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Soil and land use factors control organic carbon status and accumulation in agricultural soils of Lower Austria

Walter W. Wenzel *, Olivier Duboc, Alireza Golestanifard, Christian Holzinger, Kilian Mayr, Johanna Reiter, Anna Schiefer

University of Natural Resources and Life Sciences, Vienna (BOKU), Department of Forest and Soil Sciences, Institute of Soil Research, Konrad Lorenz Strasse 24, Tulln A-3430, Austria

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A B S T R A C T

Policies for restoring soil health and mitigating climate change require information on soil organic carbon (*SOC*) stocks and their spatial and temporal variation, and related sequestration potentials. Using the province of Lower Austria as environmentally diverse model region, we present a detailed analysis of *SOC* stocks, saturation potentials and deficits along with *SOC* monitoring data for the past three decades.

Using the provincés soil database (sampling 1991) we estimated the saturation potential for mineral-protected C (C_{sat}) by boundary line regression to the particle fraction <20 µm ($f_{<20\mu m}$). Compared to published work, parameterization by Lower Austrian data yielded considerably larger C_{sat} values which agree well with independent maximum organic C load estimates based on specific surface area. C_{sat} is particularly small in topsoils of regions with non-calcareous igneous rock, but large on fine-textured quaternary and tertiary sediments, and weathering residuals of carbonate rock. During the past three decades, the medians of *SOC* in grassland topsoils (0–20 cm) increased significantly ($p < 0.05$) by 29.7% from 39.4 to 51.1 g kg^{-1}, corresponding to annual accumulation rates of 0.87 Mg C ha^{-1}. *SOC* in cultivated soils increased by 17.1% from 12.7 to 14.8 g kg^{-1} (0.20 Mg C ha^{-1} y^{-1}). Because of the large initial C gap, the observed *SOC* accumulation is not related to C_{def} but likely reflects improved soil management.

1. Introduction

Soil organic carbon (*SOC*) is an important soil component with manifold positive effects on physical, chemical and biological soil properties and related soil services such as soil fertility, water holding capacity, water infiltration, and soil biodiversity. Appropriate *SOC* levels are protective against major soil threats such as soil erosion and soil compaction, and essential for ecosystem functioning and resilience. In recent years, the *SOC* pool has attracted attention in the climate change debate as potential sink for mitigating CO_2 emissions to the atmosphere (Lal, 2004; Paustian et al., 2019).

Carbon storage in soil is related to various drivers including climate, topography, parent material, soil properties, interaction with organisms (vegetation, animals, soil biota), and land use and management. Among the soil factors, soil aggregation and texture, mineralogy, and the related specific surface area are considered as key variables controlling the potential of carbon storage in soils (Wiesmeier et al., 2019).

Generally, soil carbon sequestration is a non-linear process. Long-term experiments showed that increases in *SOC* are often greatest after a change in land use or land management (Lal, 2004) following a sigmoid curve with decreasing rates of C changes until a new *SOC* equilibrium (after 20 to 100 years) is reached (Lal, 2004; Sanderman et al., 2010). The equilibrium at which *SOC* stabilizes depends mainly upon site-specific properties, comprising soil (e.g., clay content, cation exchange capacity) and climatic characteristics such as soil moisture and ambient temperature (Wiesmeier et al., 2019).

Six et al. (2002) developed a conceptual *SOM* model defining a maximum soil C storage potential comprising (1) silt- and clay protected SOM (chemical stabilization), (2) micro-aggregate protected *SOM* (physical stabilization), (3) biochemically protected *SOM* (biochemical stabilization), and (4) a non-protected C pool. Each of these C pools shows its own dynamics and stabilizing mechanisms determining the overall C saturation potential (C_{sat}) of soil. This conceptual model was tested by long-term experiments showing that small soil size fractions

* Corresponding author.
E-mail address: walter.wenzel@boku.ac.at (W.W. Wenzel).

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including free micro-aggregates, the free silt- and clay fraction, and the silt- and clay micro-aggregates reached carbon saturation through chemical, partly biochemical and physical protection after application of organic amendments (Chung et al., 2008; Stewart et al., 2007; Stewart et al., 2008). After C saturation in small soil size fractions was reached, C was sequestered in larger fractions following a hierarchy from smaller to larger sized C pools based on the hierarchical model of aggregate formation by Tisdall and Oades (1982). *SOC* located in larger-sized soil aggregates or non-protected C pools shows faster turnover and is more sensitive to site variables, such as climate, topography, soil and crop management (Tan et al., 2004; Chung et al., 2008) than *SOC* protected by the clay- and silt-sized fraction (Jolivet et al. 2003). Therefore, the theoretical C_{sat} may not be reached due to agricultural management practices destroying C-rich macro-aggregates as well as stimulating decomposition (Chung et al., 2008). On the other hand, similar management practices could result in *SOC* accumulation in one soil that is far from its C_{sat}, while no change is observed in another soil much closer to its saturation level (Sanderman et al., 2010).

