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Organic carbon contents of mineral grassland soils in Switzerland over the last 30 years

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ABSTRACT

Long-term monitoring data on the evolution of soil organic carbon (SOC) content and soil stocks are crucial considering climate change and carbon sequestration. Due to the slow processes involved, it takes longer time spans until changes in SOC stocks become detectable. Thus, long-term monitoring is essential.

Beside measurements, models are often used to report changes in SOC stocks for greenhouse gas (GHG) inventories. Consistent field data over extended periods are essential for validating such models whereby data from long-term monitoring is of great value. We present SOC measurements from the topsoil (uppermost 0.2 m) of 24 grassland sites of the Swiss Soil Monitoring Network (NABO) for six sampling campaigns from 1985 to 2014 and compare their evolution with predictions by the RothC model, which, for example, is used for the Swiss GHG Inventory. While there was no general temporal trend of the measured data over all sites, from a single site perspective, SOC stocks decreased/increased on 8% and 25% of the sites respectively. When comparing the measured data with the predictions from modelled values from RothC, a good fit was achieved in two thirds of the cases. However, at 5 out of the 24 sites (21%), the modelled SOC changes over time were lower than the measured ones. For 3 out of the 24 sites (12%) the model predicted higher SOC stock increases than were measured. Overall the modelled data fit well together with the measured ones. Most of the sites revealed stable values over time or increasing trends between 1985 and 2014, which might indicate that the SOC stocks potentially are in equilibrium and that the farming management used in Switzerland is suited to maintain these stocks in topsoils. There might not be a big potential for C sequestration, however, it is also very important to maintain SOC stocks over time. For future climate change scenarios accurate long-term SOC data and modelling are important for gathering the required information.

1. Introduction

Soil organic carbon (SOC) plays a major role in regulating general soil quality and soil functions, such as filtering and buffering of pollutants, nutrient cycling and water regulation (Wiesmeier et al., 2019). It strongly influences physical, chemical, and biological properties of soil, and it is, in turn, also influenced by them. The upper 100 cm of soil hold approximately 1500 Gt of carbon (C) globally (Batjes, 1996; Jobbágy and Jackson, 2000), whereby the SOC stock is estimated to be between 480 and 790 Gt (Minasny et al., 2017). Considering global climate change scenarios and greenhouse gas (GHG) emission budgets, SOC is of major importance because it can both act as sink and as source for atmospheric C. For example, land use change can lead to higher soil respiration or decomposition and thus C loss into the atmosphere. While Lal (2004a) estimated that approximately one-thirds of the anthropogenic C released into the atmosphere originates from land use change, and two-thirds from combustion of fossil fuel, Friedlingstein et al. (2020) reported emissions of 9.6 GtC yr^{-1} from fossil fuels and 1.6 GtC yr^{-1} from land use change for 2010–2019. However, the balance between C emissions and uptake might strongly differ between different ecosystems and sites (Don et al., 2011). Additionally, soil degradation can lead to methane (CH_4) release due to methanogenesis and nitrous oxide (N_2O) release due to nitrification and denitrification, and these are two strong GHG (Chang et al., 2021; Lal, 2021; Wang et al., 2021). In contrast, atmospheric C (CO_2 and CH_4) can also be sequestered from the atmosphere into soils mainly by biological processes, e.g., by plant

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uptake of CO_2 by photosynthesis and thereafter C release to soils by root exudation of photosynthesis products or microbial decomposition of plant-tissues. In less extend C can be sequestered to soils by chemical processes, such as mineral carbonation (Lal et al., 2008). It remains debated whether soils act as sink or source for atmospheric C (Chang et al., 2021) since interactions between increasing temperatures, the activity of soil microorganisms, plant growth, soil degradation etc. and their impact on SOC are complex (Jansson and Hofmockel, 2020; Naylor et al., 2020).

To investigate changes in SOC content and stocks, long-term soil monitoring programmes deliver an indispensable basis. In Europe, several countries and regions run programmes assessing SOC contents and stocks over different landscapes and different land uses. European soil monitoring programmes, which include SOC as target and with at least two different time-points measured, are listed in Table 1. There is only a limited number of programmes with data for more than two time points. Summarising the findings of the considered European monitoring programmes (Table 1), no general trend for permanent grasslands was found: depending on region and management strategies, increasing, decreasing, and stable SOC evolutions were reported. Similar results were reported for arable land as summarized by Gubler et al. (2019) for different European monitoring programmes. For instance, for arable land in Switzerland, no general trend for the 30 assessed sites of the Swiss Soil Monitoring Network (German: Nationale Bodenbeobachtung, NABO) was observed from 1990 to 2014. However, when focusing on single sites, 9 out of 30 sites showed an increasing and 4 a decreasing trend for SOC gradually (Gubler et al., 2019). In addition to monitoring programmes, there are several field trials investigating SOC contents or stocks over time, for example, the long-term experiments in the United Kingdom (Hopkins et al., 2009; Johnston et al., 2009) or in Switzerland (Fliessbach et al., 2007; Keel et al., 2019).

Moreover, soil monitoring programmes are important bases for spatial modelling of SOC stocks (van Wesemael et al., 2011), as shown for France (Martin et al., 2014, 2011), Belgium (Chartin et al., 2017; Meersmans et al., 2011), Germany ("Boden-Dauerbeobachtungsflächen", (FAO, ITPS, 2020; Nerger et al., 2017), and for the whole of Europe (Aksoy et al., 2016; Lugato et al., 2014). For Belgium, the RothC model (Coleman et al., 1997; Jenkinson et al., 1990) was used to assess drivers for SOC stock changes between two sampling campaigns in 1960 and 2006, respectively (Van Wesemael et al., 2010). In Switzerland, RothC is used to model SOC stocks in mineral agricultural soils for the national GHG inventory (FOEN, 2021; Wüst-Galley et al., 2021, 2020).

Modelling the evolution of SOC is important for predicting the C sink or source function of soils and ecosystems. To verify model output, long-term data on SOC stocks provide a useful basis. Because only few monitoring programmes for SOC exist, the modelled evolutions of SOC over long timeframes (decades) has rarely been compared against measured SOC at the same timeframe for specific land uses. In Denmark, Taghizadeh-Toosi et al. (2014) compared the modelled results for different soil types from the model "C-TOOL" with the measured data of the Danish national monitoring network. Application of organic manure, use of cover-crops, and converting croplands to grassland showed the potential to increase SOC according to the model used. For Belgium, Meersmans et al., (2009, 2011) modelled, based on data from two time points (1960 and 2006), spatial SOC distribution changes and also depth dependent spatial SOC distribution.

We assess the evolution of SOC contents in the topsoil (0–20 cm, mineral soils) of 24 Swiss permanent grassland sites of the Swiss Soil Monitoring Network (NABO) from 1985 to 2014 (representing 6 sampling campaigns). In a second step, the evolution of SOC for the same sites was modelled using RothC, and the results were compared with the measured data. For modelling, we used site specific information on the agricultural management and also the estimations used for Switzerland's GHG inventory. We addressed the following research questions: (i) How did SOC contents evolve over three decades in general and at individual sites? (ii) Which soil properties or environmental and management conditions might explain differences in evolution between sites? (iii) How effective are temporal SOC evolutions estimated by the RothC model? (iv) What factors explain differences between measured data and the used model?

2. Methods

2.1. Long-term monitoring sites

The Swiss Soil Monitoring Network (NABO) operates on approximately 100 long-term soil monitoring sites across Switzerland (Gubler et al., 2015). Most of them were sampled for the first time between 1985 and 1989 and re-sampled every five years. The present study considers 24 monitoring sites, which are used as permanent grassland on mineral soils, and were part of the monitoring programme since the first sampling campaign (Fig. 1). These grassland sites are managed by independent farmers without conditions, i.e. they are not experimental sites. Out of these 24 sites, 13 are located above 1000 m a.s.l. including 3 sites above 2000 m a.s.l. (Table 2).

Only the topsoil layers (0–20 cm) were investigated, and the SOC contents ranged roughly from 30 to 100 g C kg^{-1}, the pH ($CaCl_2$) values ranged from 3.6 to 6.4, whereas the fractions of clay and silt ranged from 9% to 46% and 23–55%, respectively (Table 2). The apparent density (AD; defined as mass of fine earth [particles < 2 mm] per total soil volume [including stones and pores]) ranged from 0.58 to 1.15 g cm^{-3}, with 5 of the 24 sites showing densities above 1.0 g cm^{-3}. The variety of soil conditions and agricultural management of these sites reflect typical situations for different agro-ecological zones of Switzerland, but the sites cannot be considered as representative in a statistical sense as they were initially targeted to monitor heavy metal pollution.

Agricultural management data has been recorded annually since 1985 for 11 of the 24 investigated sites, based on farmers' declarations (Gross et al., 2021a). Available annual data included among others the application of solid and liquid farmyard manure for the specific parcel of land, on which the soil monitoring site is located (Table 3, Gross et al., 2021b). Considerable variation was observed for manure inputs over the whole period, and mean annual inputs per site ranged from 0 to 2900 kg dry matter (d.m.) ha^{-1} yr^{-1} for solid manure and from 0 to 4400 kg d.m. ha^{-1} yr^{-1} for liquid manure (Table 2, Figure SI 1 and 2). Farming management intensities were based on the manure input of the whole farm (not specific for the sampling lot) for these 11 sites. For the remaining 13 sites, management intensity was estimated based on experiences from visits of the site, the flora on the parcel, and occasionally the discussions with local farmers.

2.2. Soil sampling and sample preparation

From 1985–2014, all sites were sampled during six sampling campaigns at five-year intervals, except for site 22 (only 5 sampling campaigns available). For each (re-)sampling, four replicate soil samples of the topsoil were collected from a precisely located area of 10 m x 10 m using a stratified random sampling design as described by Gubler et al. (2019). Each of the four replicate soil samples consisted of 25 bulked subsamples. Site 13 was an exception where only one sample (instead of four replicate samples) consisting of 25 subsamples was taken at the first sampling campaign. Soil samples were taken using a gouge auger (Eijkelkamp) of 2.5 cm diameter (c.f. SI Fig. 3). The 10 m x 10 m sampling square was accurately re-localised by well-documented reference points and buried magnets. At each of the four edges of the 10 m x 10 m sampling square one volumetric soil sample (HS Impact probe, GreenGround, 4.8 cm diameter, 25 cm length, sampling 0–20 cm depth) was taken to measure the apparent density of the fine earth (< 2 mm). All samples were oven-dried at 40 °C and subsequently crushed and sieved to remove coarse soil components (> 2 mm diameter). The samples were mixed using a Turbula shaker (Type T2C, Willy A. Bachofen AG) prior to

Table 1
European soil monitoring programmes, which include(d) SOC and investigate(d) at least two time-points. Abbreviations: Purpose: A routine analysis for agriculture, M Monitoring, map soil map, R Research. Land use and Sampled depth: C crops, G grassland, F forest. Analytical methods: WO: wet oxidation method, e.g. Walkley-Black, DC: dry combustion (CN analyser), LOI: loss on ignition. LOI[x]: LOI for carbon-rich soils; WO[y]: WO until 1994, DC thereafter.

