the soil surface[2]. *1998 and 2007* Samples were collected using a 15 cm long by 5 cm diameter plastic corer. Surface vegetation was removed. One core was used to measure SOC from each X plot, i.e. 5 per site (but note that these were not replicates).
Re-locating sample sites	10 sites were re-visited by 3 surveyors (using the grid reference and field notes to relocate the site) who recorded the site location using GPS; comparisons showed that accuracy of re-location was ≤20 m on enclosed land and ≤50 m on open land. SOC measurements made at 0, 10 and 50 m from the original sample were not significantly different from each other (3 replicates) but intra sample variability was large at some sites (i.e. although the mean was similar the range of values recorded at each site was sometimes large).	1990: permanent metal markers were placed adjacent to the south corner of the X plots. Where this was not possible the plate was inserted into the nearest field boundary along a cardinal bearing line.
Sample treatment	Refrigerated at 4°C on the day of sampling and transported to the lab within 2 weeks. At the lab samples were air dried before milling to pass 2 mm aperture sieve. Half of the sample was retained before milling.	The core was placed in a labelled plastic bag which was sealed and stored at ambient temperature before returning to CEH where they were stored at 4°C until analysis. 1978. Soils were air dried at room temperature and then sieved using a 2 mm aperture sieve. 1998. Soils weighed before they were air dried at room temperature and then sieved using a 2 mm aperture sieve. 2007. Core length measured before they were air dried at room temperature and then sieved using a 2 mm aperture sieve

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[2] Kirk *et al*. (2011) report that the sampling procedure used in 1978 followed the methodology in the 1971 Woodlands Survey rather than in the 1978 CS Handbook.

Organic carbon	1978-1983 samples: Modified Walkley-Black (MWB) method 1994-2003 samples: WB method except for non-agricultural soils when those with OC >150 g/kg were analysed by loss on ignition (LOI). (The small sample size used for MWB is unsuitable for organic soils). LOI values were converted to OC by OC=0.5 x LOI. In 2004, to check for differences in methods, earlier non-agricultural soils with OC >150 g/kg were reanalysed by WB. It was noted that estimates by LOI were lower. Kirk *et al*. reanalysed samples from the second sampling period (1994-2003) that were originally measured by LOI using MWB. This resulted in a dataset where data from both samples was measured by MWB.	SOC measured by LOI. In 1978 and 2007, sample size was 10 g of oven dried soil at 375°C for 16 hours. In 1998, sample size was 1 g of oven dried soil at 550°C for ≥2 hours. 74% of the samples were subsequently reanalysed using the same method as used in 1978 and 2007. The LOI values for the 1998 soils that could not be re-analysed were corrected using a regression equation derived from the samples that were analysed using both methods. Total soil C also determined by elemental analysis (EA) in 1998 and 2007; LOI and EA results showed that 55% of SOM was accounted for by C. LOI values were converted to OC by OC=0.55 x LOI x 10.
Quality control	Based on use of standards, certified reference materials and inter-laboratory validation schemes.	Defra/NERC joint Codes of Practice were followed. Also, used reference samples and repeated one sample per batch.

2.3 Defra SP1101

- The project (SP1101: Comparison of topsoil carbon changes across England and Wales estimated in the Countryside Survey and the National Soil Inventory) was commissioned by Defra to compare the findings of the NSI and CS surveys in 2010 and reported in 2011. It was undertaken by staff from Cranfield University and the Centre for Ecology and Hydrology, the two organisations that were responsible for the NSI and the CS, respectively. The review focused on three key areas
    - A comparison of survey design, sampling, analytical methods and data analysis
    - Repeat analysis of a limited number of samples by the modified Walkley Black method to ensure consistent methodology across time periods.
    - Additional statistical analyses of the NSI and CS datasets.

2.3.1 Survey design comparison

- The survey design for NSI and CS are compared in Table 3. Both surveys began in 1978 and repeated samples at the same location on subsequent occasions in 1994-2003 for NSI and in 1998 and 2007 for CS. Similarly, both have analysed SOC from soil samples taken from 0-15 cm and for both surveys soil was dried and sieved prior to laboratory analysis. In contrast, the methodology used to measure SOC differed between surveys, NSI mainly used the modified Walkley Black-MWB method (except for non-agricultural soils with OC >150 g/kg which were analysed by LOI), whereas CS used LOI. In addition, for soils sampled by LOI a different multiplier was used to convert from LOI (which measures SOM) to SOC. Likewise, the pedotransfer function, used in the calculation of SOC stocks was also different (i.e. NSI: $y = 1.3 - 0.275\ln(SOC/10)$, CS: $y = 1.29e^{-0.0206x} + 2.51e^{-0.0003x} - 2.027$).
- Emmett *et al*. (2010) noted that differences in the SOM to SOC conversion factors and the two BD equations have implications for the estimates of topsoil C stock. The CS estimate of 949 Tg for the total topsoil C stock of England and Wales in 1978 is based on a SOM to SOC conversion factor of 0.55. In contrast the NSI estimate is based on a conversion factor of 0.5. If the NSI conversion factor of 0.5 was applied to the CS data the total topsoil C stock of England and Wales in 1978 would be 863 which is very similar to the 864 Tg estimated by Bellamy *et al*. (2005).
- Moreover, the different BD equations produce estimates of C stock at specific C concentrations that vary considerably; Bellamy *et al*. (2005) estimated that soils in England and Wales which contained >300 g C/kg in 1978 had a total topsoil (0-15 cm) C stock of 122 Tg (Table 1). Using the CS equation to estimate topsoil C stock for the same soils would reduce this value to *c*.73 Tg, and on the same basis would reduce the estimated loss of C from these soils from 2.1 to *c*.1.2 Tg/yr. However, these results do not explain the different trends in topsoil concentration and stock noted in the two surveys.
- Soil texture has been identified as a factor controlling SOC. Particle size distribution data (i.e. % clay, silt and sand) is available for the NSI however, CS samples were hand textured and allocated to broad textural classes (e.g. sand or clay). This difference in the methodology used to determine soil texture makes a comparison between the two surveys (based on soil textural classes) difficult (see SPEP2018-19/12 ALC Technical Review Part 1, Section 7.3. ‘Comparison of hand texturing with laboratory determination’ (Rollett and Williams, 2019) for further information.

2.3.2 *Repeat analyses*

- As noted above, the non-agricultural NSI samples with OC >150 kg were analysed by LOI rather than MWB. This was done because it was considered that LOI gave more accurate results for highly organic soils. However, this change of method introduces some uncertainty in the comparison between the two samplings. As a result, Kirk *et al.* (2011), reanalysed 100 of the 129 samples originally analysed by LOI by MWB (29 samples were either missing or had insufficient sample remaining for reanalysis). A correlation (coefficient 0.963) comparing the original LOI results with the new MWB values showed that the mean difference between the methods was 7.69 g/kg; on average, SOC was higher when measured by MWB than by LOI. Kirk *et al*. (2011) noted that this was a small but statistically significant bias introduced by the change of laboratory method. To check that SOC in the archived samples had not changed over time a number of samples that had been originally analysed by MWB were reanalysed using the same method and the results compared. The correlation (coefficient 0.985) showed that there was a difference of -2.40 g/kg (original minus reanalysed), indicating only a small change as a result of storage or variable implementation of the MWB method.

2.3.3 *Additional statistical analyses*

- In order to further examine the differences between the two surveys Kirk *et al*. (2011) undertook some additional statistical analyses of the two datasets. The analyses made a number of adjustments to the data, i.e.
    - Weighting of individual CS samples (for example, where CS land classes had more samples than would be expected for their area, these were given lower weights) to allow comparison with the NSI.
    - To facilitate comparison the NSI data were normalised to the CS intervals and sampling years by assuming changes were linear.
    - The new laboratory results for NSI samples with OC > approx. 150 g/kg made by MWB, were used in place of the values originally obtained by LOI.
    - Samples were allocated to broad land use types based on the categories defined for the NSI: i.e. arable and rotational grass; permanent grass; woodlands/forest; and everything else.
    - For each survey, the distributions of OC content were assessed across the following classes: 0–20, 20–30, 30–50, 50–100, 100–300, >300 g/kg.

2.3.4 *Results of additional statistical comparisons*

- There was no significant difference between the mean OC content determined by the 2 surveys in 1978. However, the OC contents reported by the two surveys in 1998 were different. Annual rates of change for 1978-1998 were calculated as -0.68 g/kg/year for NSI compared to 0.24 g/kg/year for CS. For CS, the annual rate of change for 1998-2007 was -0.40 g/kg/year; in comparison, the annual rate of change from 1978-2007 was 0.04 g/kg/year (Table 4a).
- Although the mean OC levels for 1978 are similar in the two surveys, the OC distributions from which these means are obtained were not the same. The NSI original survey had a greater proportion of the population in the lowest OC class (0-20 g/kg class: CS 10% of sample; NSI 19% of sample) and fewer in the intermediate classes than the CS population for 1978 (Table 4b). When the data was grouped according to the OC classes in Table 4 there were significant differences between surveys for the rate of change in SOC in the low and intermediate classes; differences in the highest classes were large but not significant (Figure 6).

- The results of new analyses by Kirk *et al*. (2011) showed that the mean changes in OC between 1978 and 1998 by land use differed substantially between the two datasets (Table 4c). For example, for arable and rotational grass (AR/LE), NSI reported a mean change of -7.9 ± 2.7 g/kg between 1978 and 1998, in comparison, CS reported a mean change of -3.7 ± 4.8 g/kg.