Recent developments in soil organic matter research suggest that physical disconnection and sorption to the soil mineral phase largely control *SOM* accumulation in soil whereas stabilization by condensation reactions of biomolecules could not be confirmed by novel methods of direct in-situ imaging and spectroscopy (Schmidt et al., 2011; Lehmann and Kleber, 2015). The emerging understanding of *SOM* stabilization in soil therefore emphasizes the importance of site-specific environmental conditions such as physical heterogeneity whereas the molecular structure and size of organic molecules appears to be less important (Schmidt et al., 2011). However, evidence for biochemical protection of char-type C by minerals (Brodowski et al., 2006), and the recently proposed C stabilization by strong hydrogen bonds in supramolecular structures (Wells, 2019) and by cations (Galicia-Andrés et al., 2021) should not be ignored, but currently their contribution to C_{sat} cannot be quantified.

Using least-squares linear regression (LSR), Hassink (1997) found a positive correlation between the textural fraction <20 µm ($f_{<20\mu m}$) and its associated C, but no correlation between the amount of C and $f_{>20\mu m}$. Considering the $f_{<20\mu m}$-associated *SOM* fraction as stable, several attempts have been made to estimate C_{sat} at regional or national level (Angers et al., 2011; Beare et al., 2014; Carter et al., 2003; Chan 2001; Conant et al., 2003; Di et al., 2017; Sparrow et al., 2006; Wiesmeier et al., 2014; Zhao et al., 2006). Most studies adopted Hassinḱs empirical equation to predict C_{sat} and compared their results to the actual stable organic carbon concentrations (*SSOC*). The difference between C_{sat} and *SSOC* is interpreted as the carbon saturation deficit (C_{def}).

However, Feng et al. (2013) showed that Hassinḱs equation may underestimate the maximum protective capacity of soils for *SOC*. Using boundary line regression analyses (BL) they found that the upper envelope of *SSOC* plotted against $f_{<20\ \mu m}$ yields, compared to Hassinḱs model, almost doubled estimates of C_{sat}. They also calculated C_{sat} by assuming an average specific surface area (*SSA*) of 80 $m^2\ g^{-1}$ and a maximum C load (OCL) of 1 mg m^{-2}. For soils dominated by 2:1-minerals, they found close agreement of C_{sat} estimated by the two approaches (Feng et al., 2013). Similarly, Beare et al. (2014) estimated *SSA* from the water content of air-dry samples (Parfitt et al., 2001), and predicted C_{sat} of New Zealand soils using the same OCL of 1 mg m^{-2} but accounting for the variation of *SSA* among soils.

Information on the soil carbon potentials and status, and its change in the recent past at regional level are a prerequisite for the evaluation of soil health and the role of soils in greenhouse gas mitigation. Here we explore the data of a soil inventory of the province of Lower Austria to calculate soil organic *SOC* stocks and to compare their estimated stable proportion with calculated C_{sat} in cultivated and grassland soils using boundary line (BL), and a modified organic carbon load (OCL) approach. Further we compare C_{sat} with *SSOC* concentrations measured in the early 1990ies, to derive C_{def}, and present *SOC* data obtained from topsoil monitoring to assess changes of the *SOC* status of Lower Austrian soils during the past three decades. The results are discussed in relation to the effect of land use, soil formation and environmental factors, soil management, and the potential of C sequestration for greenhouse gas mitigation. Based on published information, we hypothesized that (1) the BL parameterized with Lower Austrian data agrees well with that for 2:1-mineral-soils published by Feng et al. (2013) and provides a better estimate of C_{sat} in Lower Austrian topsoils than Hassinḱs equation; (2) grassland soils have larger *SOC* stocks and are generally closer to saturation than cultivated soils; (3) improved soil management resulted in increased *SOC* concentrations in agricultural topsoils, and (4) the C accumulation rate increases with the initial C_{def}, and therefore generally larger in cultivated soils.

By linking the *SOC* status and its observed temporal change with SOC storage potentials and deficits we present unique information for an ecologically diverse area with relevance beyond the regional scale.