Location / Source	Purpose	Land use	Period assessed in publication	Time points	Design	Relocation accuracy (m)	Sampled depth (cm)	Sampled area (m x m)	#of subsamples	Analytical method	QA of lab	Main findings for grassland over all investigated sites		
												conc	stocks	Summary
Schleswig-Holstein (Germany) (Woloszczyk et al., 2020)	M	C, G, F	1995–2015	3–8	paired	n.s. high	G:0–10, C:0–30, F:2–5	(32 m transects)	4 × 15	DC	There were different labs involved over time.	≈		SOC contents were compared between the two time periods of 1995–2002 and 2005–2015. Mean SOC contents over all sites were 30.9 at the beginning and 34.3 g C kg^{-1} at the end, and this change was not significant (p-value > 0.05). Three sites showed SOC content declines, five sites slight carbon gains with major increase at SOC-poor sites (n = 15 Grassland sites).
Bavaria (Germany) (Capriel, 2013)	M	C,G	1986–2007	4	paired	n.s. high (GPS located & magnet)	G: 0–10, C: 0–15	1000	4 × 25	DC	?	≈		Between 1986 and 2007 from a total of n = 21 grassland sites 12 did not show any changes in the SOC content, six sites had a significant decline (by average 1.6%) and three sites showed a significant increase by 1.7%. A negative relationship between initial SOC content in 1986 and the amount of OC change has been observed.
Bavaria (Germany)(Kühnel et al., 2019)	M	G	1989–2016	5	paired	n.s. high	0–10	1000	4 × 25	DC	?		≈	Between 1989/1990 and 1996/1997 mean SOC stock decreased by 2.6%. Between 1996/1997 and 2005/2006 the mean increase was 0.4%. Between 2005/2006 and 2012 an increase of 2.6% occurred. From 2012–2015/2016 a mean decrease of 3.3% occurred. (four years, n = 20 sites, 69 observations because some sites were established later than 1989 or were excluded because of changes in land use).
	M	C, G	1955–2005	2	paired	n.s.	0–30	radius 4 m	5	WO, LOI[x]	n.s.	+	+	

(continued on next page)

Table 1 (*continued*)

Location / Source	Purpose	Land use	Period assessed in publication	Time points	Design	Relocation accuracy (m)	Sampled depth (cm)	Sampled area (m x m)	#of subsamples	Analytical method	QA of lab	Main findings for grassland over all investigated sites		
												conc	stocks	Summary
Wallonia (Southern Belgium) (Goidts and van Wesemael, 2007)														SOC contents were compared between the sampling campaigns of 1955–1970 and the re-sampling campaign 2005 (50 ± 3 years between re-samplings, n = 37 sites). Significant SOC increase of 5.9 g C kg^{-1} for a mean SOC content of 16.0 g C kg^{-1} in 1955 on units under grassland. This results in an increase of 0.12 g C kg^{-1} $year^{-1}$. SOC stock on grassland increased by 21.9 t C ha^{-1} resulting in a stock change of 0.42 t C ha^{-1} $year^{-1}$.
North-western France (Ellili et al., 2019)	M	C,G	2009–2016	2	paired	3 m	0–7.5, 7.5–15, 15–30, 30–45	1 m radius	4	DC	n.s.		+	From 2009–2016 SOC stock increased by 1.40 ± 0.31 t C ha^{-1} $year^{-1}$ (n = 14 sites).
The Netherlands (Provinces Derenthe, Overijssel, Gelderland and Noord-Brabant) (Hanegraaf et al., 2009)	A	C, G	1984–2004	4–5	paired	n.s.	0–5 until 2003, thereafter 0–10	20,000 m^2	40	WO, DC, LOI (Methods see Reijneveld et al., 2009)	See Reijneveld et al. (2009)	≈		No overall trends in SOM contents over 20 years (1984–2004). SOM declined in ca. 25% of all grassland sites and 75% of the sites were stable in SOM content or showed an increase of > 1%. Carbon accumulation in grassland sandy soils in province Derenthe was calculated at 39 g C m^{-2} $year^{-1}$ (top 5 cm). SOM contents were linearly and negatively related to absolute changes in SOM. Sampling depth was different in the time spans before and after 2003. Values from 2003 on were recalculated to sampling depths of 0–5 cm.
The Netherlands (9 regions) (Reijneveld et al., 2009)	A	C, G	1984–2000	depending on site app. Every four years	paired	n.s.	0–5	max. 20000 m^2	1 × 40	WO, DC, $LOI^{x,y}$	same laboratory and reference samples included.	≈		Mean annual change in SOC contents of 0.1 ± 0.14 g C kg^{-1} $year^{-1}$. Annual changes in SOC varied between the different regions (3 regions with increasing

(*continued on next page*)

Table 1 (*continued*)

Location / Source	Purpose	Land use	Period assessed in publication	Time points	Design	Relocation accuracy (m)	Sampled depth (cm)	Sampled area (m x m)	#of subsamples	Analytical method	QA of lab	Main findings for grassland over all investigated sites		
												conc	stocks	Summary
														and 4 regions with decreasing SOC changes).
The Netherlands (Knotters et al., 2022)	map	G, A	1998 and 2018	2	paired	n.s.	0–30, 30–100	2 m radius	1 × 5	DC, LOI	n.s.		≈	SOC stocks of mineral soils under grassland did not change significantly between 1998 and 2018 (0–30 cm: 103.66 and 100.84 t C ha^{-1} respectively, and 30–100 cm: 144.71 and 127.15 t C ha^{-1} respectively).
Scotland (Chapman et al., 2013)	M	C, G, F	1978–1988, 2007–2009	2	paired	n.s. high	0–75–0–100	from soil profile	n.s.	DC, LOI	reanalysing old and new samples together	- 0	0 0	In 'improved grassland' (ley in rotations or permanent grassland occasionally ploughed) SOM decreased significantly from 27.0 ± 3.8–23.6 ± 3.2 g C kg^{-1} (0–100 cm all horizons, not depth-weighted). For semi-natural grasslands no significant changes were detected (94.3 ± 14.9 g C kg^{-1} at first campaign and 99.6 ± 15.5 g C kg^{-1} at the second one). No significant change in SOC stock in 0–100 cm depth.
Majorca Island (Spain) (Rodríguez Martín et al., 2019)	A,M	C, F, G	2006 and 2016/17	2	paired (?)	n.s.	0–25–0–30	?	1 x ≥ 10	WO	same methods and lab for both time points.	+		Increase from 2006 to 2017 from 2.00 t ha^{-1} to 2.31 t ha^{-1} (16.7%).
Ireland, south eastern region (Zhang and McGrath, 2004)	A	G	1964, 1995/96	2	unpaired	n.s.	0–10	n.s for 1964; 20 × 20 in 1995/96	15 (1964) and 25 (1995/96)	WO	n.s.	≈		220 sites from two sampling campaigns in 1964 and 1996 were compared by modelling a map (unpaired data, different sites, but same region). No general trend between the two campaigns was found. However, on a regional level the mean SOC concentration in the eastern costal sector increased by 30% and a mean decrease of 16% occurred in the inland sector.
Belgium (Lettens et al., 2005a)	A, R	C, G	1990, 2000	3	unpaired	n.s.	0–6/0–15 or sampling by	n.s.	n.s.	WO	n.s.		–	Comparing data from different independent

(*continued on next page*)

Table 1 (*continued*)

Location / Source	Purpose	Land use	Period assessed in publication	Time points	Design	Relocation accuracy (m)	Sampled depth (cm)	Sampled area (m x m)	#of subsamples	Analytical method	QA of lab	Main findings for grassland over all investigated sites		
												conc	stocks	Summary
							horizon, depending on region and survey program							surveys (unpaired design): Grassland decrease of SOC stock from 6.4 to 6.0 kg C m^{-2} in 0–20 cm soil depth, from 8.4 to 7.9 kg C m^{-2} for 0–30 cm, and from 13.9 to 13.0 kg C m^{-2} for 0–100 cm soil depth for 1990 and 2000, respectively.
Belgium (Lettens et al., 2005b)	A, R	C, G	1950–70, 1990, 2000	2	unpaired	n.s.	1950–70 horizon dependent, 0–6/0–15, 0–30 depending on region and land-use	n.s.	n.s.	WO	n.s.		+	Landscape units were formed and unpaired samples per Unites were compared between 1950 and 70, 1990, and 2000. This data originates from different independent surveys. For grassland SOC stock increases in the upper 30 cm (+9 t C ha^{-1}) between the sampling campaign 1950–70 and 2000. Overall sites first an SOC stock increase occurred between the first (70 t C ha^{-1}) and second sampling (84 t C ha^{-1}), and then a decrease to 79 t C ha^{-1} in 2000 (0–30 cm).
Switzerland (Fribourg Kanton) (Guillaume et al., 2021)	M	C, G	1987–2016	6		n.s. high	0–20	10 × 10	25	WO	n.s.	+	n.s.	All permanent grassland sites, which exhibited a significant trend over the 30 year time-period, revealed an SOC content increase (0.3 ± 0.04 mg C g^{-1} soil yr^{-1}).

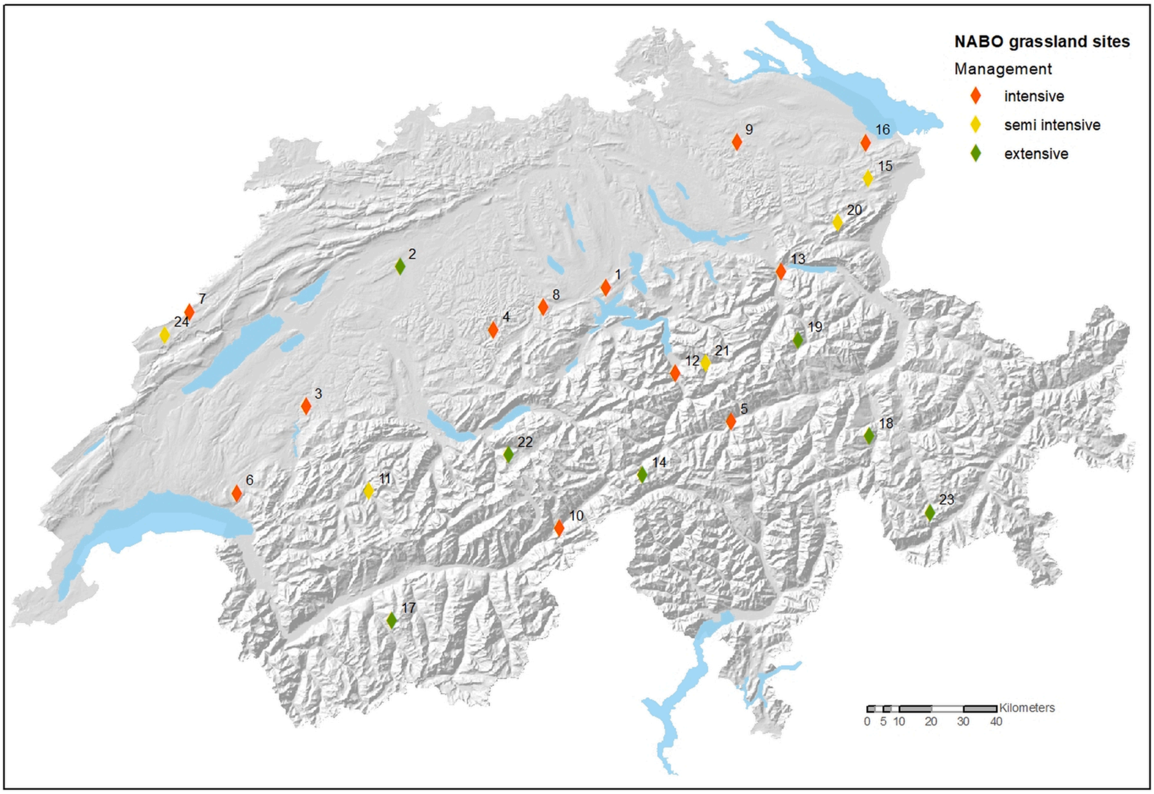

Fig. 1. Long-term monitoring sites (Swiss Soil Monitoring Network NABO) considered by the present study. Management intensities are indicated by colour.