Table 4. Comparison between the estimated[*] Countryside Survey (CS) and National Soils Inventory (NSI) for a) mean SOC (± 95% confidence interval) and change in SOC over time, b) proportion of samples in each SOC class and c) mean (± 95% confidence interval) change in SOC for broad land use types

a)	**Organic C (g/kg)**			**Change in OC (g/kg/year)**		
	1978	*1998*	*2007*	*1978-1998*	*1998-2007*	*1978-2007*
CS	79.2 ± 10.8	83.9 ± 10.6	80.3 ± 9.6	0.24 ± 0.26	-0.40 ± 0.54	0.04 ± 0.20
NSI	70.0 ± 4.6	56.3 ± 3.4		-0.68 ± 0.12		

b)	**CS 1978**		**NSI 1978**		**Difference**
OC class (g/kg)	*n*	*%*	*n*	*%*	
0-20	73	10.4	1061	18.7	+8.3
20-30	150	21.4	1158	20.5	0.9
30-50	238	34.0	1607	28.4	-5.6
50-100	147	21.0	1140	20.1	-0.9
100-300	63	9.0	313	5.5	--3.5
>300	30	4.3	383	6.8	+2.4
Total	701	100	5662	100	

c)	**CS**		**NSI**	
	N	*Mean change (g/kg)*	*N*	*Mean change (g/kg)*
Arable/ley grass	82	-3.7 ± 4.8	842	-7.9 ± 2.7
Permanent grass	197	+7.3 ± 4.9	745	-10.8 ± 3.1
Wood	25	+56.7 ± 44.9	204	-10.4 ± 9.7
Other	35	+12.9 ± 72.4	358	-36.7 ± 10.7
Mixed[**]	92	-7.1 ± 13.2		

[*] Tabulated values are estimated values based on additional analyses undertaken by Kirk *et al.*, 2011

[**] Land use changed between samplings

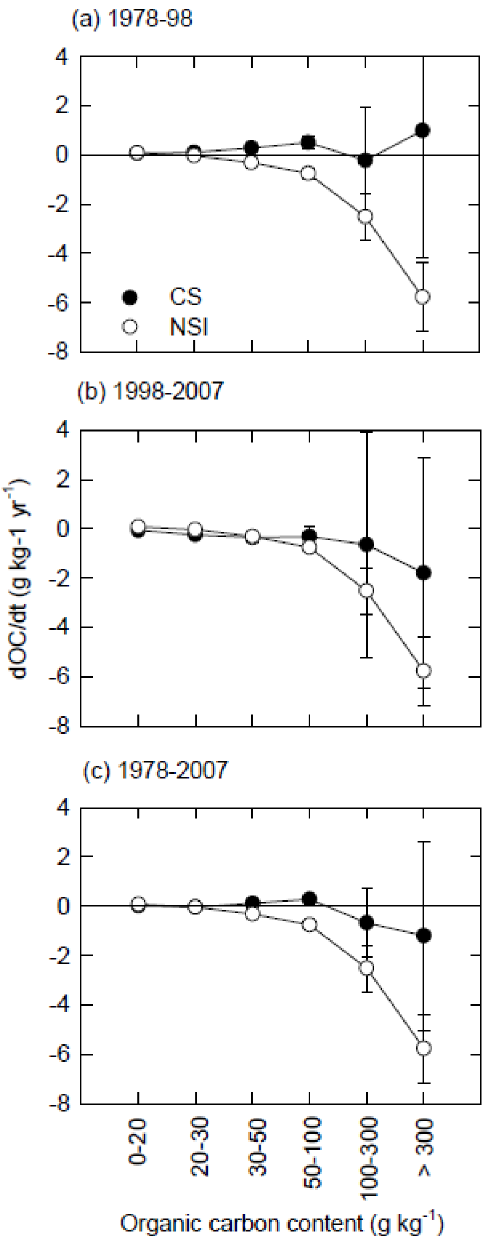


Figure 6. Changes in OC between sampling dates for CS and NSI grouped by OC classes. Means and confidence intervals derived from weighted data.

2.3.5 Conclusions

- Kirk *et al*. (2011) concluded that:
    - The change in the laboratory method for OC used in the NSI was not responsible for the observed large decreases in topsoil C between the first and second sampling. Repeated analysis of archived samples from the first sampling showed that the change in methodology, had a small effect on measured SOC, but that this was too small to explain the observed changes in SOC between the first and second surveys.
    - The different sampling methods (CS: 1 core from each of the 5 X plots; NSI: 25 cores bulked into 1 sample) would not cause the observed changes and were unlikely to introduce any consistent bias at the national scale. Similarly, relocation errors were also unlikely to introduce any bias in the estimates of change at a national scale.

  - The detection of a proportional change in SOC is much more difficult when the initial SOC level is low. The two surveys vary in the distribution of sites into SOC classes (proportionally more sites with low initial SOC in NSRI) and this complicates the interpretation of the changes in SOC stock measured.
  - Whilst some of the potential explanations for the differences between the two surveys have been ruled out no factors that conclusively explained the difference were identified.

3 Methods for measuring soil organic matter and soil organic carbon

3.1 Soil organic matter v soil organic carbon

- Soil organic matter (SOM) and soil organic carbon (SOC) are often confused and mistakenly used interchangeably; further soil carbon (i.e. organic and inorganic carbon) is also erroneously used on some occasions. However, while soil organic carbon is a component of organic matter it is not the same as organic matter, which also includes other elements such as hydrogen, oxygen and nitrogen as well as fresh (living) and decomposed plant/animal (dead) materials (FAO, 2019). The 'living' part includes plant roots and micro-organisms and the larger 'dead' part is root and leaf litter, a light fraction (water soluble organic compounds and soil enzymes) and humus (the largest constituent of the 'dead' part).
- The soil carbon pool is composed of both organic (materials derived from the decomposition of plants and animals) and inorganic carbon. The most common inorganic carbon forms are carbonates from geological or parent material sources.
- Precise determination of changes in SOC stocks is prerequisite to understanding the role of soils in the global cycling of carbon and to verify changes in stocks due to management (Schrumpf *et al*., 2011). Hence, although estimates of soil carbon based on converted estimates of organic matter can be useful, if the purpose of a study is to measure soil carbon, then carbon should be determined directly rather than estimated from a measurement of what is assumed to be organic matter.

3.2 Loss on ignition

- Loss-on-Ignition (LOI) is a quick and relatively inexpensive method used to estimate OM directly but also seems to generate the most controversy (Pribyl, 2010). Some authors recommend its use whilst others are critical (e.g. De Vos *et al*., 2005). The LOI method gives an estimate of SOM content but does not give direct information on SOC content, which is a proportion of SOM that typically ranges between 43 and 58% (FAO, 2019). An important assumption is that LOI is due only to combustion of SOM and that the content of SOC in SOM is constant (Christensen and Malmros, 1982 cited by Pribyl, 2010).
- The fundamental principle is simple: SOM content is the difference between the soil mass before and after ignition at high temperatures (typically 350-500°C). The accuracy of that assumption depends on the time and temperature of ignition, and on the composition of the sample. The temperature used with LOI must be high enough to completely remove organic matter but low enough to prevent dehydroxylation (the addition of hydrogen to a hydroxyl group creating water and an alkyl) of clay minerals and oxidation of carbonates (Wang *et al*, 2012). As the amount of organic matter in a sample increases, complete removal of organic matter becomes more difficult, and as the amount of organic matter decreases, dehydroxylation of clays becomes more likely (Pribyl 2010). Generally, if the ignition temperature remains below about 450°C, adequate removal of organic matter with little dehydroxylation or loss of carbonates would be expected (Nelson and Sommers, 1996 cited by

Pribyl 2010). Failure to adequately dry the sample at low temperature before removing the organic matter at high temperature can inflate the estimate of organic matter in the sample since the weight loss associated with removal of hydration water will be assumed to be organic matter.

- No standard protocol exists for LOI analysis, but it is well documented that LOI is affected by ignition temperature, duration of ignition and ignited sample mass (e.g. Salehi *et al.*, 2011; Hoogsteen *et al*., 2015). Analytical errors are also dependent on differences in important soil properties, such as the amount and type of clay and the amount of carbonates and sesquioxides (FAO, 2019). Such differences make the standardisation of the LOI method challenging. For instance, special precautions should be taken when the method is applied to strongly carbonaceous soils (i.e. soils with a high carbonate content) and soils containing free iron.
- The advantages of the LOI method are: it is simple to carry out, does not require reagents and large samples can be analysed. Large samples may reduce analytical error due to a more representative sample mass (FAO, 2019).
- The main disadvantage of LOI is that it does not provide a direct measurement of SOC, also the lack of standard protocol for measuring LOI complicates comparison between different surveys or experiments. As a result the FAO (2019) suggest that other methods should be used in preference where possible. However, LOI remains a widely used method for assessing SOC in agricultural and forest soils, with LOI being converted to SOC either by a fixed conversion factor or by regression analyses.

3.2.1 Conversion of SOM to SOC

- The conventional (Van Bemmelen) conversion factor for SOM to SOC is 0.58, or its reciprocal of 1.724 (SOC to SOM) (Van Bemmelen, 1891) although a 2010 review of the literature suggested that a more appropriate conversion factor should be 2 (or its inverse, 0.5) Pribyl (2010). More recently, Jensen *et al*. (2018) reported conversion factors of 0.45-0.52 based on the relationship between SOC (measured by elemental analyser and LOI) (Figure 7) for three datasets (one from the UK and two from Denmark). There was a strong positive relationship between LOI and SOC (SOC = 0.39 x LOI -0.28, Figure 7a); in general sandy soils were above the regression line and clayey soils below. All of the points were below the line representing the conventional conversion factor of 0.58, meaning that the use of this factor would overestimate the SOC content. For example, for a sample with a LOI of 4 g/100g the conventional conversion factor of 0.58 would predict a SOC content of 2.32 g/100g, in comparison for the same LOI (4 g/100g) the equation in Figure 7 would predict SOC of 1.28 g/100g.
- Further analysis showed that when the conventional conversion factor of 0.58 was used, the overestimation of SOC increased significantly with increasing contents of clay (mineral particles <2 µm) and Fines20 (mineral particles <20 µm (Figure 8 ab). For example, where clay content was 20%, converting LOI to SOC using the 0.58 conversion factor over predicted SOC by *c.*1 g/100g, when clay content was 60% the over prediction was 2.5-3 g/100g. In comparison, the regression model in Figure 7 (SOC = 0.39 x LOI -0.28) also overestimated SOC for soils with clay contents greater than 30% and Fines20 content greater than 50% but underestimated SOC at lower levels of clay (Figure 8 cd). The inclusion of a quadratic clay or Fines20 factor (the variable raised to the power) removed the systematic error (Figure 8 ef) and there was little difference between predicted and measured SOC (i.e. the points on the chart all sit on, or around, 0 so there is no over or under prediction of SOC).