2. Materials and methods

2.1. Study area

Lower Austria is one of nine Austrian provinces located in the surroundings of Vienna. The province is characterized by highly variable climatic conditions, with mean annual temperatures ranging between ~0.7 and ~10 °C, mean annual precipitation between ~470 and ~2100 mm (Strauss et al., 2013), and related variation of the soil cover and land use categories. Due to favourable temperatures and moderate to low precipitation, the lowlands in the eastern regions are dominated by Phaeozems and Chernozems on fine sediments (Loess, fluviatile sediments) under cultivation. The lowlands of the central region are covered by Stagnosols and Luvisols, formed on clayey and silty quaternary and tertiary substrates. Like in the eastern lowlands, the climate conditions are favourable for cultivation, with precipitation increasing from east to west. The north-eastern part of the province belongs to the highlands of the Bohemian massif. Owing to igneous (granite) and metamorphic (schists) rocks as parent material, along with lower temperatures and higher precipitation, the soil cover is dominated by Cambisols with widespread inclusions of Stagnosols and Gleysols, and, in the western highlands, also Podzols. The main land use category is cultivated land, with increasing proportion of grassland towards the northern and western parts of the Bohemian massif. The southern regions of Lower Austria belong to the Alps, with steep gradients of decreasing temperatures and increasing precipitation towards higher elevation. Most of the soils are derived from residual weathering on limestone and dolomite rock, resulting in clayey textures. The soil cover is dominated by Cambisols associated with a mosaic of Leptosols/Regosols. Due to the prevailing climate conditions, soil use is restricted to grasslands. Only the south-easternmost, lower parts of the Alpine zone are covered by Cambisols developed on non-calcareous parent materials (mainly schists). Soil and climate conditions allow for a considerable share of cultivated land.

2.2. Soil sampling and measurements of soil organic carbon (SOC) and soil texture in the frame of the Lower Austrian soil inventory and Austrian soil mapping programme

Measured *SOC* values for cultivated and grassland soils were obtained from the Lower Austrian soil inventory database (Amt der Niederösterreichischen Landesregierung, 1994). Sampling locations had been arranged in the nodes of a regular, 3.9x3.9 km grid system, and additionally in the centres of the grid cells, resulting in a distance of 2.75 km between sampling locations. Out of 1730 grid points in agricultural land (i.e., cultivated and grassland), 1449 (725 from regular nodes, 724 from grid cell centres) had been sampled because the remaining were located in areas which were inaccessible or covered (sealed) by infrastructure. Under cultivation, 576 soil profiles were sampled at regular locations according to the depth increments 0–20, 20–40 and 40–50 cm. In grassland 149 samples were collected from regular grid nodes according to the depth increments 0–5, 5–10, 10–20, 20–40 and 40–50 cm. This more detailed sampling accounts for the typically stronger differentiation of genetic horizons in the grassland topsoils. In the centres of each grid cell, only topsoils had been sampled from 0 to 20 cm depth, resulting in 575 cultivated, and 149 grassland samples. At the regular grid locations, samples had been pooled for each depth increment from four subsamples collected in open soil pits located within a circle area of 314 m^2. Sampling at additional locations had been pooled from 20 subsamples collected with a Pürckhauer corer. Soil sampling had been performed in the period from May 1990 to May 1992.

For the calculation of C_{sat}, additional data was retrieved from the Legend of the Austrian Soil Map (Bundesamt für Wald, 2019). Soil samples had been collected according to genetic horizons from soil pits representing the mapping units. Here we only use topsoils from Lower Austrian grassland (251 samples) soils.

Samples had been air-dried and passed through a 2-mm screen. The fine earth fraction (<2 mm) had been analysed for the following characteristics:

- textural classes sand (2000–63 μm), silt (63–2 μm) and clay (<2mm) using a combined sieving and sedimentation method after dispersion with sodium pyrophosphate, and – at humus contents >50 g kg^{-1} – pre-treatment with H_2O_2 (ÖNORM L (1061), 1988);
- soil organic matter (*SOM*) using wet oxidation with $K_2Cr_2O_7$ solution and concentrated sulfuric acid, and subsequent colorimetric measurement of the resulting Cr(III) (ÖNORM L (1081), 1989).

Soil sampling and analysis had been conducted by the former Bundesanstalt für Bodenwirtschaft, Vienna.

For use in the present work, we converted the *SOM* data to organic carbon dividing by the widely accepted factor of 1.724. To account for the differences between *SOC* calculated from *SOM*, and combustion-based total elemental analysis, the data was corrected using a factor of 1.2 determined for Austrian soils (Gerzabek et al., 2005), yielding the *SOC* data as further used in this study. Using re-measurements of *SOC* by dry combustion on a subset of archived soils (N = 517) of the Lower Austrian inventory and the corresponding values calculated as described above, we found a close ($r^2 = 0.97$), near 1:1 relation ($SOC_{measured} = 1.03$ $SOC_{predicted}$) between measured and predicted *SOC* concentrations, indicating that on average the calculated values are by 3% less than measured *SOC*.