Table 2
Meta-data and soil properties (topsoil, 0–20 cm depth) of each of the 24 grassland sites (mean of the six sampling campaigns and four replicates per site and campaign between 1985 and 2014) for altitude, SOC content, SOC stock, apparent density, pH, clay, silt, amount of applied solid and liquid manure, sum of annual precipitation, and average annual temperature (average 1985–2014). NA: no management data available.

site No.	altitude	SOC content	SOC stock	apparent density	pH	clay	silt	manure application (kg d.m. ha^{-1} year^{-1})		sum of annual precipitation	average annual temperature
	m a.s.l.	g C kg^{-1}	t C ha^{-1}	g cm^{-3}	($CaCl_2$)	%	%	solid manure	liquid manure	mm	°C
1	635	30.8	64.3	1.04	5.1	20	32	0	4378	1211	9.1
2	464	30.8	66.3	1.08	4.7	24	34	NA	NA	1112	9.2
3	735	31.0	71.7	1.15	5.3	19	35	724	2364	1177	8.6
4	998	35.7	63.6	0.89	4.4	18	52	1248	1684	1551	7.1
5	1105	36.6	63.7	0.87	5.2	12	37	NA	NA	1121	7.4
6	818	37.7	73.3	0.97	6.0	26	37	0	3762	1436	8.9
7	1093	38.5	77.9	1.01	5.6	28	49	2886	664	1556	5.4
8	955	42.3	80.2	0.95	5.4	17	26	0	2344	1465	7.3
9	537	42.5	87.8	1.03	6.4	35	34	70	3666	1150	8.9
10	1220	43.8	83.9	0.96	5.3	16	36	NA	NA	860	6.3
11	1030	44.7	82.4	0.92	6.0	46	32	NA	NA	1452	7.0
12	455	47.5	82.7	0.87	6.1	11	23	NA	NA	1220	9.9
13	431	47.6	77.4	0.81	6.1	33	55	2602	2811	1650	9.6
14	2120	48.6	80.9	0.83	3.8	9	28	NA	NA	1989	1.7
15	935	48.7	87.8	0.9	4.8	22	31	1165	450	1657	7.6
16	526	49.0	88.1	0.9	5.5	38	42	2165	3519	1148	9.4
17	2340	52.8	86.4	0.82	4.3	17	30	NA	NA	800	1.2
18	1818	53.7	91.7	0.85	4.5	22	38	NA	NA	1210	3.2
19	1880	54.3	72.7	0.67	4.3	27	38	NA	NA	1875	2.8
20	1338	54.7	83.5	0.76	5.4	27	26	NA	NA	2034	5.5
21	1100	56.4	86.7	0.77	4.7	33	27	2184	449	1778	6.8
22	1915	63.8	91.0	0.71	3.6	24	50	NA	NA	1624	2.8
23	2118	67.0	78.4	0.58	4.7	26	33	NA	NA	1203	1.0
24	1215	96.6	131.1	0.68	5.2	41	42	NA	NA	1571	5.1
Mean	1158	48.1	81.4	0.88	5.1	25	36	1186	2372	1410	6.3

taking sub-samples for SOC analyses. The remaining soil was archived for future analytical assessments. The whole process from sampling to lab analysis followed standard operation protocols.

The organic C content was determined either by the Swiss Standard Method (FAL, 1996) based on wet oxidation and re-titration of potassium dichromate, or by determining the total C content by means of dry

Table 3
Manure composition for the sites with management data. Animal types (cattle, pigs, sheep, and goats, ordered by the amount of phosphorus in manure applied from 1985 to 2017), median of annual nitrogen, phosphorus, and potassium inputs via manure on the parcel between 1985 and 2017 (Gross et al., 2021b), and mean SOC stocks in the soil.

Site No.	Manure types (animals)	Nitrogen (kg/ha/yr)	Phosphorus (kg/ha/yr)	Potassium (kg/ha/yr)	SOC stock (t C ha^{-1})
1	Cattle > pigs	242	59	271	64.3
3	Cattle	184	27	268	71.7
4	Cattle	160	38	222	63.6
6	Cattle > pigs	191	43	235	73.3
7	Cattle	113	25	167	77.9
8	Cattle > pigs	136	31	164	80.2
9	Cattle > pigs	160	34	219	87.8
13	Cattle > pigs	244	61	316	77.4
15	Cattle > sheep	43	10	64	87.8
16	Cattle > pigs	276	53	320	88.1
21	Cattle > sheep	90	21	121	86.7

Fig. 2. SOC content changes (difference of logarithm of SOC [%weight] and logarithm of SOC [%weight] of first sampling campaign for each site) of the six measuring campaigns between 1985 and 2014. The boxplots consist of 24 grassland sites with one exception (2010–2014, n = 23, site 22 is missing and was not included for the statistics). Different letters indicate significant differences (p-value < 0.05).

combustion (CN-analyser, LECO True Spec CN, year 2010) and subtracting the inorganic C-value where appropriate. The contents of inorganic C were determined by digestion with sulphuric acid and by the volumetric measurement of the produced CO_2 using a Scheibler-apparatus (FAL, 1996). The contents of the four sites containing inorganic C ranged between 0.6 g C kg^{-1} and 2.6 g kg^{-1} inorganic C (mean = 1.3 g C kg^{-1}, median = 1.2 g C kg^{-1}). Wet oxidation methods yield lower SOC content values compared with dry combustion and conversion factors to recalculate wet oxidation results to the level of the dry combustion method were used as described by Gubler et al. (2018).

2.3. Modelling SOC evolution using RothC

RothC is a widely used soil C model developed in the UK for crop systems approximately 30 years ago by Jenkinson et al. (1990), and it was further developed by Coleman et al. (1997). For simulations of SOC stock changes, we used the function RothC Model in the package SoilR (Sierra et al., 2012) in the software R (R core team 2020), modified by adding rate modifying factors to comply with the original RothC version 26.3 (Coleman and Jenkinson, 2008).

The model uses a monthly time step. The total SOC stock (total organic carbons stock, TOC) is split into five conceptual fractions: decomposable plant material (DPM), resistant plant material (RPM), microbial biomass (BIO), humified organic matter (HUM), and inert organic matter (IOM). IOM is resistant to decomposition and remains constant over time. Its amount is dependent on the total SOC based on the equation by Falloon et al. (1998), which is the standard method used by RothC if no radiocarbon measurements are available (Coleman and Jenkinson, 2008). The other compartments decompose by a first-order process, each with their own characteristic rate k.DPM = 10 yr^{-1}, k.RPM = 0.3 yr^{-1}, k.BIO = 0.66 yr^{-1}, k.HUM = 0.02 yr^{-1}. New C from plant residues is always added as DPM and RPM. For agricultural crops and improved grassland (e.g. pastures), C inputs are allocated to these two pools at a fixed ratio (DPM/RPM = 1.44, or 59% DPM and 41% RPM). Both DPM and RPM decompose to form CO_2, BIO and HUM. The proportion that goes to CO_2 or BIO/HUM is dependent on the clay content. Organic amendments, such as farmyard manure, are assumed to be partly decomposed, and 2% of SOC is presumed to be HUM while DPM and RPM each contribute 49%. Decomposition rates are affected by three factors; they increase with higher temperature and decrease if the soil is dry (high topsoil moisture deficit) or is covered by plants. Decomposition rates are also influenced by clay contents of the soil, with higher clay leading to lower decomposition rates. The initial pool distribution was calculated based on TOC stocks. For the IOM pool, the equation by Falloon et al. (1998) was applied.

$$IOM = 0.049 \text{ x } TOC^{1.139} \text{ (t ha}^{-1}) \quad (1)$$

For the calculation of RPM, BIO, and HUM, we used Weihermüller et al.'s (2013) simple pedotransfer functions. The equations also depend on TOC (t ha^{-1}) and the clay content in percentage mass.

$$RPM = (0.1847 \text{ x } TOC + 0.1555) \text{ x } (clay + 1.2750)^{-0.1158}, \quad (2)$$

$$HUM = (0.7148 \text{ x } TOC + 0.5069) \text{ x } (clay + 0.3421)^{0.0184}, \quad (3)$$

$$BIO = (0.0140 \text{ x } TOC + 0.0075) \text{ x } (clay + 8.8473)^{0.0567}. \quad (4)$$

The DPM pool is very small (0.2–1% of TOC for the long-term experiments) and turns over rapidly (Wüst-Galley et al., 2020). It was therefore assumed to be zero at the start of a simulation. The pool distribution based on the above equations was compared to equilibrium runs ("spin-up") for five Swiss long-term experiments. The correlation coefficient was 0.89 for the rather small RPM pool and 0.99 for the largest pool HUM and the tiny BIO pool (Wüst-Galley et al., 2020).

Where available (11 of the 24 sites), farmyard manure inputs reported by the farmers were used in the model (cf. Section 2.1). For the remaining sites, farmyard manure inputs of the same management intensity and production region (referred to as strata) were used as for SOC simulations in Switzerland's GHG inventory (Wüst-Galley et al., 2020). For the inventory, the amount of farmyard manure produced by the different animal categories was calculated and allocated to six grassland categories (intensively managed pasture, intensively managed meadow, less intensively managed meadow, extensively managed pasture, extensively managed meadow, summer pastures) as a function of farmer behaviour, the movement of cattle and other animals to summer pastures, and permissible fertilization rates. Within a simulation unit (stratum), manure was evenly distributed. To assess the impact of varying manure inputs, we ran additional RothC simulations, increasing manure inputs by 50% and 100%.