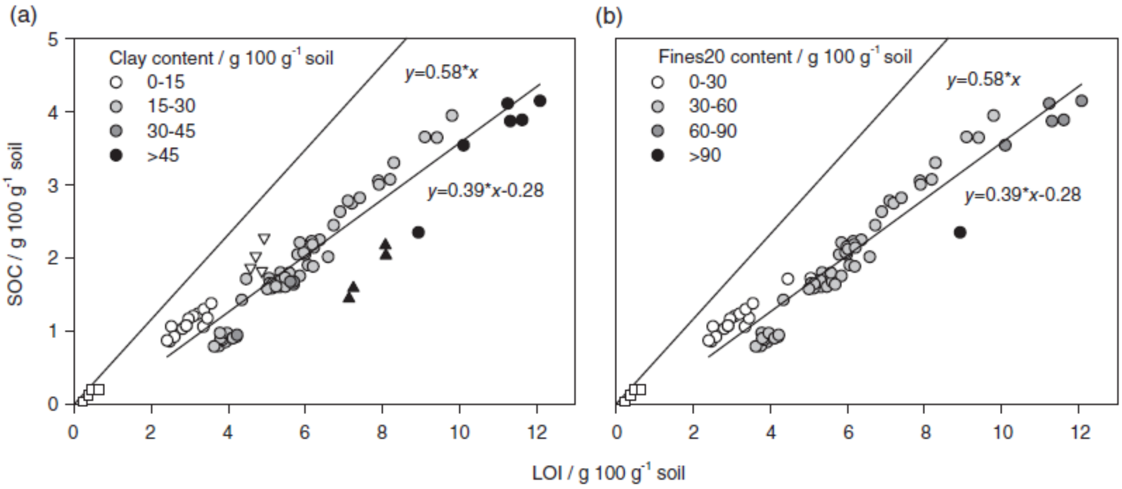


Figure 7. SOC as a function of LOI for samples grouped according to a) clay content and b) soil mineral particles <20 µm. The line representing the conventional relationship between LOI and SOC (SOC = 0.58 x LOI) is also shown (Source: Jensen *et al.*, 2018).

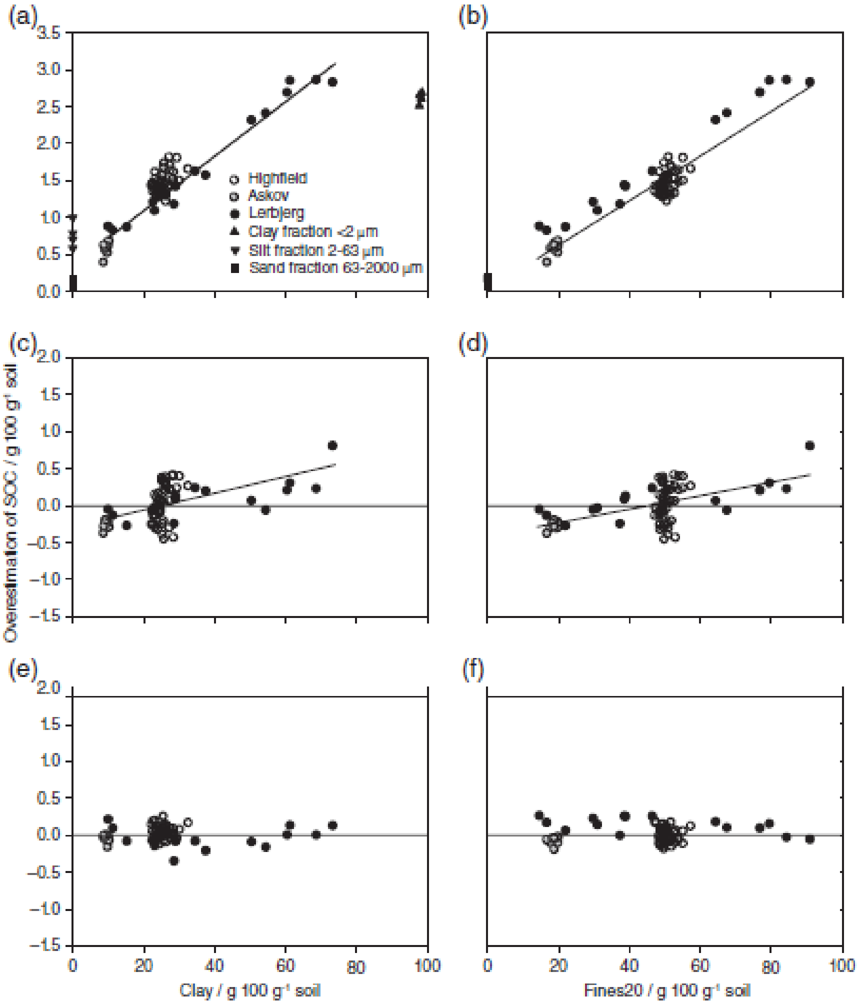


Figure 8. Overestimation (predicted minus measured values) of SOC as function of clay or mineral particles <20 µm (Fines20), a) and b) SOC = LOI x 0.58, c) to f) SOC, estimated based on modelled relationships. (Source: Jensen *et al.*, 2018).

3.3 Walkley Black method

- The Walkley and Black (1934) (WB) method involves rapid wet oxidation of organic carbon compounds by dichromate ions ($Cr_2O_7^{2-}$) followed by oxidation-reduction titration with ferrous ammonium sulphate. Authors have reported that the method can lead to incomplete oxidation of organic C, and poor digestion of elemental C forms (Wallace *et al.*, 2011). In addition, Walkley (1947) reported that the WB method recovered only 2–11% of SOC in carbonized materials.
- Walkley and Black (1934) showed that recovery of organic C using the WB procedure ranged from 60 to 86% with a mean recovery of 76% (a correction factor of 1.33 is commonly applied to the results to adjust the organic C recovery). This led to the development of a modified method which did not require a correction factor (Tinsley, 1950 and Mebius, 1960). The modified WB (MWB) method included extensive heating of the sample during digestion however, strict temperature control is required due to the possible decomposition of acid dichromate above 150˚C. Modified WB is no longer provided by commercial laboratories as the chromium reagents are carcinogenic and the residues must be stabilised before landfill disposal. Potassium dichromate is listed as a substance of very high concern in REACH regulation so authorisation is required to use it.
- Modified WB requires only a small soil sample (0.3-0.5 g of soil). However, the method is labour-intensive, requires a great deal of analytical skill, employs strong oxidants and acids that must be heated, and generates hazardous waste. Another disadvantage is that, apart from recalcitrant material, such as charcoal, the presence of iron and manganese oxides in weathered soils can be source of errors. Depending on soil type and C content, underestimation can be large (Davis *et al.*, 2018).

3.4 Dry combustion methods

- Dry combustion is a direct method that measures SOC content using combustion at high temperatures (1000°C) in an elemental analyser (Wallace *et al.*, 2011). The procedure usually takes place in pure oxygen to ensure complete combustion and the end product is CO_2 which is quantified by gas chromatography using a thermal conductivity or flame ionisation detector (FAO, 2019). Inorganic carbonates (part of the soil inorganic carbon pool-SIC) are removed by acidification prior to analysis, otherwise they will be incorrectly measured as part of the SOC pool. Alternatively, inorganic carbon present in the sample, is measured by a separate procedure and subtracted from total C after processing. In the absence of inorganic carbonates, dry combustion is considered the most reliable measure of SOC (Roper *et al.*, 2019).
- The main advantages are accuracy (results are taken directly from the instrument), complete combustion of all SOC and quick processing. Disadvantages include the cost of the laboratory instruments and the small sample size (8-10 mg to >1 g, depending on the equipment) which may increase the risk of errors caused by unrepresentative sub sampling (FAO, 2019).

3.5 Other methods

- New spectral methods for measuring SOC concentration and stocks are rapidly becoming available for direct laboratory and field measurements of SOC, but also for measurement of patterns at larger scales across landscapes and regions (Smith *et al.*, 2020). Soil spectroscopy uses the interaction of electromagnetic radiation with mineral and organic matter to characterize the physical and biochemical composition of soil (FAO, 2019). Light is shone on a soil sample and properties of the reflected light (visible-near-infrared, near infrared, or mid-infrared) are representative of molecular vibrations that respond to the mineral and organic composition of soils. Reflected or absorbed light is collected at different wavelengths by a detector and a mathematical model (based on spectral library datasets) is used to estimate SOC. These spectral libraries can also be used to calibrate field spectrometers, although accuracy will often be lower, mostly due to soil moisture content and surface roughness (Smith *et al*. 2020).

3.6 Comparison of methods

- Roper *et al*. (2019) compared four different techniques for quantifying SOM using 84 samples from long-term agricultural trials (comparing tillage practices and conventional and organic management) in three regions of North Carolina. The methods were MWB, LOI, automated dry combustion (ADC) and humic matter (HM) colourimetry.
- Methodologies for MWB, LOI and dry combustion were similar to those outlined above. Colourimetric measurements of humic matter followed conventional procedures for extracting humic acid from soils (Mehlich, 1984 cited by Roper *et al*., 2019; Hardy, 2014). In brief, extractant (containing sodium hydroxide, pentasodium diethylenetriaiminepentaacetic acid and ethyl alcohol) was mixed with air dried soil (<2 mm) and allowed to settle overnight. Absorbance of the undisturbed supernatant (diluted with water) at 650 nm was then measured in reference to a solution with only extractant and water; absorbance values were converted to a humic matter concentration based on calibration data.
- The comparison showed that on average SOC measured using MWB was 56% of the mass of SOM (by LOI), in line with the conventional conversion factor of 0.58. In comparison, the average mass of SOC using ADC was approximately 33% of the average mass of SOM (Table 5) measured by LOI. The average SOC measured by MWB (12.9 g C/kg) was 1.75 times greater than measured by dry combustion (7.7 g C/kg). In comparison, as would be expected, the average mass of HM was smaller (2.37 g HM/kg), approximately 10% of the average mass of SOM, reflecting the fact that HM is only part of SOM.
- As an elemental analyser provides a direct measure of SOC it can be considered as the reference value and used to assess the accuracy of the other methods. Accepting the assumption that the ADC value is correct then the data suggests that the MWB method is overestimating SOC; the authors were unable to explain why WB measured more SOC than ADC. As noted by Roper *et al.* (2019) if MWB results were corrected for incomplete oxidation then the overestimation of SOC content would be even greater. As reported above, results from MWB were in line with the common assumption that approximately 58% of the mass of SOM is C (although the range was 46-77%), whereas ADC results suggest that this assumption is inaccurate (range 22-53%). The site with the sandiest soil had the largest SOC fraction of SOM for both ADC and MWB results.
- An understanding of the relationships between results from the different methods would be helpful with interpretation of SOM concentrations measured using different techniques. Roper *et al*. (2019) noted that SOC results obtained using MWB and LOI were highly correlated, although the precise relationship varied on a site by site basis. Both ADC and MWB measurements were correlated to LOI but the authors noted that the correlation between ADC and LOI was greater than between MWB and LOI. This may reflect that ADC and LOI both involve dry combustion of organic material in contrast to MWB which relies on chemical oxidation of SOC.
- When comparing samples taken from different agricultural management practices results varied for different methods and locations. Roper *et al*. (2019) concluded that until a reliable method of converting results obtained using different analytical methods was established,

comparisons of SOM and SOC obtained using different techniques and soil types should be made with great caution.