Selected soil characteristics are compiled in Table 1.

2.3. Calculation of stable soil organic carbon (SSOC) and the particle fraction <20 μm ($f_{<20\mu m}$)

Stable soil organic carbon (*SSOC*) was calculated by multiplying *SOC* with the factor 0.85 (Angers et al., 2011) to correct for organic carbon that is not protected by the fraction $f_{<20\mu}$.

Predicting C_{sat} from soil textural data according to Hassink (1997) or Feng et al. (2013) requires information on $f_{<20\mu}$. As the Lower Austrian soil database only provides information on the clay ($f_{<2\mu m}$), silt ($f_{2\text{-}63\mu m}$) and sand ($f_{63\text{-}2000\mu m}$) fractions, we used published data from various other sources (Jandl, 1987; Alge et al., 1993; Bröcker and Nestroy, 1995; Katzensteiner et al., 2001; Nelhiebl, 2001; Schneider, 2001; Rampazzo et al., 2001; Mentler et al., 2001; Strauss et al., 2001, Nestroy, 2001) to establish a regression equation (Eq. (1)) predicting $f_{<20\mu}$ (g 100 g^{-1}) from measured $f_{<63\mu m}$ (g 100 g^{-1}):

$$f_{<20\mu m} = 0.3171(\pm 1.1547) f_{<63\mu m}^{1.1647\ (\pm 0.0358)} \left(n = 258; r^2 = 0.8053; RMSE = 1.3491\right) \tag{1}$$

The fraction $f_{<63\mu m}$ was calculated as the sum of the measured $f_{<2\mu m}$ and $f_{2\text{-}63\mu m}$ fractions. Our database for predicting $f_{<20\mu m}$ covers soils of different land use (cultivated, grassland, forest) and various major WRB soil groups (IUSS Working Group WRB, 2014), including Leptosols, Fluvisols, Gleysols, Chernozems, Phaeozems, Podsols, Stagnosols, Umbrisols, Cambisols, Regosols from Lower Austria and neighbouring provinces. It includes data from different soil horizons as preliminary calculations showed no relevant difference between data subsets from different soil depth.

2.4. Calculation of the carbon saturation potential (C_{sat}) and the saturation deficit (C_{def})

Using the BL method proposed by Feng et al. (2013), we calculated C_{sat} based on the 149 grassland topsoils (0–20 cm) collected from the center of the grid cells of the Lower Austrian soil inventory, and 251 grassland topsoils (sampled according to genetic horizons) obtained from the Austrian Digital Soil Map 1:25:000 (Bundesamt für Wald, 2019). Grassland soils were chosen as they are generally closer to saturation than cultivated soils (Feng et al., 2013; Hassink, 1997). To this end we sorted the $f_{<20\mu}$ data into groups with intervals of 10 g $f_{<20\mu}$ 100 g^{-1} soil. Then the 90th percentile of the *SSOC* concentrations in each group was calculated and related to the medians of each $f_{<20\mu}$ group in a linear regression analysis with the intercept being forced through zero. C_{def} (g kg^{-1}) was calculated as the difference between C_{sat} and *SSOC* for each individual soil.

To obtain independent estimates, we also calculated C_{sat} from the specific surface area (*SSA*, m^2 g^{-1} soil) of the mineral fraction <2 mm (Eq. (2)), assuming a maximum C load (*OCL*) of 1 mg C m^{-2} soil (Feng et al., 2013; Beare et al., 2014):

$$C_{sat} = SSA\ OCL \tag{2}$$

As direct measurements of *SSA* are not available, we used literature data for European soils (Ersahin et al., 2006; Petersen et al., 1996; Zehetner and Wenzel, 2000) to derive a pedotransfer function (Eq. (3)) between measured cation exchange capacity (*CEC*, $mmol_c$ kg^{-1}) and *SSA* (m^2 g^{-1} soil):

$$SSA = 0.5435 CEC\left(r^2 = 0.9025, RMSE = 38.95, n = 56\right) \tag{3}$$

CEC was corrected for the contribution of *SOM*. We employed curve fitting (EXCEL Solver, Version 15.33, GRG-non-linear mode) to estimate the contributions of *SOM* (g kg^{-1}) and the mineral phase, i.e., clay ($f_{<2\mu m}$, g 100 g^{-1}) and silt ($f_{2\text{-}63\mu m}$, g 100 g^{-1}) fractions, yielding

$$CEC = 3.7819 f_{<2\mu m} + 0,9503 f_{2-63\mu m} + 15.58 SOM\left(r^2 = 0.6448, RMSE = 36.33, n = 1220\right) \tag{4}$$

The difference between the fitted total *CEC* and the *CEC* share of *SOM* was considered to represent the *CEC* of the mineral soil. Using these estimates of mineral-associated *CEC*, we calculated C_{sat} from Eq. (4).