2.4. Statistical analyses

The calculations were done using the statistical software R (R core

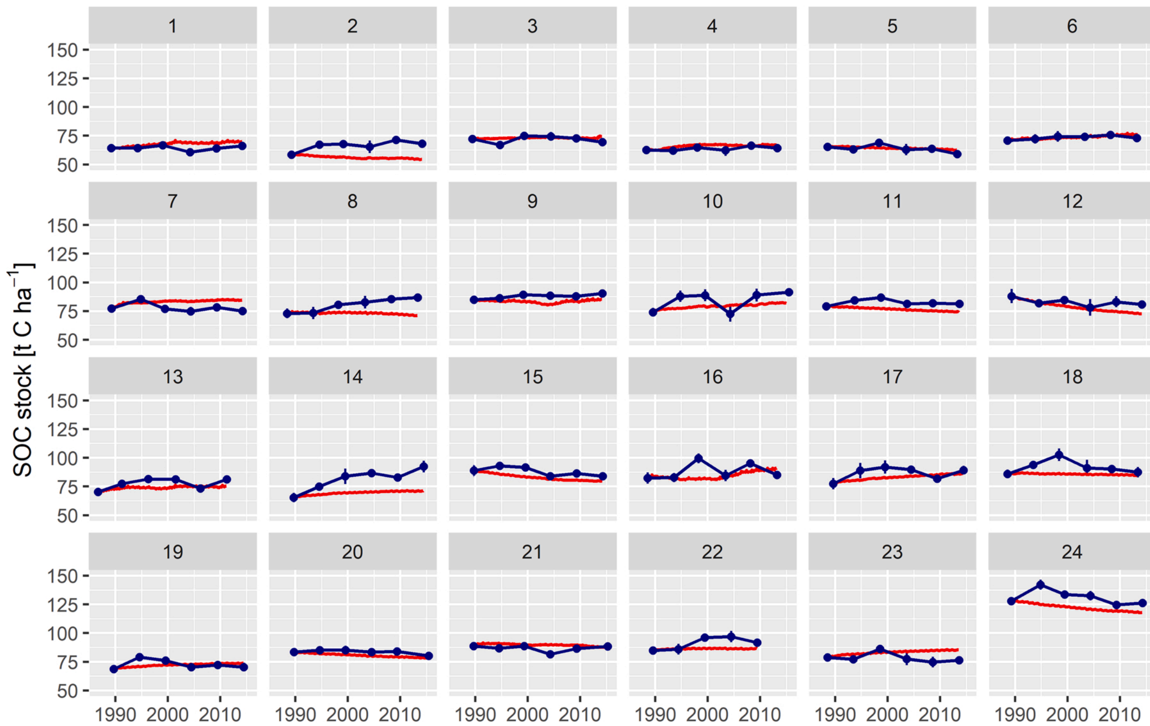

Fig. 3. SOC stock [t ha^{-1}] over time for the 24 grassland sites of the Swiss soil monitoring Network NABO. In blue: measured SOC contents recalculated to SOC stocks (see methods Section 2.4.2) and error bars (standard deviations of the four replicates) per sampling campaign. In red: modelled SOC stocks (RothC-Model see methods Section 2.3) with initial SOC-content equal to the measured one.

team 2020) in RStudio (© 2009–2022 RStudio, PCB). Meteorological grid data were obtained from MeteoSwiss (2021a). Daily data for precipitation, mean temperature, and sunshine duration covering the time-span from 1981 to 2018, as well as daily data for solar radiation covering the period from 1984 to 2018 were extracted from the Spatial Climate Analyses provided by the Federal Office of Meteorology and Climatology at a spatial resolution of 2 km x 2 km (MeteoSwiss, 2021a; MeteoSwiss, 2021b), considering the nearest grid points as representative for the site locations.

Solar radiation for the years 1981–1983 was estimated from sunshine duration following Rietveld (1978). In addition, daily evapotranspiration rates were calculated according to Priestley and Taylor (1972), assuming net radiation to be a linear function of solar radiation (Calanca, 2004; Davies, 1967).

As the Spatial Climate Analyses were developed relative to a mean topography (mean elevation within each of the 2 km x 2 km grid cells), the difference in elevation to the nearest grid point was considerable (> 250 m, in absolute) for some sites. This has implications in particular concerning temperature. In deriving monthly mean values we therefore corrected temperature using 4th-degree polynomials fitted, for each month, to the monthly climatic norms 1981–2010 provided by the Federal Office of Meteorology and Climatology for the use in the RothC model (MeteoSwiss, 2021b).

For the correlations with the measured data, temperature was adjusted for the difference in altitude between the NABO site and the closest grid point by −0.6 °C per 100 m higher altitude.

2.4.1. SOC evolution over all sites

To assess whether there is a general trend in SOC content evolution across Switzerland over time, differences of each sampling campaign to the first sampling campaign were assessed for each site. This was done based on log-transformed SOC contents for achieving a constant variance over the whole SOC range. If not stated otherwise, the mean of the four replicates per site and sampling was used.

To assess the general trend over all sites, log(SOC differences to first sampling campaign) were compared between the sampling campaigns using Friedman rank sum test (friedman.test), followed by a pairwise Wilcoxon rank sum post hoc-test (pairwise.wilcox.test) using Benjamini–Hochberg p-value adjustment for multiple testing.

Metadata (mean per site) were correlated with SOC using Spearman's rank correlation coefficients.

2.4.2. End-start comparison of SOC evolution

The measured SOC contents were converted to SOC stocks [t C ha^{-1}] by multiplying SOC content of fine earth [g kg^{-1}] by apparent density of fine earth (AD, mass of fine earth [< 2 mm] per total soil volume, g cm^{-3}) and 20 cm soil depth (Eq. 5).

$$SOC_{stock} = \left(\frac{SOC_{content}}{10}\right) * AD * 20 \quad cm. \tag{5}$$

For comparing the evolution of modelled and measured SOC stocks per site, the difference between the end and start of the time series were calculated for the measured data (Eq.6) and the modelled evolution (Eq.7).

$$d_{mes\ \ t2-t1} = (m_{mes\ \ t2} - m_{mes\ \ t1}) \ / \ (t2 - t1), \tag{6}$$

$$d_{mod\ \ t2-t1} = (m_{mod\ \ t2} - m_{mod\ \ t1}) \ / \ (t2 - t1), \tag{7}$$

$$Dif = d_{mes\ \ t2-t1} - d_{mod\ \ t2-t1} \tag{8}$$

Here, $m_{mes\ t2}$ is the mean of the second last and last measurement and $m_{mes\ t1}$ the mean of the first and second measurement per site. The differences for the modelled data were calculated similarly ($m_{mod\ t2}$ and $m_{mod\ t1}$, respectively). The means of the two first and two last measurements were used as a more robust value for the comparison. The differences were normalized by the difference in time t2 – t1 of the considered measurements. In analogy, t1 and t2 are defined as the means of the first and the second sampling and the mean of the last and second but last sampling, respectively.

The differences describing the temporal evolution measured and modelled, respectively, were compared by building their difference (Dif, Eq. 8). Minimal detectable change (MDC, Smith, 2004) was calculated using linear regression as described by Gubler et al. (2019), and there exists a relative MDC of 0.30% annually when considering 20 sites over 20 years and 0.25% per year when assessing 30 years. For the average SOC content of 48.1 g C kg^{-1} (Table 2) a change of 0.3% annually would mean an increase or decrease of 0.144 g C kg^{-1} annually. Calculating the SOC stock for topsoil (0–20 cm) according to Eq. 5 using an average AD of 0.88 g cm^{-3} (Table 2) and a SOC content of 0.144 g C kg^{-1} results in 0.25 t C ha^{-1} annually or 5 t C ha^{-1} over 20 years. We considered the threshold of ± 0.25 t C ha^{-1} yr^{-1} as relevant SOC change in the comparison between the end and start of the time series of measured and modelled data and looked closer at the sites with absolute differences bigger than 0.25 t C ha^{-1} yr^{-1}. Differences bigger than the arbitrary threshold of 0.25 t C ha^{-1} y^{-1} were assumed as "underestimated of the measured evolution by the RothC model" and differences smaller than − 0.25 t C ha^{-1} y^{-1} as "overestimated by the model". This comparison between measured and modelled data was also performed using the two scenarios in the RothC model with 50% and 100% higher manure input than in the original dataset (see Section 2.3).

Spearman's rank correlations of the mean values over all measuring campaigns per site were applied to compare SOC differences *Dif* with the metadata.

3. Results and discussion

3.1. Time-series of measured SOC-contents

We assessed the evolution of SOC-contents in the topsoil (upper 20 cm) of grasslands in Switzerland from 1985 to 2014 based on six sampling campaigns on a five year interval (Fig. 2). Roughly, SOC content of fine earth ranged from 30 to 100 g C kg^{-1} with a mean of 48.1 g C kg-1 and a median of 47.6 g C kg^{-1}. SOC stocks in the topsoil ranged from 63 to 131 t C ha^{-1} (mean = 81.7 t C ha^{-1}, median = 81.4 t C ha^{-1}). The pH values ranged from 3.6 to 6.4 (mean = 5.1, median = 5.2), whereas the fractions of clay and silt ranged from 9% to 46% (mean = 24.6%, median = 24.0%) and 23–55% (mean = 36.1%, median = 34.5%), respectively (Table 2).

Regarding the ensemble of 24 grassland sites, SOC contents in the topsoil remained constant with only slight differences between sampling campaigns (average SOC content per sampling campaign over all sites ranges from 46.1 g C kg^{-1} to 50.3 g C kg^{-1}; Fig. 2). SOC content changes (difference of logarithm of SOC [%weight] and logarithm of SOC [% weight] of first sampling campaign for each site) were highest for the sampling campaign from 1995 to 1999 and lowest for the very first sampling campaign. The Friedman test showed a significant difference (p-value = < 0.001), and the post-hoc test established a significant difference between the first and the third sampling campaign (p-value $_{\text{Benjamini Hochberg adjustment}}$ < 0.001). Thereby, site 22 has been excluded, because no data is available for the measuring campaign during 2010–2014. The slightly elevated SOC contents in the third sampling campaign are most likely an artefact, introduced either by difficult sampling conditions during this soil sampling or different time point of sampling within the year. There is no sound explanation for the observed increase within five years and the subsequent decline back to the original level. Therefore, the SOC content in all sites showed no trend over time. The highest outliers in sampling campaigns 2000–04, 2005–09, and 2010–2014 origin from the same site (No. 15). This site shows the highest increase in SOC stock over time. Management data of this site are not available, we cannot state whether this is related to changes in farm yard manure application.

Focussing on individual sites, the difference in SOC stocks between the end and start of the time series ($d_{mes\ t2\text{-}t1}$) of 6 of the 24 grassland sites (No. 2, 8, 10, 14, 16, and 22) were above the threshold of 0.25 t C ha^{-1} (see Section 2.4.2). While two sites (No. 15 and 24) revealed a difference smaller than the threshold of −0.25 t C ha^{-1} (Fig. 3 & Fig. 4, c.f. method Section 2.4.2). For the majority of sites (16) the difference between the end and the start of the time period was within the threshold of ± 0.25 t ha^{-1}, however, some fluctuations occur in between the sampling campaigns. These fluctuations between sampling campaigns might arise through the influence of weather conditions, manure applications, grazing intensities etc.