Table 5. Comparison of four methods[*] to measure the soil organic matter (SOM), organic carbon (SOC) and humic matter (HM) of 84 North Carolina soils (Source Roper et al., 2019).

Method	Mean	Standard deviation	Min	Max	Range
	SOM (g/kg)				
LOI	23.2	5.6	14.5	38.8	24.3
	SOC (g/kg)				
MWB	12.9	2.1	9.7	18.8	9.1
ADC	7.7	2.1	3.7	13.3	9.6
	HM (g/kg)				
HM	2.37	1.29	0.3	6.6	6.3

[*] The methods used were loss on ignition (LOI), Modified Walkley-Black (WB), mass automated dry combustion (ADC), and humic matter (HM) colourimetry.

3.7 Conclusions

- One of the main differences between the NSI and CS databases was the methods used to measure SOC concentrations. NSI used MWB, which measures SOC directly and the CS used LOI, which measures SOM and requires a calculation to convert to SOC. The selection of an appropriate conversion factor introduces potential errors into the reporting of SOC, although, the use of a dataset specific conversion factor is likely to have minimised this error. However, where a measure of SOC (rather than SOM) is required best practice suggests that a direct measurement is most appropriate.
- In the comparison of methods undertaken by Roper *et al.* (2019) it was noted that the MWB method (used by the NSI) overestimated SOC in comparison to data from an elemental analyser. The overestimation was large (x 1.75) suggesting that this method would overestimate SOC concentration. Overestimation would lead to errors in reported values of SOC, however, it would not affect the reporting of change over time (assuming that the overestimation of SOC concentration was consistent at the start and end of the monitoring period).
- Importantly, Roper *et al*. (2019) concluded that until a reliable method of converting results between different analytical methods of measuring SOC/SOM is developed comparisons of SOM and SOC among different analytical methods and different soils should be made with great caution.

4 SOC concentration v SOC stock

- Calculation of soil carbon stocks requires determination of SOC concentration, bulk density (BD), stone content, and soil depth, which are spatially variable and have different measurement errors associated. There is a need to understand if all these variables contribute similarly to the variability of SOC stocks (Schrumpf *et al*., 2011).
- A comparison of methodologies for measuring SOC concentration have been discussed in the previous section, this section will discuss the other factors that influence the calculation of SOC stocks.

4.1 Sampling depth

- In most temperate mineral soils the majority of soil carbon is found in the topsoil where most of the organic material is returned from crop residues, organic material additions etc. The exceptions are organo-mineral soils which have an organic layer of ≤40 cm and deep peat soils which can have high organic matter levels beyond 1 m depth. Samples for SOC measurement are usually taken in the topsoil with 30 cm recommended by the Intergovernmental Panel for Climate Change (IPCC) and specified for national SOC inventories. However, the top 0-30 cm of the soil is more susceptible to short-term changes in SOC so samples taken to a greater depth may more accurately reflect long-term changes in SOC.
- To determine change of soil C stocks as a result of a change in land management, some authors have suggested that sampling should ideally cover rooting depths (Wendt and Hauser, 2013). However, this is only likely to practicable or worthwhile for field experiments comparing management practices over the long-term rather than for national scale monitoring. For example, Gregory *et al.* (2016) tested the hypothesis that contrasting long-term management systems changed the dynamics of SOC in the topsoil and subsoil (to 75 cm). The field experiment was established at site where control grassland was changed to arable (59 years before) and bare fallow (49 years before) systems. Losses of SOC were 65% and 78% under arable and fallow, respectively in the upper 0-15 cm, compared to grassland. In comparison, at 30-60 cm depth losses of SOC were 41% and 52%, respectively, compared to grassland. The authors concluded that there was a clear need to monitor, model and predict changes in soil properties not just near the surface but for the whole profile. The authors recommended a controlled experimental approach.
- Sampling depths of less than 30 cm may result in flawed conclusions regarding OC changes induced by tillage and land-use practices (e.g. Baker *et al*., 2007).
- Jobbáry and Jackson (2000) analysed data on >2700 mineral soil profiles (1271 under agricultural use and 802 under natural vegetation) from three global databases (United States Department of Agriculture National Soil Characterisation Database, World Inventory of Soil Emission Potential Database and the Canadian Forest Service soil database) to assess the vertical profile of soil organic carbon to 1 metre; data was also extrapolated to estimate deep SOC (100-300 cm depth). Data values were transformed to a volumetric basis using accompanying values of bulk density. For the upper 1 m of soil, the authors estimated that on average, *c*.40% of SOC was in the upper 0-20 cm of the soil and almost 65% was in the upper 0-40 cm; the lower 40-100 cm of soil contained *c*.35% of SOC (Figure 9). This data suggests, that for most mineral soils, a sample depth of 30 cm (as recommended by the IPCC) will capture changes in the majority of the SOC stock.

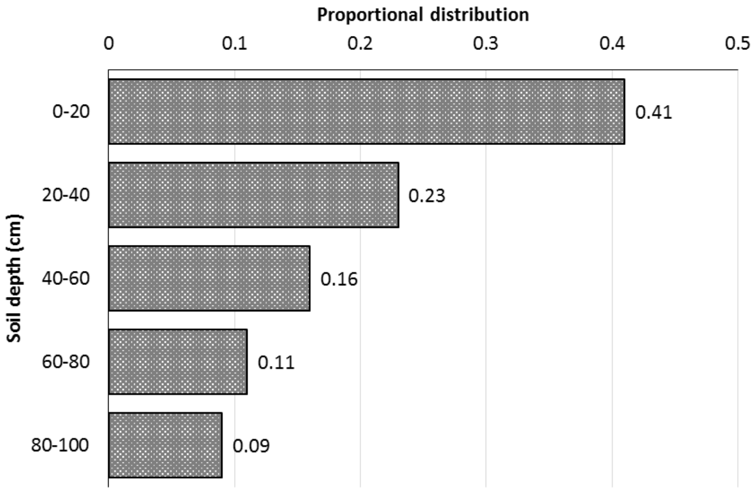
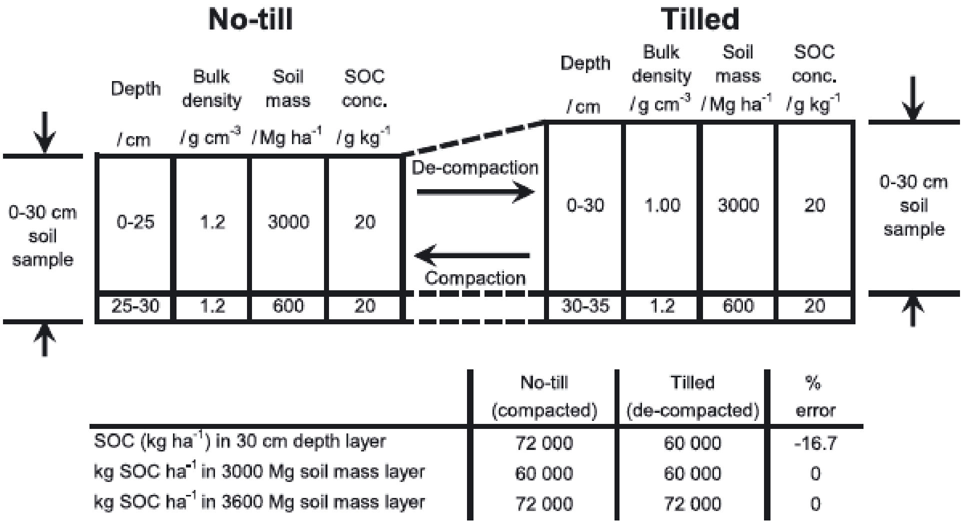


Figure 9. The global average vertical distribution of soil organic carbon in the upper 1 m of the soil profile (Source: Jobbáry and Jackson (2000)

4.2 Soil bulk density

- The SOC stock for a given depth is calculated from SOC concentration, bulk density (BD) and soil depth. Wendt and Hauser, (2013) reported how variations in BD across the depth of sampling due to short-term management changes can introduce bias in SOC determinations, especially in the surface horizon due to the prevalence of root and stubble biomass. In Figure 10 the effect of tillage on two treatments is shown, one soil is tilled to a depth of 25 cm whilst the other is left untilled resulting in a reduction in bulk density in the tilled soil (from 1.20 to 1.00 g/cm^3). The change in bulk density caused by tillage results in the 25 cm layer occupying 30cm depth. A comparison of the SOC concentration at the same depth (0-30 cm) in both soils would suggest that the untilled soil had 72,000 kg SOC and the tilled soil 60,000 kg SOC. This could lead to the conclusion that tillage decreased SOC by *c*.17%. In fact, this difference merely reflects differences in soil masses in the 0-30 cm layers, not changes in SOC stocks.



	No-till (compacted)	Tilled (de-compacted)	% error
SOC (kg ha^{-1}) in 30 cm depth layer	72 000	60 000	-16.7
kg SOC ha^{-1} in 3000 Mg soil mass layer	60 000	60 000	0
kg SOC ha^{-1} in 3600 Mg soil mass layer	72 000	72 000	0

Figure 10. Error bias introduced by quantifying SOC stocks at fixed depths when differences in bulk density exist (Source: Wendt and Hauser, 2013).

- Several methods are used to correct total SOC data in response to land use or management changes inherently coupled with concomitant alteration to bulk density (Lee *et al*., 2009), however, comprehensive evaluations of these methods are still lacking (Nayak *et al*., 2019).
- Soil bulk density varies with time reflecting the impacts of weather (e.g. freezing and thawing; wetting and drying) and land management. To account for these temporal changes BD should ideally be determined in the same core from which SOC concentration is measured (Schrumpf *et al*., 2011). This is because soil sampling using cores almost invariably results in some compaction and typically the soil density in the core exceeds the bulk density of the field (FAO, 2019).