The values for *CEC*, *SOM*, $f_{<2\mu m}$ and $f_{2\text{-}63\mu m}$ were obtained from the Lower Austria soil database and comprise 149 grassland topsoils (0–20 cm; topsoil samples from grid cell centres), and 1070 cultivated topsoils (0–20 cm; topsoils from regular grid cell nods and grid cell centres).

2.5. Calculation of carbon stocks and sequestration potentials

Soil organic carbon stocks (*SSOC*; Mg ha^{-1}) were calculated according to Eq. (5):

$$SSOC = SOC\rho 10000t(1\ - -f_{>2mm}/100) \tag{5}$$

where ρ (Mg m^{-3}) refers to the bulk density and *t* (m) to the depth of the soil layer, and *SOC* is given in g kg^{-1}. The factor 10,000 is used to convert from square meters to hectares. $f_{>2mm}$ (%v/v) relates to the soil's content of rock and mineral fragments >2 mm equivalent diameter. This fraction is assumed to be free of organic carbon. The values for $f_{>2mm}$ were obtained by converting the classes (0–10, 10–20, 20–40, 40–70, 70–100 %v/v) available from the soil description (Amt der Niederösterreichischen Landesregierung, 1994) into discrete numbers by using

Table 1
Selected characteristics of the cultivated and grassland soils of the Lower Austrian soil inventory (data source: Amt der Niederösterreichischen Landesregierung, 1994). Data represent arithmetic means and standard deviations (SD).

Land use category	Depth increment	Number of samples	Statistical parameter	Sand[a]	Silt[a]	Clay[a]	Carbonate equivalent[b]	OM[c]	N[d]	CEC[e]	Base saturation %	pH (0.01M$CaCl_2$)[f]
	cm						g kg^{-1}			$mmol_c$ kg^{-1}		
							Regular grid nodes					
	0–20	576	Mean ± SD	302 ± 184	503 ± 143	196 ± 87	73 ± 108	22.8 ± 15.5	1.75 ± 0.85	159 ± 61	97 ± 7.6	6.63 ± 1.04
			Range	20–480	80–810	10–510	0–756	4–258	0.3–12.4	20.2–526	33–100	3.98–7.75
	20–40	571	Mean ± SD	299 ± 191	494 ± 148	207 ± 89	82 ± 118	15.1 ± 15.3	1.21 ± 0.81	150 ± 65	97 ± 10	6.70 ± 1.06
			Range	20–840	20–810	20–540	0–772	1–289	0.1–11.5	23.6–545	27–100	4.01–7.79
Cultivated soils	40–50	539	Mean ± SD	298 ± 206	492 ± 158	210 ± 85	102 ± 134	8.91 ± 7.50	0.77 ± 0.56	141 ± 62	97 ± 9.6	6.77 ± 1.08
			Range	10–830	90–800	20–490	0–776	1–72	0.1–6.8	4.7–390	19–100	3.89–7.92
							Additional samples (grid cell centres)					
	0–20	575	Mean ± SD	307 ± 179	498 ± 137	195 ± 87	73 ± 113	23.1 ± 14.6	1.71 ± 0.73	161 ± 55	98 ± 6.8	6.72 ± 0.98
			Range	20–760	130–830	30–440	0–743	4–129	0.2–6.7	45.2–351	46–100	4.01–7.82
Grassland soils							Regular grid nodes					
	0–5	149	Mean ± SD	267 ± 160	545 ± 116	189 ± 84	27 ± 85	83.9 ± 49.5	4.57 ± 2.13	224 ± 80	95 ± 8.4	5.71 ± 0.76
			Range	30–660	250–810	40–410	0–439	19–361	0.5–17.3	69.8–591	53–100	4.21–7.33
	5–10	148	Mean ± SD	270 ± 166	538 ± 117	182 ± 89	30 ± 94	54.1 ± 37.8	3.39 ± 1.62	197 ± 75	94 ± 12	5.67 ± 0.84
			Range	20–670	250–810	40–450	0–497	14–244	0.2–12.6	62.3–539	31–100	3.93–7.37
	10–20	149	Mean ± SD	276 ± 173	525 ± 123	199 ± 95	35 ± 112	33.4 ± 28.5	2.33 ± 1.13	170 ± 68	92 ± 14	5.67 ± 0.93
			Range	20–740	190–800	40–410	0–709	9–250	0.6–7.8	50.3–577	28–100	4.05–7.46
	20–40	143	Mean ± SD	280 ± 184	496 ± 127	223 ± 109	38 ± 117	14.8 ± 14.9	1.17 ± 0.76	135 ± 66	91 ± 14	5.69 ± 1.04
			Range	10–840	110–780	40–470	0–740	2–114	0.2–4.9	19.8–320	35–100	3.99–7.61
	40–50	129	Mean ± SD	286 ± 202	473 ± 136	242 ± 120	33 ± 112	9.79 ± 16.6	0.79 ± 0.59	130 ± 67	92 ± 14	5.72 ± 1.05
			Range	10–880	90–910	30–540	0–711	1–150	0.1–4.2	8.9–321	38–100	4.15–7.66
							Additional samples (grid cell centres)					
	0–20	149	Mean ± SD	282 ± 176	522 ± 119	196 ± 99	39 ± 115	57.4 ± 36.3	3.54 ± 1.65	205 ± 80	95 ± 8.6	5.81 ± 0.84
			Range	10–720	190–760	50–350	0–680	14–237	1.2–11	76.5–506	47–100	3.97–7.46