Thus, for the research question (i) we can summarize that the general SOC-content evolution over all sites is stable over the investigated time period between 1985 and 2014. However, at individual sites temporal trends occurred: 8% of the sites revealed a negative SOC-stock trend, while at 25% of the sites SOC-stock increased over time.

For assessing which environmental and farm management factors influence temporal trends (research question ii), Spearman correlations using the available meta-data for each site were analysed (SI Fig. 1). SOC content was found to be positively correlated with the altitude of the site, the apparent density of the soil, and the C:N ratio. These findings are in accordance with expectations because at higher altitude a colder and wetter climate prevails which favours SOC accumulation, apparent density is directly influenced by SOC and clay contents, and C:N ratios show how well degradable SOC is. Neither pH, clay- and silt-content, ratio of SOC to clay content, manure (liquid and solid) application rates, nor mean precipitation were found to be significantly correlating with SOC contents and stocks. This contrasts samples obtained from Swiss arable land inside the NABO network, where SOC content was correlated with pH, clay content, manure input, altitude, and percentage of temporary grassland in the crop rotation (Gubler et al., 2019). In the case of manure application data was only available for 11 out of the 24 here presented grassland sites while in the study by Gubler et al. (2019) the information was available for more sites (24 out of 30).

Farming management plays a major role in maintaining SOC stocks. On one hand, manure application increases organic matter on the fields and thus also has a positive influence on SOC stocks as shown by Maillard and Angers (2014). In their study, the amount of manure applied, and application of manure instead of mineral fertilizer played the major role for relative SOC changes in a worldwide meta-study comparing unfertilized or mineral fertilized fields (both grassland and cropland) with fields getting manure applications (Maillard and Angers, 2014). Thereby the source of the manure (cattle, pig, farmyard-manure),

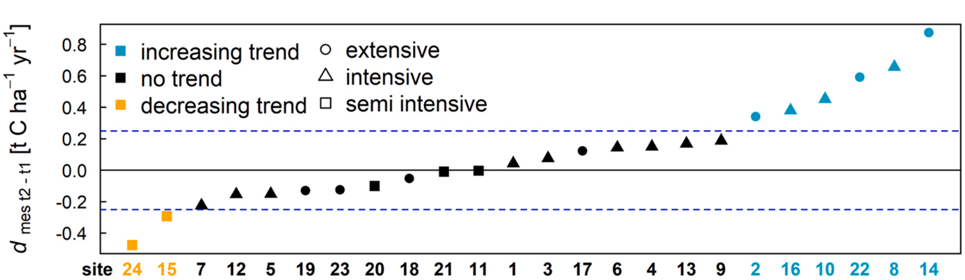

Fig. 4. SOC stock difference $d_{mes\ t2\text{-}t1}$, calculated based on the measured SOC contents (see method Section 2.4.2 Eq. 5 & 6), between the end and the beginning of the time-series divided by the years between start and end (i.e., 20 yr.). Blue dashed lines indicate threshold line for an increase of 5 t C ha^{-1} over 20 years or 0.25 t C ha^{-1} annually.

tillage, initial SOC content, clay content, pH and climate zone had an influence on if SOC stock increases by manure application (Gross and Glaser, 2021). However, potential C saturation should also be considered in this context (Gross and Glaser, 2021). In the presented long-term data no correlation between SOC stock evolution ($d_{mes\ t2-t1}$) and pH, clay content and manure application could be detected. Additionally, for most of the sites with a SOC stock trend, management data were unavailable and it is not known if there was a change in manure application over the investigated time.

The six sites with SOC stock increases higher than 0.25 t ha^{-1} established an increase between 0.34 and 0.87 t ha yr^{-1} (Fig. 4). The sites did not have much in common and covered the three management intensities extensive, semi intensive, and intensive and altitudes between 460 and 2120 m a.s.l. No pattern in pH (3.6–5.5), clay content (9–38%), precipitation (sum per year: 860–1989 mm) or C:N ratio (8.0–12.8) could be detected for these six sites (Table 2).

Site 24 revealed a relevant decrease of − 0.48 t C ha^{-1} yr^{-1}, whereas site 15 showed a relevant decrease of 0.29 t C ha^{-1} yr^{-1}. These two sites are semi intensively managed. The meta-data (mean over all time-points) of these two sites did not establish any pattern which could explain the negative trends (Table 2). However, for site 15, manure input decreased over time with much higher manure inputs in the 1980 s than from the 1990 s, which can also explain the decrease in SOC. In addition, for site 7, which established a decrease in SOC slightly smaller than our threshold of 0.25 t ha^{-1} yr^{-1}, management data show the same pattern with decreasing manure inputs (SI Fig. 2). For site 24, management data were unavailable. Comparable sites (management and clay content) in the GHG inventory established decreasing SOC stock trends in the simulations (FOEN, 2021).

No general trend of SOC content evolution of the Swiss national monitoring grassland sites was found here. This is in accordance with results for arable sites of the Swiss Soil Monitoring Network for the period between 1990 and 2014 (Gubler et al., 2019). However, in other European countries the SOC evolution in grasslands is inconsistent: increasing, decreasing, or no trends were reported, when focussing on the general trend of all assessed sites (Table 1). Also, at single sites, and at regional scales, trends can be observed. Similar to our findings, various European studies showed no general trend, but trends for some individual sites and the bigger part of these sites are showing an increasing than a decreasing trend (Table 1). Such trends may be explained by changing management practices, the timespan and analytical protocols of the soil observation and climatic conditions.

An increasing trend of SOC evolution was reported for the two Belgian regions (Flanders and Wallonia), and in whole Belgium between 1950 and 2000 (Lettens et al., 2005b). However, those regions of Belgium simultaneously established a decrease in SOC stocks when focussing on the time span between 1990 and 2000 (Lettens et al., 2005a). According to them, changes in manure application based on new regulations might have an impact on the losses but also sampling and analytical issues were discussed as possible reasons. They highlighted for example: 1) the importance of precise sampling at the same location over the years; 2) potential uncertainties from estimating site parameters, such as bulk density instead of measuring them; 3) the difficulties of using the same analytical methods over long timeframes (decades); and 4) the issues which can occur by combining different datasets of various origin. The data presented in our study is based on soils sampled from the same locations during each sampling campaign, and meta-data (e.g., apparent density, pH, annual precipitation) were measured; hence some of the aforementioned uncertainties could be avoided. We decided to use the start-end difference method (c.f. Section 2.4.2) instead of a linear model to also gradually cover non-linear changes.

The climate, seasonal patterns as well as the weather conditions can influence SOC decomposition and accumulation respectively. Under arid climates, soils hold less SOC as organic soils in colder regions (Lal, 2004b). Seasonal variability of SOC also occurs within the course of the year (Scheidung et al., 2017; Wuest, 2014). Climate change is expected to influence SOC decomposition and accumulation in soils strongly. Different modelling approaches have been used to estimate the impact of climate change. For example, RothC was used to model SOC-stocks for Bavaria under different climate and C-input scenarios predicting decreasing SOC-stocks for the different scenarios between 2000 and 2095 (Wiesmeier et al., 2016). Focusing on measured monitoring data, the Bavarian soil monitoring (Kühnel et al., 2019), assessed the time span between 1989 and 2016 (five sampling campaigns) and determined explanatory variables for variations in SOC stock trends using a random forest model. Thereby, the main part of the variation (73%) in the measured time series could be explained by changes in autumn precipitation, spring soil moisture, change of spring soil moisture, change of summer precipitation, and organic fertilizer (Kühnel et al., 2019). Therefore, seasonal changes in temperature and precipitation play an important role. To assess if similar patterns are found in our data, we correlated average winter (Jan–Mar), spring (Apr–June), summer (July–Sept) and autumn (Oct–Dec) precipitation and temperature of the year previous to the sampling (for example SOC sampling in spring 2004 correlated with autumn precipitation in 2003) obtained from the grid data provided by MeteoSwiss (2021a), temperature adjusted for altitude of the respective site location) with SOC content, stock and evolution (end-start method) (Figures SI 4–7). Thereby the difference in seasonal precipitation from 1985 to 2000 and 2005–2014 was also assessed; however, no significant correlations with SOC content and stock were detected (Figure SI 4). The average SOC contents per site were found to be significantly positively correlated with summer precipitation (Rho = 0.41, $p < 0.05$), however, no correlations could be found between seasonal precipitations and SOC stock evolution. Temperature was not considered because of its strong correlation with the altitude of the investigated sites.

For research question (ii) it can be summarized, that from the assessed environmental and farming management factors, SOC-contents were mainly influenced by the altitude of the sampling site, the apparent density and the C:N ratio of the soil. Sites with decreasing SOC-stock trends most probably received more manure in the 1980ies than later in the investigated time-period. Thus, manure input has an influence on SOC stocks, even tough, our measured SOC-stocks were not significantly correlated with the manure inputs, which can also be influenced by the fact, that manure applications are only available for a subset of the sites.

3.2. SOC stocks predicted fairly well by RothC

Comparing the evolution from 1985 to 2014 of SOC stocks measured and those modelled using RothC for each of the 24 sites, we observe generally a higher variability over time for the measured data compared with the modelled ones (Fig. 3). This was expected due to the variability introduced by the soil sampling and sample preparation, as well as seasonal fluctuations affecting the measured data. For 16 out of the total 24 sites, the difference of modelled and measured evolution of the SOC stock (see method Section 2.4.2 Eq. 8) was situated between the thresholds of ± 0.25 t C ha^{-1} yr^{-1} and is assumed as good fit (Fig. 5). These 16 sites included sites with positive, negative and no trend in SOC stock over time in the measured data (Fig. 4). This outcome indicates for research question (iii) that the model provides reasonable predictions, even though many variables, such as C-uptake by plants and manure inputs of some sites are based on estimates and were not based on site-specific measurements. More specifically, plant-derived C-inputs were assumed to be constant and manure-derived C inputs were either based on manure application amounts obtained from the farmers managing the grassland sites or based on estimates that apply to much larger regions (at sites where no agricultural management data were available). It is worth noting that six out of the eight sites with a deviation of more than 0.25 t C ha^{-1} yr^{-1} were sites where management data were not known.