4.3 Pedotransfer function

- A pedotransfer function (PTF) can used to predict a soil property from other measured variables (often properties that are quicker and cheaper to measure). For example, PTFs for bulk density reflect other soil properties which influence soil compaction, e.g. clay and SOC content (e.g. de Vos *et al*., 2005). Using PTFs often increases the variance and uncertainty of estimated SOC stocks if the error associated with the application of the function is not correctly accounted for. This may lead to a systematic bias of calculated SOC stock (Schrumpf *et al*., 2011) and high uncertainty in SOC estimation at regional scales (Xu *et al*., 2015).
- Schrumpf *et al*. (2011) used data from the CarboEurope network to investigate the potential use of PTFs for estimating BD in repeated inventories. The network included monitoring sites across Europe, under different land use, and in different climatic regions. In 2004, 100 geo-referenced soil samples were taken at 12 sites to a depth of at least 60 cm and analysed for OC, bulk density and stone/root content. The negative relationship between OC and BD was used to derive a pedotransfer function with known errors.
- For 10 sites, the factors contributing to the variance of OC stocks were compared following calculation using a) measured values and b) the PTF. The relative contribution of individual factors to the variance of SOC stocks was affected by the application of the PTF (Figure 11). The relative contribution of the uncertainty of BD to OC stock variances was on average 11–19% higher for estimated (including error propagation) than for measured values in the upper 30 cm of the mineral soil. The relevance of OC concentrations was 8–12% smaller.
- The estimation of BD using PTF also influences the variance of OC stocks and thus the possibility to achieve significant results. Schrumpf *et al*. (2011) concluded that not accounting for the additional uncertainty introduced to C stock estimates by the PTF leads to significant underestimation of the total variance at the plot/field scale. They suggest that where PTF has been used to estimate BD without accounting for the additional error of the PTF, incorrect assumptions about the significance of changes in SOC stocks are likely to have occurred.
- Schrumpf *et al*. (2011) suggested that changes claimed by past soil OC inventories where bulk density was not measured but derived from PTF, need to be re-considered carefully since most of the reports did not account for the uncertainty added to stock estimates by the function. They also concluded that the application of PTF to derive bulk densities was not recommended for future monitoring projects as changes are easier to detect with measured values and no reasonable procedure is available to allow for accounting of changes in BD with time (Schrumpf *et al*., 2011). Overall, although direct measurement of BD at the time of sampling might be laborious it improves change detectability and leads to more accurate results.

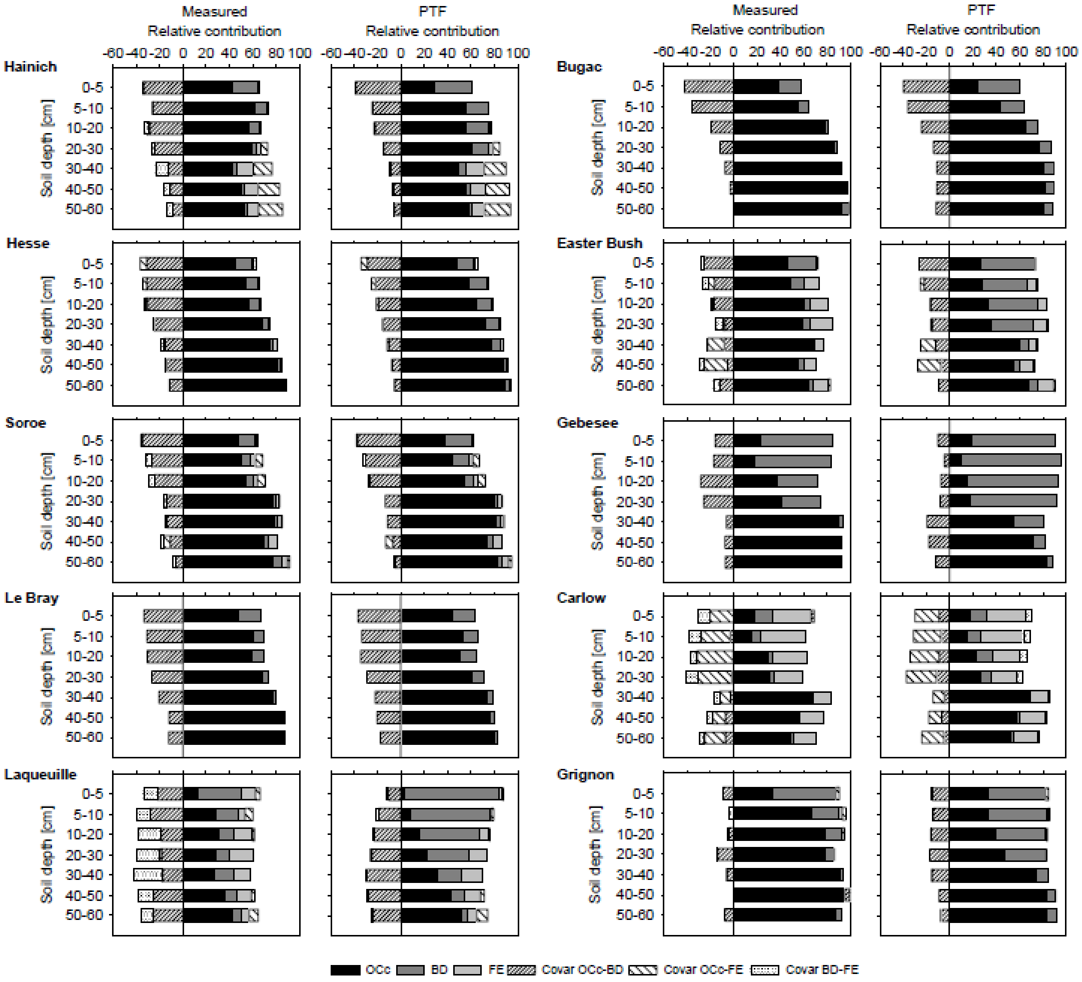


Figure 11. Relative contribution of organic carbon concentration (OCc), bulk density (BD), fine earth content (FE) and the covariance of OCc-BD, OCc-FE and BD-FE to the variance of SOC stocks (soil depth, cm; relative contribution %).

- Xu *et al.* (2015) used data from the second national soil survey in China (SNSSC) (8,210 soil samples) to investigate the effect of 8 different PTFs on estimates of SOC content. The observed and predicted SOC content of each PTF was compared (Figure 12). The predicted SOC content was underestimated by six PTFS and overestimated by two of the PTFs (M5 and M7). Estimates of SOC storage (0-100 cm) varied between 64.19 Pg C (M2) to 95.97 Pg C (M7), the average (of all 8 estimates) was 87.36 ± 8.93 Pg C. This research confirms that PTFs are an important source of uncertainty in SOC estimates.

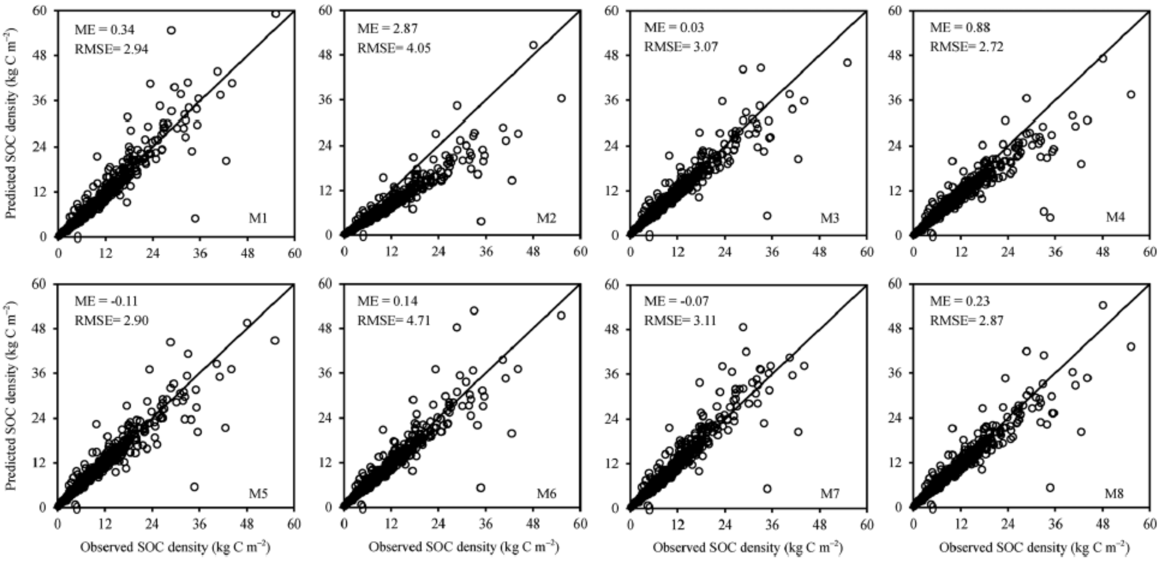


Figure 12. Comparison between observed and predicted SOC density by 8 pedotransfer functions with mean error (ME) (indicator of difference between predicted and actual values) and root-mean square error (RMSE) (an indication of how concentrated the data is around the line of best fit).