[a] Combined sieve and piptte method according to ÖNORM L-1061 (1988).
[b] Calcimetric Scheibler method according to ÖNORM L-1084 (1989).
[c] Organic matter measured by wet oxidation using $K_2Cr_2O_7$ and concentrated H_2SO_4 according to ÖNORM L-1081 (1989).
[d] Total nitrogen using the Kjeldahl method according to ÖNORM L-1082 (1989).
[e] Cation exchange capacity using 0.1 M $BaCl_2$ and backchange by HCl according to ÖNORM L-1086 (1989).
[f] According to ÖNORM L 1083 (1989).

the average of the class limits, yielding 5, 15, 30, 55, and 85 %v/v. To estimate the effect of the uncertainty of $f_{>2mm}$ we also calculated *SSOC* for the lower and upper boundary of the rock fragment classes.

As bulk density (ρ) was not directly available from the Lower Austrian soil inventory database, we calculated ρ from *SOC* using the regression (Eq. (6)) of Ruehlmann and Körschens (2009):

$$\rho = (2.684 - 140.943(\pm 7.226)b)exp(-b\ SOC) \quad (6)$$

where ρ is given in Mg m^{-3}, and *SOC* in g kg^{-1}.

For the parameter *b* we chose the value 0.008 ($\pm$3.5x10^{-4}) valid for cultivated soils (Ruehlmann and Körschens, 2009).

We evaluated the average error of the predictions using data (n = 28) of Rampazzo et al. (2001) and Mentler et al. (2001) who reported measured values of ρ (range: 830 – 1580 kg m^{-3}) and *SOC* (range: 1.0 – 86 g kg^{-1}) that are representative for most soils of our study. The median of relative errors of the predictions was + 4.01% (range: −16.8% – +14.8%).

2.6. Monitoring of temporal changes of SOC

In the years 2015–2020 we revisited sites for which archived samples from soil profiles of the Austrian Digital Soil Map (Bundesamt für Wald, 2019) collected during the period 1985–2000 are available. At each site, a composite topsoil sample was obtained by combining >10 subsamples collected from the same depth increment (A horizons) as indicated in the soil map database (Bundesamt für Wald, 2019). The subsamples were distributed randomly within a circle of 10 m diameter around the initial sampling point. In addition, in 2020 we re-collected cultivated topsoils (0–20 cm) from 298 sites of the Lower Austrian soil inventory using the same sampling approach. Overall, this resulted in 754 pairs of archived and re-sampled topsoils, with 559 sites under cultivation and 195 under permanent grassland. Note that archived soil samples were only available for selected ecological regions. Most sampling sites are therefore located in the Lower Austrian Alps and the eastern part of the Vienna Woods with large share of grasslands, and in the cultivated north-eastern and central lowlands, and the northern Vienna Basin.

Archived and re-sampled soils were air-dried, passed through a 2-mm screen, milled, and measured for total organic carbon by dry combustion (ÖNORM L 1080, 2013) (Vario Macro Cube, Elementar) and carbonate equivalent using the calcimetric Scheibler method (ÖNORM L 1084, 1989). *SOC* was calculated as the difference of the total and inorganic C by assuming that all carbonate was present as $CaCO_3$. Alternatively, for part of the samples (298 cultivated sites of the soil inventory) we used the element analyzer soli TOC cube of the same manufacturer (Elementar) which allows direct measurement of the total organic carbon after heating to 600° without need to correct for the inorganic C.