For sites 10 and 16, the increasing trend in measured SOC stocks (dif

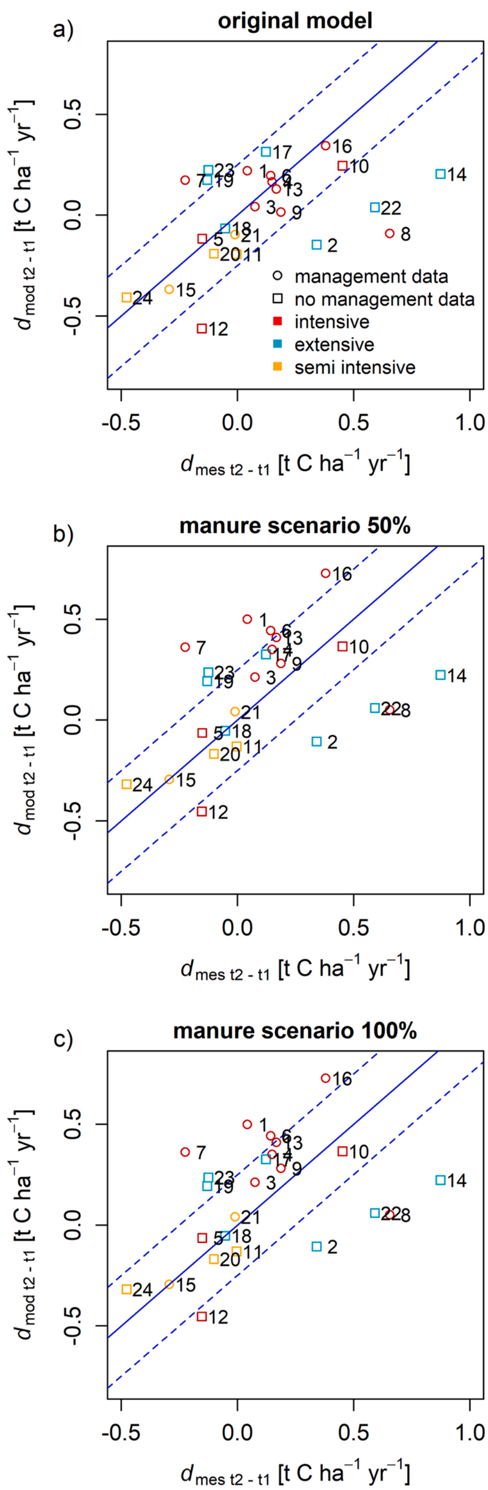

Fig. 5. Difference of the end and the start of the time series of measured ($d_{mes\ t2–t1}$) and modelled data ($d_{mod\ t2–t1}$) divided through the years between start and end. In red, orange, and turquoise are the management intensities intensive, semi intensive, and extensive, respectively, indicated. Squares stand for sites without detailed management data and points stand for sites with known amount of manure application. The blue solid line indicates equal values of modelled and measured data and the dashed blue lines indicate the threshold line for increases and decreases of 5 t C ha^{-1} soil over a time span of 20 years or 0.25 t C ha^{-1} annually, which is assumed as relevant change (see chapter 2.4.2).

$_{mes\ t2\text{-}t1}$ > 0.25 t ha^{-1}) was predicted properly by the model (dif $_{mod\ t2\text{-}t1}$), and for sites 15 and 24, expressing a negative trend in the measured data (dif $_{mes\ t2\text{-}t1}$ < 0.25 t ha^{-1}); RothC detected the negative trend over time as indicated in Fig. 5 with these sites laying between the threshold of ± 0.25 t C ha^{-1} yr^{-1}. Focussing on sites of which the differences between the measured and modelled data (*Dif*) are outside the ± 0.25 t C ha^{-1} yr^{-1} threshold, for 5 of the 24 sites, the model estimated a more negative evolution of SOC than observed for the measured values (Fig. 5). Conversely, the model indicated more positive evolution than actually measured for three sites. For the simulations we assume that initial SOC stocks (in starting year) are at equilibrium. In reality, SOC stocks might follow a trend induced through a former land-use change (e.g. conversion from cropland to permanent grassland). Such changes can affect SOC stocks for 20–30 years (Poeplau et al., 2011). For the here presented sites, no land-use change occurred since the start of the measurements in 1985. Sites 2, 8, 14, and 22 with increasing trend form the end-start method of the measured data were not well predicted by the RothC model (Fig. 4, Fig. 5). For all of these sites the increasing SOC trend is not visible in the modelled data; however, management information about manure applications are not available for these sites and were based on estimations according to the methods of the Swiss GHG Inventory (FOEN, 2021). For sites without management data, an even distribution of the liquid and solid manure is assumed for the whole area. However, due to animal movement, practical issues on spreading of manure, e.g. distance between the stable and the field, or automation of manure distribution, local inhomogeneous C-inputs, might occur. In the RothC modelling according to the Swiss GHG inventory, C-inputs from plants are assumed to be constant because modelled inputs based on allometric functions for grasslands establish extremely high C-inputs (Keel et al., 2017). This might further explain the differences between the modelled and measured data.

Four out of the eight sites with no good model fit (Fig. 5) are located above 1800 m a.s.l. At this altitude, soils are often shallow and contain more soil skeleton compared with soils located at lower altitude. This makes the sampling more difficult and leads to an increased variability in the measured parameters, as e.g., SOC content and apparent density (SI Table 1). The mean of the SOC content standard deviations of the four replicates annually and site is higher for sites at elevations above 1800 m a.s.l. (average over all sites in category higher than 1800 m a.s. l.: mean 0.26, min. 0.15, max. 0.4) than for lower located sites (average over all sites in category lower than 1800 m a.s.l.: mean 0.15, min. 0.07, max. 0.39) (Table SI 1, Figure SI 9). Microclimate might play a more important role at these alpine sites: for example snow valleys with longer snow coverage, can have a strong impact on the microclimate of the site and might be different than the estimations based on the closest point of the grid data by MeteoSwiss. Therefore, alpine sites are much more difficult for estimating the model parameters.

Grazing is known to influence SOC stocks and GHG emissions as reviewed by Wang et al. (2021). By modelling grassland GHG fluxes between 1750 and 2012, Chang et al. (2021) showed that emissions of N_2O and CH_4 from grasslands have increased by a factor of 2.5, mainly due to intensification of management with increasing numbers of cattle, and higher manure and mineral fertilizer input. For eleven grassland sites in our study management data are known for the specific parcel of land, thus the data on manure inputs were expected to be very good for these instances. However, one of these sites (No. 8, Fig. 5) shows a very poor agreement between measured and modelled data. The actual inputs of manure seem to be much higher than indicated by the management data, and therefore, it is important to validate the data provided by the farmers as well.

To verify this hypothesis and also check whether estimated C-input was too low, we ran the RothC model again with 50%, and 100% higher C-inputs originating from manure application (Fig. 5 & Figure SI 8). Differences between the scenarios for the sites with increasing trend were marginal, showing no additional sites with good fitting (within ± 0.25 t C ha^{-1} yr^{-1} threshold) model-values for both scenarios (Fig. 5).

However, three sites (No. 1, 6, and 16) established a worse fit than in the original model while the other sites stayed in the same category as in the original model (Fig. 5 and Figure SI 8). Interestingly, there are a lot of sites where the additional manure input did not show a big effect on the modelled evolution of SOC stocks in the topsoil. This might be explained by the fact, that for these sites a low C input through manure was assumed as these values were derived from national scale simulations for the GHG inventory and twice this small amount is still very low compared to intensive managed sites. However, three of the ten cases, where only minimal differences occurred between the different manure scenarios, were semi-intensively or intensively managed. Therefore, other factors might play a role for small effects of additional manure inputs.

Considering the characteristics of the sites further, which showed smaller SOC trends by modelling than the measured data, no significant Spearman-correlations could be found between the differences between measured and modelled data (*Dif*) and soil and site properties (SI Fig. 1). For Belgium, measured SOC stocks from two sampling campaigns in the 1960 s and 2006 were compared to RothC modelled data for different landscape units including arable land and grassland (Van Wesemael et al., 2010). Modelled and measured data were significantly ($p < 0.05$) but not strongly correlated ($r^2 = 0.337$) and the outliers were mainly from sites, where the biggest changes in SOC stock occurred (sites with decreasing and increasing trends). It was accentuated that other factors additional to manure and plant C input should be investigated (Van Wesemael et al., 2010). These findings are similar to the presented comparison between RothC and the measured data of NABO grassland sites.

For the Swiss national GHG inventory, which also uses RothC, it is important to cover the general trend of different regions and not for single sites. Comparing the modelled data with the measured data of the NABO grassland sites (research question (iii)), in most of the cases we found a good fit and differences between the modelled and measured SOC evolutions were within the threshold of ± 0.25 t C ha^{-1} yr^{-1}. Regarding research question (iv), the main gap for accurate data lays in the higher located and especially alpine grasslands. By documenting and measuring manure inputs and microclimatic conditions more closely for these sites, an improvement could be achieved. Most of the outliers of the model-measurement comparison were underestimating SOC increases and management data were unavailable for these sites. This is most probably due to the fact that if no site-specific data were available, the manure inputs used for RothC modelling were derived from a national scale SOC inventory that assumes equal distribution of manure over large areas whereas in reality different amounts are distributed on single parcels of land. There are also cases, where the RothC model showed higher SOC values than the measured data, and thus balanced out the general trend, which is of importance for the GHG inventory, and not the single plot perspective. However, it needs to be tested, if this is also true for arable land. Accurate management data and initial SOC stocks are crucial for obtaining reliable modelled data. This is especially true for point simulations, as we performed in the current study, but also holds for large scale applications as in the GHG inventory.

4. Conclusions

While the general evolution over all NABO grassland sites was stable and no changes in SOC contents were observed from 1985 to 2014, single sites revealed positive or negative trends, when changes outside of the thresholds ± 0.25 t ha^{-1} yr^{-1} are considered as relevant changes. The stable values of SOC stocks might indicate that the farming management practices used in Switzerland are well suited to maintain SOC stocks in mineral topsoils, which is of great importance regarding climate change.

RothC predicted the general long-term trend of SOC stock-evolution per site quite well (66% of the cases), although site specific management data was only partly available. However, RothC showed a tendency to underestimate increasing trends and could not capture short term fluctuations between the sampling campaigns (five year intervals). Accurate input values of initial SOC content, meteorological data, as well as manure input data are of high importance. Continuing the long-term SOC measurements and further assessing SOC stock changes by modelling based on the measured data can help to better understand whether soils under grasslands influenced by climate change can function as sink for atmospheric carbon in the future. Future studies might, for example, focus on grasslands at higher altitudes (such as alpine pastures) to reduce the knowledge gaps revealed by our study.

Declaration of Competing Interest

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

Data Availability

Data will be made available on request by signing a usage agreement.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.agee.2022.108258.