4.4 Equivalent soil mass

- Equivalent soil mass is defined as “the reference soil mass per unit area chosen in a layer” and equivalent C mass is “C mass stored in an equivalent soil mass” (Ellert *et al.*, 2001).
- As noted above (and in Figure 10), it is clear that if bulk densities differ, soil masses will differ, resulting in an unequal basis for comparison. In the above example, OC in 3000 Mg soil/ha in the tilled soil is compared with OC in 3600 Mg soil/ha in the soil before tillage, resulting in a substantial error. The source of this error is that OC stocks are being compared at fixed depths that contain different soil masses. Hence, some authors have argued that tracking changes in SOC stocks over time requires that the SOC stocks are compared for the same mass of soil, that is, by estimating SOC stocks on an equivalent soil mass (ESM) basis.
- Wendt and Hauser (2013) suggest that equivalent soil mass is best understood by visualising soil profiles in terms of mass layers instead of depth layers. The mass of a soil in a given depth layer will vary with bulk density, whereas the mass of soil in a mass layer is fixed and thus provides a consistent basis for comparing changes (Figure 13). To calculate the soil mass represented by a soil sample depth layer (*DL*), the dry sample mass (^{M}SAMPLE*(*DL*)*, g) is divided by the area sampled by the probe or auger ($\pi(D/2)^2$):

$$M_{\mathrm{SOIL(DL)}} = \frac{mass}{area} = \frac{M_{\mathrm{SAMPLE(DL)}}}{\pi\left(\frac{D}{2}\right)^2 \times N} \times 10000.$$

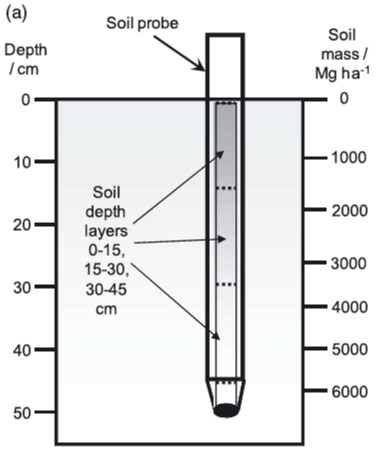


Figure 13. Soil depth layers (cm) and mass layers (Mg/ha) (Source Wendt & Hauser, 2013)

- The method of Ellert and Bettany (1995) is perhaps the most commonly used ESM method. To account for different soil masses Ellert and Bettany (1995) calculated the amount of C in an identical or 'equivalent' mass of soil under contrasting management regimes (forest, recently cleared, alfalfa/oilseed and wheat/fallow). SOC to a fixed depth was compared with SOC calculated using the ESM. Ellert and Bettany (1995) selected the pasture treatment as the equivalent mass treatment (soil mass 2,340 t/ha measured to 18 cm depth). Their calculations showed that to a fixed depth of 18 cm SOC was 52.4 t/ha for pasture compared to 45.7 t/ha for alfalfa/oilseed. However, when SOC was re-calculated on an ESM basis (i.e. using the same mass as the reference pasture treatment, which increased the calculation depth to 18.6 cm rather than 18 cm) the alfalfa/oilseed SOC content increased to 46.2 t/ha. The authors concluded that for management induced changes in SOC to be assessed reliably the masses of soil being compared must be equivalent.
- When the ESM layer is recorded and reported, it allows the possibility of returning to the same sampling site at a later date and calculating OC stocks at that soil mass, which is necessary when monitoring changes in soil OC over time (Wendt and Hauser (2013). The ESM approach is only likely to be practicable where management practices are being compared at a single site rather than for large scale national monitoring programmes.

4.5 Soil stoniness

- Laboratory measurements of SOC are generally done on sieved soil so that gravel or stones are removed before analysis. To correct this, laboratory results should be adjusted to reflect the original composition of the soil sample. For example, for a laboratory result of 1.4% organic carbon, measured on a sample with 10% gravel or stone, then the organic carbon content of that soil would be 1.26% (i.e. 90% of 1.4%). Such adjustments are sometimes overlooked and can lead to reports of rapid or unusually large changes in total soil organic carbon (GRDC, 2013).
- As noted previously, in regional-scale studies or national soil inventories, BD is often approximated using pedotransfer functions and the fraction of rock fragment is often ignored (Poeplau *et al.*, 2017). The IPCC regard stoniness and bulk density as the two major sources of uncertainty in SOC stock estimates for mineral soils (IPCC, 2003). However, even when all parameters are recorded, considerable difference can arise from varying use of the parameters in equations used for calculating SOC stocks.

- Poeplau *et al*., reviewed the literature on soil bulk density methodology and identified four methods for accounting for rock fragments, which are detailed in Table 6 below. All four methods use the same measurement of SOC (i.e. SOC of dried/sieved soil) but varied in the manner in which the parameters for bulk density and rock fragment were accounted for. In the first method (M1) the calculation 'fills' the space occupied by rock with fine soil (with an overestimated BD: based on the sample not the fine soil fraction) and the rock fraction is not subtracted from the total soil volume leading to an overestimation of BD and SOC stock. Similarly, in the second method (M2) SOC stock is also overestimated by 'filling' the space occupied by rock fragments with fine soil (BD estimated on the fine soil fraction). However, as both BD and SOC concentration are calculated on the fine soil fraction (in comparison to M1 where BD is based on the sample not fine earth fraction) the over estimation of SOC is smaller than in M1. The third method (M3) accounts for the rock fragment that can be assumed to be SOC free but overestimates BD by using sample BD rather than fine earth BD. The final method (M4) is suggested to be the closest to reality as rock fragments are accounted for and the BD is calculated only on the fine earth part of the sample. To compare the effect of the different methodologies on SOC stock the authors used data from the German Agricultural Soil Inventory (>3000 soil samples), which showed that difference in SOC stocks between calculation methods increased with rock fragment content (Figure 14).
- However, Hobley *et al*. (2017), disagreed that M4 was the best method to account for SOC suggesting that the extra sources of error in M4 (assumption of rock density) made this method less precise than the method that they proposed. The authors suggested a method based on the C content of the fine soil, mass proportion of rocks and bulk density in the whole sample would result in a more precise estimate of C stocks (Table 6).

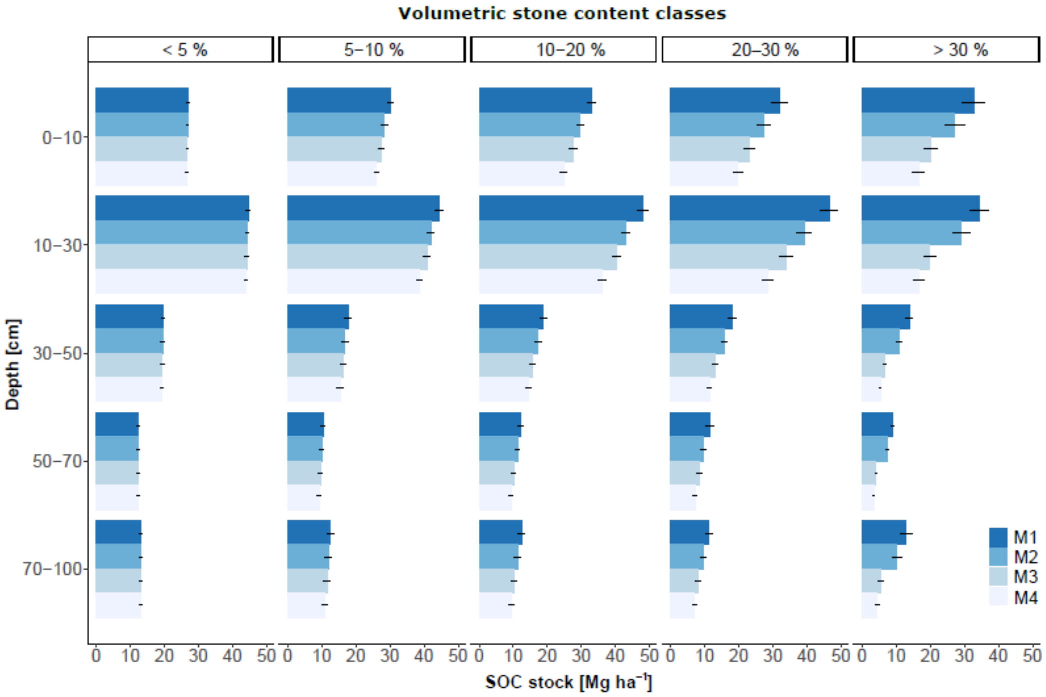
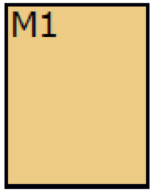
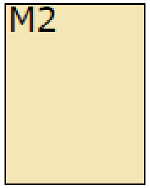
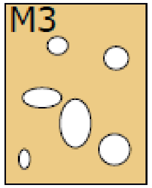
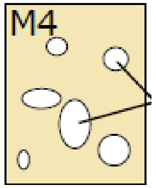


Figure 14. Average SOC stock of the German Agricultural Soil Inventory at different depth increments calculated by different calculation methods for five volumetric rock fragment content classes. Error bars indicate standard errors. (Source: Poeplau *et al*., 2017).

4.6 Conclusions

- Soil bulk density is a crucial factor that should be measured when quantifying SOC stocks (Schrumpf *et al*., 2011). A comparison of SOC stock at the same depth (e.g. 0-15 cm) in soils with different BD but the same SOC concentration would suggest that the soil with the lower BD had a lower SOC stock. In fact, this difference merely reflects differences in soil masses in the 0-30 cm layers, not changes in SOC stocks. Changes in BD are likely to occur more frequently in the surface horizons so that changes over time would be likely in the relatively shallow soil samples in both the CS and NSI.
- The CS in 2007 was the first national UK survey to estimate topsoil carbon stock from co-located bulk density and soil C concentration measurements. This gives greater confidence in the calculation of SOC for the most recent survey.
- For previous CS samplings (i.e. 1978 and 1998) a pedotransfer function based on the 2007 relationship between BD and SOC concentration was used to calculate SOC stock. Similarly, the NSI used a pedotransfer function (a different PTF from that used in the CS), rather than measured BD to calculate SOC stock. Using PTFs often increases the variance and uncertainty of estimated SOC stocks if the error associated with the application of the function is not correctly accounted for.
- Schrumpf *et al*. (2011) suggest that changes claimed by past soil OC inventories where bulk density was not measured but derived from PTF, should be re-considered carefully since most of them did not account for the uncertainty added to stock estimates by the function. They also conclude that the application of PTF to derive bulk densities is not recommended for future monitoring projects as changes are easier to detect with measured values and no reasonable procedure is available to allow for accounting of changes in BD with time (Schrumpf *et al*., 2011). Overall, although direct measurement of BD at the time of sampling might be laborious it improves change detectability and leads to more accurate results.
- The challenge remains to accurately estimate the rock content of sampled soils, which can significantly affect soil bulk density (e.g. Poeplau *et al*., 2017; Hobley *et al*., 2018). PTF often ignore the proportion of the soil made up of rock fragments, which is necessary to reflect the original composition of the soil sample. Such adjustments are sometimes overlooked and can lead to reports of rapid or unusually large changes in total soil organic carbon (GRDC, 2013). For the CS 2007 samples BD was calculated on samples excluding stones; where PTF were used to calculate BD then stones will not be accounted for.
- It is clear that if bulk densities differ, soil masses will differ, resulting in an unequal basis for comparison. Hence, some authors have argued that tracking changes in SOC stocks over time requires that the SOC stocks are compared for the same mass of soil, which can be done by estimating SOC stocks on an equivalent soil mass (ESM) basis. Changes in SOC over time in both the CS and NSI were compared over a fixed depth rather than on an equivalent mass basis.
- Overall, changes in management that influence carbon content also affect the bulk density of the soil (Haynes and Naidu, 1998), and thereby the amount of soil that is sampled within a given sampling depth. An ‘equivalent mass basis’ approach has been suggested as most appropriate for comparing SOC stocks across land uses and different management regimes (e.g. Ellert and Bettany, 1995; Wendt and Hauser, 2013). However, this approach may be impractical for national scale monitoring programmes.