2.7. Statistics

All statistics were processed in EXCEL Version 16.0 using standard functions and own computations. Descriptive statistics presented include arithmetic means, standard deviations, medians, ranges, 25 and 75% quartiles. Differences between medians of *SOC* at different sampling times were evaluated using the Wilcoxon signed-rank test for dependent samples as normal distribution could not be confirmed by the Kolmogorov-Smirnov test. For significant ($p < 0.05$) differences the effect size ***r*** was calculated as the ratio between the Wilcoxon test statistics *W* and the total rank sum *S* to determine the relevance of the difference. According to Cohensclassification the effect was considered small for r ~0.1, moderate for r ~0.3, and large for r ~0.5.

After rejecting the null hypothesis of the Kolmogorov-Smirnov test, differences of medians of C_{sat}, C_{def} and *SOC* between ecological regions and WRB soil groups, respectively, were evaluated using the test of Kruskal-Wallis at $p < 0.05$.

Single linear and non-linear regression and correlation analysis was employed to explore relationships and derive pedotransfer functions between predictor and response variables. The predictive power of regression equations was evaluated by the squared correlation coefficients (r^2) and the root mean square error (RMSE), related uncertainties by prediction bands. Multiple correlation analysis was used to assess the effect of climate and soil characteristics on *SOC*.

3. Results and discussion

3.1. Soil organic carbon concentrations (SOC) in Lower Austrian agricultural soils

Measured *SOC* concentrations in cultivated and grassland soils sampled at the regular grid points of the soil inventory are compiled in Table 2. Within the top 20 cm the medians of *SOC* in grassland soils decrease from 54.3 (0–5 cm) to 18.8 g kg^{-1} (10–20 cm) which compares to a considerably lower concentration of 13.9 g kg^{-1} throughout the 0–20 cm layer in cultivated soils. In deeper layers the differences between land uses are less pronounced, with slightly lager *SOC* concentrations in cultivated as compared to grassland soils in the layer 20–40 cm, and slightly smaller *SOC* concentrations in the layer 40–50 cm (Table 2). The data obtained for the additional topsoil samples located in the grid cell centres is similar in terms of the medians but biased towards lower mean and maximum concentrations in the cultivated soils (Table 2).

The median of *SOC* concentrations in cultivated Lower Austrian topsoils (13.9 g kg^{-1}) is smaller than in the neighbouring provinces Upper Austria (15.7), Styria (17.4) and Burgenland (15.1) (Gerzabek et al., 2005). While in Styria and Upper Austria this may be related to cooler and more humid climate, the reasons are not obvious for Burgenland. In contrast, the *SOC* concentrations in the uppermost three layers (0–5, 5–10, 10–20 cm) of Lower Austrian grassland topsoils (54.3 – 28.5–18.8 g kg^{-1}) clearly exceed those of Upper Austria (42.3 – 27.8 – 18.6 g kg^{-1}) and Burgenland (36.2 – 25.6 – 18.4 g kg^{-1}) especially in the top five centimetres. This could at least partly be attributed to the fact that the benchmarks from the neighbouring provinces are representative of intensively used grasslands whereas Lower Austrian data include also Alpine meadows which typically show larger *SOC* concentrations (Gerzabek et al., 2005). Wiesmeier et al. (2012) found *SOC* concentrations of 30 and 15 g kg^{-1} for grassland (0–20 cm) and cultivated (0–30 cm) topsoils in Bavaria, respectively, which is close to those reported here (Table 2).

3.2. Stocks of SOC and SSOC in Lower Austrian soils

The medians of the *SOC* stocks in cultivated and grassland soils are compiled in Table 3. The data is corrected for the average of the rock fragment abundance class obtained from the field description of the soils. As we observed relatively small deviations from the mean (in cultivated soils < ±4.7%, in grassland soils < ±7.0%) among the estimates of the carbon stocks when we either accounted for the lower or upper rock fragment abundance class boundary, we only report those corrected for the average rock fragment abundance. The effect of rock fragment abundance is small (up to ~5% stock decrease compared to no account for rock fragments) in the cultivated topsoils (0–20 cm) but is more pronounced in the subsoils (20–50 cm; ~13%). Generally larger rock fragment abundance in grassland soils decrease *SOC* by ~16% (topsoils) and ~24% (subsoils). Note that on average the used pedotransfer function of Ruehlmann and Körschens (2009) overestimated the bulk densities by ~4%, likely resulting in corresponding overestimation of *SOC* stocks.

The C stocks reported in Table 3 are by about 75% larger in grassland topsoils (0–20 cm), but by 20% smaller in the subsoils (20–50 cm) as compared to the corresponding soil layers under cultivation. The total stocks down to 50 cm are about 33% smaller in cultivated (66 Mg ha^{-1})

Table 2
Measured organic carbon (SOC) concentrations in cultivated (N = 576) and grassland soils (n = 149) sampled at regular grid points and at additional sites located in the grid centres (topsoil samples only, n = 575 and 149, for cultivated and grassland soils, respectively).