References

Aksoy, E., Yigini, Y., Montanarella, L., 2016. Combining soil databases for topsoil organic carbon mapping in Europe. PLoS One 11, 1–17. https://doi.org/10.1371/journal.pone.0152098.
Batjes, N.H., 1996. Total carbon and nitrogen in the soils of the world. Eur. J. Soil Sci. 47, 151–163. https://doi.org/10.1111/j.1365-2389.1996.tb01386.x.
Calanca, P., 2004. Interannual variability of summer mean soil moisture conditions in Switzerland during the 20th century: a look using a stochastic soil moisture model. Water Resour. Res. 40, 1–9. https://doi.org/10.1029/2004WR003254.
Capriel, P., 2013. Trends in organic carbon and nitrogen contents in agricultural soils in Bavaria (south Germany) between 1986 and 2007. Eur. J. Soil Sci. 64, 445–454. https://doi.org/10.1111/ejss.12054.
Chang, J., Ciais, P., Gasser, T., Smith, P., Herrero, M., Havlík, P., Obersteiner, M., Guenet, B., Goll, D.S., Li, W., Naipal, V., Peng, S., Qiu, C., Tian, H., Viovy, N., Yue, C., Zhu, D., 2021. Climate warming from managed grasslands cancels the cooling effect of carbon sinks in sparsely grazed and natural grasslands. Nat. Commun. 12, 1–10. https://doi.org/10.1038/s41467-020-20406-7.
Chapman, S.J., Bell, J.S., Campbell, C.D., Hudson, G., Lilly, A., Nolan, A.J., Robertson, A. H.J., Potts, J.M., Towers, W., 2013. Comparison of soil carbon stocks in Scottish soils between 1978 and 2009. Eur. J. Soil Sci. 64, 455–465. https://doi.org/10.1111/ejss.12041.
Chartin, C., Stevens, A., Goidts, E., Krüger, I., Carnol, M., van Wesemael, B., 2017. Mapping soil organic carbon stocks and estimating uncertainties at the regional scale following a legacy sampling strategy (Southern Belgium, Wallonia). Geoderma Reg. 9, 73–86. https://doi.org/10.1016/j.geodrs.2016.12.006.
Coleman, K., Jenkinson, D.S., 2008. RothC—A model for the turnover of carbon in soil Model—Model description and users guide.
Coleman, K., Jenkinson, D.S., Crocker, G.J., Grace, P.R., Klír, J., Körschens, M., Poulton, P.R., Richter, D.D., 1997. Simulating trends in soil organic carbon in long-term experiments using RothC-26.3. Geoderma 81, 29–44. https://doi.org/10.1016/S0016-7061(97)00079-7.

Davies, J.A., 1967. A note on the relationship between net radiation and solar radiation. Q. J. R. Meteorol. Soc. 93, 109–155. https://doi.org/10.1002/qj.49709339511.
Don, A., Schumacher, J., Freibauer, A., 2011. Impact of tropical land-use change on soil organic carbon stocks - a meta-analysis. Glob. Chang. Biol. 17, 1658–1670. https://doi.org/10.1111/j.1365-2486.2010.02336.x.
Ellili, Y., Walter, C., Michot, D., Pichelin, P., Lemercier, B., 2019. Mapping soil organic carbon stock change by soil monitoring and digital soil mapping at the landscape scale. Geoderma 351, 1–8. https://doi.org/10.1016/j.geoderma.2019.03.005.
FAL, 1996. Referenzmethoden der Eidg. landwirtschaftlichen Forschungsanstalten. Zürich-Reckenholz. FAL.
Falloon, P., Smith, P., Coleman, K., Marshall, S., 1998. Estimating the size of the inert organic matter pool from total soil organic carbon content for use in the Rothamsted carbon model. Soil Biol. Biochem. 30, 1207–1211. https://doi.org/10.1016/S0038-0717(97)00256-3.
FAO, ITPS, GSBI, CBD, EC, 2020. FAO, et al. "State of Knowledge of Soil Biodiversity–Status, Challenges and Potentialities, Report 2020. Rome.
Fliessbach, A., Oberholzer, H.R., Gunst, L., Mäder, P., 2007. Soil organic matter and biological soil quality indicators after 21 years of organic and conventional farming. Agric. Ecosyst. Environ. 118, 273–284. https://doi.org/10.1016/j.agee.2006.05.022.
FOEN, 2021. Switzerland ' s Greenhouse Gas Inventory 1990–2019: National Inventory Report and reporting tables (CRF), Submission of April 2021 under the United Nations Framework Convention on Climate Change and under the Kyoto Protocol. Bern.
Friedlingstein, P., O'Sullivan, M., Jones, M.W., Andrew, R.M., Hauck, J., Olsen, A., Peters, G.P., Peters, W., Pongratz, J., Sitch, S., Le Quéré, C., Canadell, J.G., Ciais, P., Jackson, R.B., Alin, S., Aragão, L.E.O.C., Arneth, A., Arora, V., Bates, N.R., Becker, M., Benoit-Cattin, A., Bittig, H.C., Bopp, L., Bultan, S., Chandra, N., Chevallier, F., Chini, L.P., Evans, W., Florentie, L., Forster, P.M., Gasser, T., Gehlen, M., Gilfillan, D., Gkritzalis, T., Gregor, L., Gruber, N., Harris, I., Hartung, K., Haverd, V., Houghton, R.A., Ilyina, T., Jain, A.K., Joetzjer, E., Kadono, K., Kato, E., Kitidis, V., Korsbakken, J.I., Landschützer, P., Lefèvre, N., Lenton, A., Lienert, S., Liu, Z., Lombardozzi, D., Marland, G., Metzl, N., Munro, D.R., Nabel, J.E.M.S., Nakaoka, S.I., Niwa, Y., O'Brien, K., Ono, T., Palmer, P.I., Pierrot, D., Poulter, B., Resplandy, L., Robertson, E., Rödenbeck, C., Schwinger, J., Séférian, R., Skjelvan, I., Smith, A.J.P., Sutton, A.J., Tanhua, T., Tans, P.P., Tian, H., Tilbrook, B., Van Der Werf, G., Vuichard, N., Walker, A.P., Wanninkhof, R., Watson, A.J., Willis, D., Wiltshire, A.J., Yuan, W., Yue, X., Zaehle, S., 2020. Global carbon budget 2020. Earth Syst. Sci. Data 12, 3269–3340. https://doi.org/10.5194/essd-12-3269-2020.
Goidts, E., van Wesemael, B., 2007. Regional assessment of soil organic carbon changes under agriculture in Southern Belgium (1955-2005). Geoderma 141, 341–354. https://doi.org/10.1016/j.geoderma.2007.06.013.
Gross, A., Glaser, B., 2021. Meta-analysis on how manure application changes soil organic carbon storage. Sci. Rep. 11, 1–13. https://doi.org/10.1038/s41598-021-82739-7.
Gross, T., Müller, M., Keller, A., Gubler, A., 2021a. Erfassung der Bewirtschaftungsdaten im Messnetz der Nationalen Bodenbeobachtung NABO. Agroscope Sci. 122 (1), 51.
Gross, T., Keller, A., Müller, M., Gubler, A., 2021b. Stoffbilanzen für Parzellen der Nationalen Bodenbeobachgung. Agroscope Sci. 123, 1:99.Gubler, A., Wächter, D., Blum, F., Bucheli, T.D., 2015. Remarkably constant PAH concentrations in Swiss soils over the last 30 years. Environ. Sci. Process. Impacts 17, 1816–1828. https://doi.org/10.1039/c5em00344j.
Gubler, A., Wächter, D., Schwab, P., 2018. Homogenisation of series of soil organic carbon: harmonising results by wet oxidation (Swiss Standard Method) and dry combustion. Agro. Sci 62.
Gubler, A., Wächter, D., Schwab, P., Müller, M., Keller, A., 2019. Twenty-five years of observations of soil organic carbon in Swiss croplands showing stability overall but with some divergent trends. Environ. Monit. Assess. 191. https://doi.org/10.1007/s10661-019-7435-y.
Guillaume, T., Bragazza, L., Levasseur, C., Libohova, Z., Sinaj, S., 2021. Long-term soil organic carbon dynamics in temperate cropland-grassland systems. Agric. Ecosyst. Environ. 305. https://doi.org/10.1016/j.agee.2020.107184.
Hanegraaf, M.C., Hoffland, E., Kuikman, P.J., Brussaard, L., 2009. Trends in soil organic matter contents in Dutch grasslands and maize fields on sandy soils. Eur. J. Soil Sci. 60, 213–222. https://doi.org/10.1111/j.1365-2389.2008.01115.x.
Hopkins, D.W., Waite, I.S., McNicol, J.W., Poulton, P.R., Macdonald, A.J., O'donnell, A. G., 2009. Soil organic carbon contents in long-term experimental grassland plots in the UK (Palace Leas and Park Grass) have not changed consistently in recent decades. Glob. Chang. Biol 15, 1739–1754. https://doi.org/10.1111/j.1365-2486.2008.01809.x.
Jansson, J.K., Hofmockel, K.S., 2020. Soil microbiomes and climate change. Nat. Rev. Microbiol. 18, 35–46. https://doi.org/10.1038/s41579-019-0265-7.
Jenkinson, D.S., Lynch, J., Goss, M.J., Tinker, P.B., 1990. The turnover of organic matter in soil. Philos. Trans. R. Soc. B-Biol. Sci. 329, 361–368.
Jobbágy, E.G., Jackson, R.B., 2000. The vertical distribution of soil organic carbon and its relation to climate and vegetation. Ecol. Appl. 10, 423–436 https://doi.org/10.1890/1051-0761(2000)010[0423:TVDOSO]2.0.CO;2.
Johnston, A.E., Poulton, P.R., Coleman, K., 2009. Chapter 1 Soil Organic Matter. Its Importance in Sustainable Agriculture and Carbon Dioxide Fluxes, 1st ed, Advances in Agronomy. Elsevier Inc. https://doi.org/10.1016/S0065–2113(08)00801–8.
Keel, S.G., Leifeld, J., Mayer, J., Taghizadeh-Toosi, A., Olesen, J.E., 2017. Large uncertainty in soil carbon modelling related to method of calculation of plant carbon input in agricultural systems. Eur. J. Soil Sci. 68, 953–963. https://doi.org/10.1111/ejss.12454.
Keel, S.G., Anken, T., Büchi, L., Chervet, A., Fliessbach, A., Flisch, R., Huguenin-Elie, O., Mäder, P., Mayer, J., Sinaj, S., Sturny, W., Wüst-Galley, C., Zihlmann, U., Leifeld, J., 2019. Loss of soil organic carbon in Swiss long-term agricultural experiments over a wide range of management practices. Agric. Ecosyst. Environ. 286, 106654 https://doi.org/10.1016/j.agee.2019.106654.
Knotters, M., Teuling, K., Reijneveld, A., Lesschen, J.P., Kuikman, P., 2022. Changes in organic matter contents and carbon stocks in Dutch soils, 1998–2018. Geoderma 414. https://doi.org/10.1016/j.geoderma.2022.115751.
Kühnel, A., Garcia-Franco, N., Wiesmeier, M., Burmeister, J., Hobley, E., Kiese, R., Dannenmann, M., Kögel-Knabner, I., 2019. Controlling factors of carbon dynamics in grassland soils of Bavaria between 1989 and 2016. Agric. Ecosyst. Environ. 280, 118–128. https://doi.org/10.1016/j.agee.2019.04.036.
Lal, R., 2004a. Agricultural activities and the global carbon cycle. Nutr. Cycl. Agroecosyst. 70, 103–116. https://doi.org/10.1023/B:FRES.0000048480.24274.0f.
Lal, R., 2004b. Soil carbon sequestration impacts on global climate change and food security. Science 304 (80), 1623–1627. https://doi.org/10.1126/science.1097396.
Lal, R., 2008. Carbon sequestration. Philos. Trans. R. Soc. B 363, 815–830.
Lal, R., 2021. Negative emission farming. J. Soil Water Conserv 76, 61A–64A. https://doi.org/10.2489/jswc.2021.0419A.
Lettens, S., Van Orshoven, J., Van Wesemael, B., De Vos, B., Muys, B., 2005a. Stocks and fluxes of soil organic carbon for landscape units in Belgium derived from heterogeneous data sets for 1990 and 2000. Geoderma 127, 11–23. https://doi.org/10.1016/j.geoderma.2004.11.001.
Lettens, S., Van Orshoven, J., Van Wesemael, B., Muys, B., Perrin, D., 2005b. Soil organic carbon changes in landscape units of Belgium between 1960 and 2000 with reference to 1990. Glob. Chang. Biol. 11, 2128–2140. https://doi.org/10.1111/j.1365-2486.2005.001074.x.
Lugato, E., Panagos, P., Bampa, F., Jones, A., Montanarella, L., 2014. A new baseline of organic carbon stock in European agricultural soils using a modelling approach. Glob. Chang. Biol. 20, 313–326. https://doi.org/10.1111/gcb.12292.
Maillard, É., Angers, D.A., 2014. Animal manure application and soil organic carbon stocks: a meta-analysis. Glob. Chang. Biol. 20, 666–679. https://doi.org/10.1111/gcb.12438.
Martin, M.P., Wattenbach, M., Smith, P., Meersmans, J., Jolivet, C., Boulonne, L., Arrouays, D., 2011. Spatial distribution of soil organic carbon stocks in France. Biogeosciences 8, 1053–1065. https://doi.org/10.5194/bg-8-1053-2011.
Martin, M.P., Orton, T.G., Lacarce, E., Meersmans, J., Saby, N.P.A., Paroissien, J.B., Jolivet, C., Boulonne, L., Arrouays, D., 2014. Evaluation of modelling approaches for predicting the spatial distribution of soil organic carbon stocks at the national scale. Geoderma 223–225, 97–107. https://doi.org/10.1016/j.geoderma.2014.01.005.
Meersmans, J., Van Wesemael, B., De Ridder, F., Dotti, M.F., De Baets, S., Van Molle, M., 2009. Changes in organic carbon distribution with depth in agricultural soils in northern Belgium, 1960-2006. Glob. Chang. Biol. 15, 2739–2750. https://doi.org/10.1111/j.1365-2486.2009.01855.x.
Meersmans, J., Van Wesemael, B., Goidts, E., Van Molle, M., De Baets, S., De Ridder, F., 2011. Spatial analysis of soil organic carbon evolution in Belgian croplands and grasslands, 1960-2006. Glob. Chang. Biol. 17, 466–479. https://doi.org/10.1111/j.1365-2486.2010.02183.x.
MeteoSwiss, (Federal Office of Meteorology and Climatology), 2021a. Spatial Climate Analyses [WWW Document]. URL https://www.meteoswiss.admin.ch/home/climate/swiss-climate-in-detail/raeumliche-klimaanalysen.html (accessed 14 January 2021).
MeteoSwiss, (Federal Office of Meteorology and Climatology), 2021b. Climate Normals [WWW Document]. URL https://www.meteoswiss.admin.ch/home/climate/swiss-climate-in-detail/climate-normals.html (accessed 14 January 2021).
Minasny, B., Malone, B.P., McBratney, A.B., Angers, D.A., Arrouays, D., Chambers, A., Chaplot, V., Chen, Z.S., Cheng, K., Das, B.S., Field, D.J., Gimona, A., Hedley, C.B., Hong, S.Y., Mandal, B., Marchant, B.P., Martin, M., McConkey, B.G., Mulder, V.L., O'Rourke, S., Richer-de-Forges, A.C., Odeh, I., Padarian, J., Paustian, K., Pan, G., Poggio, L., Savin, I., Stolbovoy, V., Stockmann, U., Sulaeman, Y., Tsui, C.C., Vågen, T.G., van Wesemael, B., Winowiecki, L., 2017. Soil carbon 4 per mille. Geoderma 292, 59–86. https://doi.org/10.1016/j.geoderma.2017.01.002.
Naylor, D., Sadler, N., Bhattacharjee, A., Graham, E.B., Anderton, C.R., McClure, R., Lipton, M., Hofmockel, K.S., Jansson, J.K., 2020. Soil microbiomes under climate change and implications for carbon cycling. Annu. Rev. Environ. Resour. 45, 29–59. https://doi.org/10.1146/annurev-environ-012320-082720.
Nerger, R., Funk, R., Cordsen, E., Fohrer, N., 2017. Application of a modeling approach to designate soil and soil organic carbon loss to wind erosion on long-term monitoring sites (BDF) in Northern Germany. Aeolian Res 25, 135–147. https://doi.org/10.1016/j.aeolia.2017.03.006.
Poeplau, C., Don, A., Vesterdal, L., Leifeld, J., Van Wesemael, B., Schumacher, J., Gensior, A., 2011. Temporal dynamics of soil organic carbon after land-use change in the temperate zone - carbon response functions as a model approach. Glob. Chang. Biol. 17, 2415–2427. https://doi.org/10.1111/j.1365-2486.2011.02408.x.
Priestley, C.H.B., Taylor, R.J., 1972. On the assessment of surface heat flux and evaporation using large-scale parameters. Mon. Weather Rev. 100, 81–92. https://doi.org/10.1175/1520-0493(1972)100<0081:otaosh>2.3.co;2.
Reijneveld, A., van Wensem, J., Oenema, O., 2009. Soil organic carbon contents of agricultural land in the Netherlands between 1984 and 2004. Geoderma 152, 231–238. https://doi.org/10.1016/j.geoderma.2009.06.007.
Rietveld, M.R., 1978. A new method for estimating the regression coefficients in the formula relating solar radiation to sunshine. Agric. Meteorol. 19, 243–252. https://doi.org/10.1016/0002-1571(78)90014-6.
Rodríguez Martín, J.A. y, Álvaro-Fuentes, J., Gabriel, J.L., Gutiérrez, C., Nanos, N., Escuer, M., Ramos-Miras, J.J., Gil, C., Martín-Lammerding, D., Boluda, R., 2019. Soil organic carbon stock on the Majorca island: temporal change in agricultural soil over the last 10 years. Catena 181, 104087. https://doi.org/10.1016/j.catena.2019.104087.