Table 6. Methods for accounting for rock fragments in soil

Method		
	$BD_{sample} = mass_{sample}/volume_{sample}$ $SOC_{stock} = SOC\ \%_{fine\ soil} \times BD_{sample} \times depth$	- Known volume of soil is sampled, dried and weighed to determine BD. No separation into fine soil and coarse soil (rock fragments, roots) fraction. BD calculated for whole sample. - C concentration is determined in a sieved fine soil sample (usually < 2 mm). - Rock fragments not accounted for.
	$BD_{fine\ soil} =$ $mass_{sample} - mass_{rock\ fragments}/volume_{sample} - (mass_{rock\ fragments}/_{rock\ fragment\ density})$ $SOC_{stock} = SOC\ \%_{fine\ soil} \times BD_{fine\ soil} \times depth$	- Known volume of soil is sampled, dried and weighed to determine BD. After sieving mass and volume of rock fragments/roots determined. BD calculated for fine soil. - C concentration is determined in a sieved fine soil sample (usually < 2 mm). - Rock fragment density assumed, *c.*2.6 g/cm^3 in BD calculation.
	$BD_{sample} = mass_{sample}/volume_{sample}$ $SOC_{stock} = SOC\ \%_{fine\ soil} \times BD_{sample} \times depth \times (1\text{-rock fragments fraction})$	- Known volume of soil is sampled, dried and weighed to determine BD. After sieving mass and volume of rock fragments/roots determined - C concentration is determined in a sieved fine soil sample (usually < 2 mm). - Rock fragment used to reduce soil volume in SOC calculation.
 SOC free soil volume	$BD_{fine\ soil} =$ ‘5 $mass_{sample} - mass_{rock\ fragments}/volume_{sample} - (mass_{rock\ fragments}/_{rock\ fragment\ density})$ $SOC_{stock} = SOC\ \%_{fine\ soil} \times BD_{fine\ soil} \times depth \times (1\text{-rock fragments fraction})$	- Known volume of soil is sampled, dried and weighed to determine BD. After sieving mass and volume of rock fragments/roots determined - C concentration is determined in a sieved fine soil sample (usually < 2 mm). - Rock fragment density assumed, *c.*2.6 g/cm^3 in BD calculation & used to reduce soil volume in SOC calculation.
Hobley *et al., 2018.*	$SOC_{stock} = C_{content\ fine}\ (1 - mass\ proportion_{coarse}) \times BD \times depth$	- $C_{content\ fine}$ is the mass proportion of C in the fine soil fraction (g/kg). - Mass proportion$_{coarse}$ is the mass proportion of the coarse soil to the whole soil.

*Different shades of brown are used to indicate different densities: the rock fragment fraction has the darkest brown and the fine soil the lightest brown.

5 Monitoring change in SOC

- An incomplete understanding of how SOC changes are influenced by climate, land use and soil management factors adds complexity to designing appropriate monitoring, reporting and verification (MRV) platforms (Smith *et al*., 2020). However, accounting for changes in SOC remains an important policy objective (Jensen *et al*., 2018). For Wales, the Natural Resources Policy states the Government's aim to 'co-ordinate and embed best practice for the sustainable management of soil resources' (Welsh Government, 2017). Best practice will be 'informed by monitoring trends in carbon stock, a better understanding of the soil resource, the functional importance of soil biodiversity, and trends in and management of macro-nutrients'.
- There are two types of change that can be measured with regard to SOC stocks: a change over time relative to a baseline (or reference period), and a change relative to an alternative management strategy. Accurate estimates of changes in SOC stock rely strongly on robust baseline values, determined by physical sampling and measurements of soil C content. The challenge is to accurately capture the baseline and its inherent variability (Table 7).

Table 7. Sources of errors in SOC measurement at sample, profile, plot and landscape scales. The sources likely to produce high errors are in bold (Source: Vanguelova *et al*., 2016).

Scale	Source of Error
Sample	Soil composite samples are not homogenised Inconsistent analytical procedures for C **Bulk density is not assessed correctly** **Coarse fragments volume not assessed** Inaccurate separation of soil horizons and layers
Profile	Sampling by horizon versus soil depth depending on research aims **Sampling depth is insufficient to account for vertical variability**
Plot	**Micro-spatial variability not accounted for (not appropriate sampling strategy)** Statistical sampling error due to different sampling schemes Inaccurate geo-referencing (of the reported values) **Insufficient sampling points** **Bulk density and coarse fraction content not analysed** Analytical (measurement) errors including sample preparation Missing values, recording and truncation errors Model errors (e.g. from the selection of inadequate pedotransfer rules or functions, inadequate model constants and conversion factors, etc. not site/soil specific calibrated)
Landscape/National	Lack of local and regional representativeness of sampling plots **Important strata are underrepresented (e.g. wet mineral soils or peat soils)** **Lack of tree species/forest cover maps** **Lack of accurate soil/hydrology maps** Landscape insufficient resolution of climatic data

- A well-designed SOC monitoring programme for Wales must cost-effectively establish baseline SOC concentrations and stocks and be able to monitor change over time with confidence. The following sections consider the components of this monitoring scheme.

5.1 *Measuring SOC concentration and stock*

- To determine SOC concentration/stocks the following measurements are required:
    - Quantification of the fine earth (<2 mm) and coarse mineral fraction (>2 mm) of the soil
    - Quantification of the SOC concentration of the fine earth fraction
    - Soil bulk density or fine earth mass.
    - Depth of soil sample and or soil mass sampled.
    - Location of sample
- In addition, sample number and design also need careful consideration.

5.1.1 SOC concentration

- Accurate and precise determination of SOC contents is fundamental for reliable estimates of SOC stocks (e.g. Goidts *et al*., 2009; Schrumpf *et al*., 2011). Although, LOI has been commonly used in the past this is a measurement of SOM rather than SOC and a conversion factor is used to determine SOC. However, there is no universal agreement of the most appropriate conversion factor. A direct measurement of SOC directly, typically by automated, high-temperature dry combustion methods (Chatterjee *et al*., 2009) is likely to provide a more consistent and accurate quantification of SOC. However, a review of analysis costs for LOI and dry combustion methods from commercial laboratories indicated that dry combustion techniques was not offered by all laboratories and the cost was double that of LOI assessments (Table 7).

Table 8. Cost of SOC and SOM analysis at UK laboratories (January 2020).

Laboratory	Method	Cost (£ per sample ex VAT)
Lab A	SOM by LOI	£10.20
	SOC by dry combustion	£22.66
Lab B	SOC by dry combustion	£8.00
Lab C	SOM by LOI	£8.00

5.1.2 Soil bulk density and mass

- Accurate measurement of soil bulk density is essential for quantifying SOC stocks (Schrumpf *et al*., 2011). Soil bulk density, especially in the topsoil, will vary with soil/land management and weather. PTFs have been used to estimate bulk density, however evidence from the literature suggests that their use often increases the variance and uncertainty of estimated SOC stocks. Overall, direct measurement at the time of SOC sampling is likely to provide the most accurate assessment of bulk density. However the technique is laborious and more costly than using estimated values.
- Changes in soil bulk density will affect the amount of soil that is sampled within a given sampling depth. The use of an 'equivalent mass' in SOC stock calculations will take account of bulk density changes over time when comparing SOC stocks across land uses and different management regimes. However, the equivalent mass method may be more suited to comparing the effect of changes in soil management on the field scale.
- Measuring SOC and BD from co-located sampling should minimise errors in quantifying SOC stocks. Ensuring SOC determination is carried out on an equivalent mass basis rather than simply to a pre-determined soil depth will also reduce errors associated with changes in soil bulk density.

5.1.3 Depth of soil sampling

- Sampling to the appropriate depth is a crucial factor for properly evaluating changes in soil C content. In most soils the vast majority of organic matter will be present in the topsoil as this is where most of the organic matter from plant residues is returned. The IPCC specifies that for soil C inventories soil sampling depth should be at least 30 cm. This is likely to capture most short-term land-use and management (e.g. cultivation, fertiliser applications etc.)-induced changes in SOC stocks, although some practices can have impacts deeper in the soil profile (Smith *et al*., 2020). Deeper soil sampling (100 cm) is recommended (FAO, 2019), but often requires specific machinery and is costly.
- A sample depth of 0-30cm (combined with an assessment of bulk density) is likely to account for the majority of SOC and represent the best cost option. This is deeper than the samples previously taken for NSI and CS but is in line with IPCC recommendations for determining the impact of management induced changes on SOC stocks

5.1.4 Sample location/re-location

- When assessing the impact of changes in SOC overtime sampling from fixed locations is likely to minimise uncertainties associated with the assessments. A fixed location will allow consideration of any effects of land use change (Black *et al*., 2008). Sufficient site details should be taken to allow relocation after 5-10 years (e.g., GPS co-ordinates etc.).

5.1.5 Sample number and design

- The most appropriate design for a soil monitoring scheme will depend on the parameters measured and the objective of the scheme. Black *et al*. (2008) suggest that monitoring may be categorised as 1) status/ambient: to characterise or quantify SOC and assess change over time, 2) trend/effect: to assess the effects of pressures or management changes on SOC and 3) regulatory/compliance: to determine whether soils are meeting set standards or targets. Often a monitoring scheme will be required to address all three aspects.
- Black *et al.* (2008) compared sampling schemes for SOC monitoring. The study compared two-model based schemes 1) grid and 2) optimised grid and two design-based schemes 1) stratified random (by country and habitat) and 2) clustered stratified random (by country, habitat and clustered sampling). Model-based schemes assume that variations in SOC arise from a random process so that sampling points can be set out in a systematic way (e.g. NSI uses a 5 km grid). In contrast, a design-based scheme is based on random sampling from a population or sub-population (strata) (e.g. CS takes a stratified random sub-sample of a systematic sample). The stratified random scheme was found to be the most suitable option for addressing the status and change in soil organic carbon. Design-based methods with stratified random sampling allow targeted sampling, ensuring optimal spatial targeting of samples and also showed lower estimation variances than model-based strategies.
- Accurate measurements rely on appropriate study designs and sampling protocols to deal with high spatial variability of SOC stocks (Minasny *et al*., 2017). Often it is not possible to detect changes in SOC unless very high sample numbers are used and the cumulative changes with time are large enough to be detectable (Schrumpf *et al*., 2011). Large countrywide surveys with a low sampling density (<1 site per 100 km^2) generally adopt a stratified design to include all important units (van Wesemael *et al*., 2011). As noted above, the most common sampling design for networks aimed at monitoring regional/national SOC stocks is either stratified (according to soil/land use/climate) or grid based. The (expected) variability within these units

should be determined to assess the optimal number of samples that are required to ensure representative sampling (Smith *et al*., 2020).