Scheidung, H., Bornemann, L., Welp, G., 2017. Seasonal variability of soil organic carbon fractions under arable land. Pedosphere 27, 380–386. https://doi.org/10.1016/S1002-0160(17)60326-6.
Sierra, C.A., Müller, M., Trumbore, S.E., 2012. Models of soil organic matter decomposition: the SoilR package, version 1.0. Geosci. Model Dev. 5, 1045–1060. https://doi.org/10.5194/gmd-5-1045-2012.
Smith, P., 2004. How long before a change in soil organic carbon can be detected? Glob. Chang. Biol. 10, 1878–1883. https://doi.org/10.1111/j.1365-2486.2004.00854.x.
Taghizadeh-Toosi, A., Olesen, J.E., Kristensen, K., Elsgaard, L., Østergaard, H.S., Lægdsmand, M., Greve, M.H., Christensen, B.T., 2014. Changes in carbon stocks of Danish agricultural mineral soils between 1986 and 2009. Eur. J. Soil Sci. 65, 730–740. https://doi.org/10.1111/ejss.12169.
Van Wesemael, B., Paustian, K., Meersmans, J., Goidts, E., Barancikova, G., Easter, M., 2010. Agricultural management explains historic changes in regional soil carbon stocks. Proc. Natl. Acad. Sci. U. S. A 107, 14926–14930. https://doi.org/10.1073/pnas.1002592107.
Wang, J., Li, Y., Bork, E.W., Richter, G.M., Chen, C., Hussain Shah, S.H., Mezbahuddin, S., 2021. Effects of grazing management on spatio-temporal heterogeneity of soil carbon and greenhouse gas emissions of grasslands and rangelands: Monitoring, assessment and scaling-up. J. Clean. Prod. 288, 125737 https://doi.org/10.1016/j.jclepro.2020.125737.
Weihermüller, L., Graf, A., Herbst, M., Vereecken, H., 2013. Simple pedotransfer functions to initialize reactive carbon pools of the RothC model. Eur. J. Soil Sci. 64, 567–575. https://doi.org/10.1111/ejss.12036.
van Wesemael, B., Paustian, K., Andrén, O., Cerri, C.E.P., Dodd, M., Etchevers, J., Goidts, E., Grace, P., Kätterer, T., McConkey, B.G., Ogle, S., Pan, G., Siebner, C., 2011. How can soil monitoring networks be used to improve predictions of organic carbon pool dynamics and CO_2 fluxes in agricultural soils? Plant Soil 338, 247–259. https://doi.org/10.1007/s11104-010-0567-z.
Wiesmeier, M., Poeplau, C., Sierra, C.A., Maier, H., Frühauf, C., Hübner, R., Kühnel, A., Spörlein, P., Geuß, U., Hangen, E., Schilling, B., Von Lützow, M., Kögel-Knabner, I., 2016. Projected loss of soil organic carbon in temperate agricultural soils in the 21 st century: effects of climate change and carbon input trends. Sci. Rep. 6, 1–17. https://doi.org/10.1038/srep32525.
Wiesmeier, M., Urbanski, L., Hobley, E., Lang, B., von Lützow, M., Marin-Spiotta, E., van Wesemael, B., Rabot, E., Ließ, M., Garcia-Franco, N., Wollschläger, U., Vogel, H.J., Kögel-Knabner, I., 2019. Soil organic carbon storage as a key function of soils - a review of drivers and indicators at various scales. Geoderma 333, 149–162. https://doi.org/10.1016/j.geoderma.2018.07.026.
Woloszczyk, P., Fiencke, C., Elsner, D.C., Cordsen, E., Pfeiffer, E.M., 2020. Spatial and temporal patterns in soil organic carbon, microbial biomass and activity under different land-use types in a long-term soil-monitoring network. Pedobiol. (Jena.) 80. https://doi.org/10.1016/j.pedobi.2020.150642.
Wuest, S., 2014. Seasonal variation in soil organic carbon. Soil Sci. Soc. Am. J. 78, 1442–1447. https://doi.org/10.2136/sssaj2013.10.0447.
Wüst-Galley, C., Keel, S.G., Leifeld, J., 2020. A model-based carbon inventory for Switzerland ' s mineral agricultural soils using RothC. Agroscope Sci.
Wüst-Galley, C., Keel, S.G., Leifeld, J., 2021. Modelling SOC in Switzerland ' s mineral agricultural soils using RothC: Sensitivity analysis Authors. Agroscope Sci.
Zhang, C., McGrath, D., 2004. Geostatistical and GIS analyses on soil organic carbon concentrations in grassland of southeastern Ireland from two different periods. Geoderma 119, 261–275. https://doi.org/10.1016/j.geoderma.2003.08.004.