- The calculation of the minimum detectable difference (i.e. the smallest difference in SOC stock that can be detected as statistically significant between two sampling periods) can be used to quantify the significance of any changes in SOC measured over The example below was reported by Lark (2009). In a 3-year field study with 5 replicates and initial SOC of 40-46 Mg C/ha (standard deviation of paired differences after the experimental period of 3 years 1.34) and expected SOC increases of 0.8 Mg C/ha/year would be able to detect differences in SOM of 2.58 Mg/ha after 5 years with a probability of 90%. In this example, the minimum detectable difference (MDD) is larger than the expected change (0.8 x 3 = 2.4 Mg C/ha) and the number of samples would need to be increased. With a sample number of 10 the MDD would be 1.55 and a sample size of 10 would be large enough to detect significant differences at the end of the experiment. However, this method assumes that the variation in samples means is already known. To help establish the most efficient use of sampling effort and establish variance, an initial exploratory resampling survey could be made to facilitate estimation of the variance of change.

$$\sqrt{\frac{1.34^2}{5}} \cdot (2.776 + 1.533) = 2.58$$

t values of 2.776 (critical value of the t distribution and 1.533 (one sided quartile of the t distribution). Degrees of freedom: 4

$$\sqrt{\frac{1.34^2}{10}} \cdot (2.262 + 1.383) = 1.55$$

t values of 2.262 (critical value of the t distribution and 1.383 (one sided quartile of the t distribution). Degrees of freedom: 9

- Another approach to monitoring change that has been proposed is the establishment of benchmark sites (Smith *et al*., 2020). Smith *et al*. (2020) suggested that the sites should be located on representative land cover/land-use types, soil types and with representative management. The sites could provide a platform to test management practices to increase SOC in fully randomized block designs, with SOC change measured over time (measurements every few years), and shorter term processes (such as GHG emissions) measured more frequently. In practice the resources required to set up benchmark sites would restrict the combinations of land use, climate, soil type and management practices measured.

5.2 Conclusions for monitoring programme

- The requirements for a comprehensive SOC monitoring programme include:
    - Direct measurements of soil organic carbon rather than infer from measurements of SOM.
    - Bulk density and SOC measurements from co-located samples to minimise uncertainties in the calculation of SOC stocks.
    - Sampling to 30 cm; this is deeper than the samples previously taken for NSI and CS but is line with IPCC recommendations.
    - An additional wider programme with more targeted paired comparisons of the effect of changes in land management practices on SOC.

6 Recommendations for a soil organic carbon monitoring in Wales

6.1 National monitoring scheme/repeated surveys

- A well designed national monitoring scheme will allow both spatial and temporal changes in SOC and SOC stock to be measured providing a countrywide overview of change. However, it may be difficult to link changes in soil carbon to any specific drivers in management between sampling dates (which may be many years) as detailed information is unlikely to be available.

- It is suggested that SOC sampling in Wales is organised using a design-based on stratified random sampling. The main reason for stratification is to improve sampling efficiency compared to simple random sampling. This method allows targeted sampling, ensuring optimal spatial targeting of samples and has been shown to have lower estimation variances than model-based strategies. Stratification might be based on land use (e.g. grassland v arable), rainfall and soil clay content; Verheijen *et al.* (2005) demonstrated the need to take account of both % clay and precipitation in SOC management and when investigating the effects of management on % SOC.
- IPCC guidance suggests that samples should be taken to a depth of 30 cm (this should also capture most short-term changes in SOC caused by changes in agricultural practice). Standard soil sampling protocols (e.g. AHDB's Nutrient Management Guide) suggest that a bulked sample made up of *c.*25 cores should be taken at each site. Sample location should be accurately recorded (e.g. with GPS) along with farm activity data (e.g. cropping, cultivations, organic manure/nutrient use), using a number of defined categories based on land management. This will allow follow up samples to be taken from the same point so that change over time can be accurately assessed.
- Accurate estimates of SOC are more likely to be achieved by direct measurements of SOC rather than by inference from measurements of SOM, which are complicated by the need to use a conversion factor to infer SOC.
- Estimates of national SOC stocks also requires sufficient sample numbers to ensure the database provides an accurate representation of the stratification. For example, in Wales, CS soil samples were taken from 107 1 km squares (in 8 land classes) equating to a sampling intensity of 0.5% of the total Welsh land area (21,091 km^2) (Wood, 2011). Sample numbers were allocated according to the land class stratification so that 0.5% of the 1 km^2 squares in each class were sampled. If sampling was again based on a 1 km grid square stratification then 210 samples would be required to achieve a sampling intensity of 1% and 2100 to achieve a sampling intensity of 10%.
- It is recommended that measurements of bulk density should be taken from co-located samples to minimise uncertainties in the calculation of SOC stocks. Pedotransfer functions (that use other variables to estimate bulk density) may be used, however evidence from the literature suggests that their use often increases the variance and uncertainty of estimated SOC stocks. Overall, direct measurement at the time of SOC sampling is likely to provide the most accurate assessment of bulk density and SOC stock. However the technique is laborious and more costly than using estimated values.
- Additional measurements of soil pH, P, K and Mg, combined with activity data describing land management practices in fields sampled for the survey, would provide useful information to help interpret and explain differences in SOC stocks between sites.

6.2 Field-scale monitoring of SOC change

- Field-scale monitoring of SOC can be used to complement national monitoring and provide more detailed information on the effect of specific management practices on SOC stocks. This could include a programme of field experiments or targeted paired comparisons of the effect of changes in land management practices (e.g. conversion of grassland to arable), on SOC. Proposed practices to increase SOC could be tested in fully randomised block designs with SOC and bulk density measured at regular intervals to monitor change. Site locations should be chosen in line with the parameters used to stratify the national monitoring scheme.

7 Overall conclusions

- In order to reward management practices that maintain or enhance SOC it is essential that any chosen metrics should firstly be able to establish baseline levels and secondly monitor either increases or decreases from the baseline. Two national soil monitoring programmes (the National Soil Inventory (NSI) and the Countryside Survey (CS) have reported conflicting evidence with regard to changes in topsoil carbon in England and Wales. The NSI reported losses in soil carbon from 1978 to 2007 whereas the CS reported no change. More recent soil samples taken for the Glastir Monitoring and Evaluation Programme (GMEP), between 2012 and 2016 also indicated no change in topsoil carbon for Wales (Emmett *et al*., 2017). This discrepancy in SOC levels reported by the two monitoring programmes makes it difficult to confidently set a baseline for SOC.
- A review by Kirk *et al.* (2011) ruled out differences in statistical interpretation, sampling or analysis methodology, sampling design or site re-location as causes for the differences between the two surveys. However, they did not identify any factors that conclusively explained the difference were identified.
- One of the main differences between the NSI and CS was the different methods used to measure SOC concentration. Whilst the NSI used MWB, which measures SOC directly the CS used LOI, which measures SOM; SOM is later converted to SOC using an appropriate conversion factor. The selection of an appropriate conversion factor (to convert SOM to SOC) introduces potential errors to the estimate of SOC, although, the use of a dataset specific conversion factor (in the CS) should have minimised the uncertainty. However, where a measure of SOC (rather than SOM) is required then best practice suggests that this should be measured directly rather than extrapolated. The different methodology used to determine SOC concentration makes a direct comparison difficult. Importantly, Roper *et al*. (2019) concluded that until a reliable method of converting results from different analytical methods of measuring SOC/SOM is developed (e.g. MWB and LOI as used in the NSI and CS, respectively), comparisons of SOM and SOC among different analytical methods and different soils should be made with great caution.
- As well as variation in SOC concentration over time soil bulk density also varies with time. This can have significant effects on the calculation of SOC stock. A comparison of SOC stock at the same depth (e.g. 0-30 cm) in soils with different BD but the same SOC concentration would suggest that the soil with the lower BD had a lower SOC stock. In fact, this difference merely reflects differences in soil masses in the 0-30 cm layers, not changes in SOC stocks. Changes in BD are likely to occur more frequently in the surface horizons so that changes over time would be likely in the soil samples taken at 0-15 cm in both the CS and NSI studies.
- The CS in 2007 was the first national UK survey to estimate topsoil carbon stock from co-located bulk density and soil C concentration measurements. This gives greater confidence in the calculation of SOC for the most recent survey.
- For previous CS samplings (i.e. 1978 and 1998) a pedotransfer function based on the 2007 relationship between BD and SOC concentration was used to calculate SOC stock. Similarly, the NSI used a pedotransfer function (a different PTF from that used in the CS), rather than actual BD measurements to calculate SOC stock. The different methods of assessing soil bulk density makes a direct comparison between the data from each survey difficult. Using PTFs often increases the variance and uncertainty of estimated SOC stocks if the error associated with the application of the function is not correctly accounted for.
- The uncertainty added to stock estimates through the use of a PTF increases the uncertainty of SOC stock estimated by the NSI and earlier CS samples. The application of PTF to derive bulk densities is not recommended for future monitoring projects as changes are more accurately

determined with measured values. Overall, although direct measurement of BD at the time of sampling might be laborious it improves change detectability and leads to more accurate results.

- It is clear that if bulk densities differ, soil masses will differ, resulting in an unequal basis for comparison. Hence, some authors have argued that tracking changes in SOC stocks over time requires that the SOC stocks are compared for the same mass of soil. This can be achieved by reporting SOC stocks on an equivalent soil mass (ESM) basis, although this approach is only practicable for field scale rather than national monitoring.

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