Depth	Cultivated soils						Grassland soils					
	Regular grid samples			Additional topsoil samples			Regular grid samples			Additional topsoil samples		
	Median	Mean	Range	Median	Mean	Range	Median	Mean	Range	Median	Mean	Range
cm	mg kg^{-1}											
0–5	13.9	15.9 ± 10.8	2.78–180	13.9	6.05 ± 10.2	2.8–89.8	54.3	58.4 ± 34.5	13.2–251	29.9	20.3 ± 25.3	9.7–165
5–10							28.5	37.6 ± 26.6	9.74–170			
10–20							18.8	23.2 ± 19.8	6.26–174			
20–40	9.0	10.5 ± 10.6	0.70–201				7.7	10.3 ± 10.4	1.39–79.4			
40–50	4.2	6.2 ± 5.2	0.70–50.1				4.9	6.8 ± 11.6	0.70–104			

Table 3
Medians for the stocks of soil organic carbon (*SOC*), stable organic carbon (*SSOC*), and the carbon saturation potentials (C_{sat}) and carbon deficits (C_{def}) in cultivated (N = 576) and grassland (N = 149) soils of Lower Austria. Data was calculated by Equation (5) using the mean of the coarse fragment fraction class obtained from the field description of the soils.

Soil depth	*SOC* stocks	*SSOC* stocks	C_{sat} stocks	C_{def} stocks
cm	Mg ha^{-1}			
	Cultivated soils			
0–20	36	31	178	143
20–50	29	25		
0–50	66	56		
	Grassland soils			
0–20	63	54	143	88
20–50	24	20		
0–50	87	74		

[1)]C_{sat} calculated using the boundary line approach of Feng et al. (2013) and own data from Lower Austrian grassland topsoils.

compared to grassland soils (87 Mg ha^{-1}). Our results indicate that the smaller *SOC* stocks in cultivated topsoils are not solely related to *SOC* losses to the atmosphere but also to redistribution to deeper layers by ploughing.

Assuming that 85% of *SOC* is present in physically protected form, i. e., associated with the fraction $f_{<20\mu m}$ (Angers et al., 2011; Beare et al., 2014), the median stocks of *SSOC* down to 50 cm soil depth amount to 56 (cultivated) and 74 (grassland) Mg ha^{-1} (Table 3).

3.3. Carbon saturation potentials (C_{sat}) and deficits (C_{def})

The carbon saturation potential delineates a theoretical upper limit of *SSOC* accumulation in soil.

Fig. 1 shows the relation between *SSOC* and the mass proportion of the fine fraction ($f_{<20\mu m}$) in topsoils (<2 mm), derived by the BL and OCL method using Lower Austrian soil data for parameterization. For both methods we find a strong linear correlation ($R^2 > 0.98$) along with reasonable predictive quality as indicated by the values of RMSE, and the standard errors of the slope (Fig. 1). As shown by the similar slopes of the regression lines, both methods agree well in predicting C_{sat}. Owing to the approach of regressing the 90th percentiles of *SSOC* to the class medians of $f_{20\mu m}$, the BL is based on rather few data points, with no data in the uppermost range of the fine fraction (Fig. 1, Panel A). As the OCL method does not require stratification, regression analysis is based on individual data, thus increasing the number of observations and the range of $f_{<20\ \mu m}$ covered. Whereas in the range of $f_{<20\mu m} < 70$ g 100 g^{-1} soil almost all data of the OCL fall within the prediction bands, there is a bias towards larger *SSOC* values in the upper range (Fig. 1, Panel B). Based on detailed exploration of the data we can exclude that this bias is caused by the *SSA* prediction (Equation (1)) or the related fitting procedure to obtain the *CEC* of mineral soil (Equation (4)). There is also no indication that this bias is caused by the transformation of textural data to obtain the fine fraction ($f_{<20\mu m}$) as we used a potency rather than a linear function (Equation (3)), tending to stretch the x-axes of the plots in Fig. 1. Moreover, we found a strikingly similar bias when predicting *SSA* using the pedotransfer function of Schulte et al. (1992). Altogether, this provides evidence for a non-linear increase of *SSA* in the uppermost $f_{<20\mu m}$ range caused by the larger contribution of particles in the clay-sized range, as can be expected if we consider the quadratic relation between radius and surface area of spherical particles. Accordingly, our linear fit may underestimate C_{sat} in extremely fine textured soils.

To explore the relation between the upper boundary of stable organic carbon in cultivated topsoils and $f_{<20\mu m}$, we calculated the 90th percentiles of *SSOC* concentrations following the same approach as for the grassland soils. Owing to the larger number of observations (N = 1449), we narrowed the interval to 5 g $f_{<20\mu m}$ 100 g soil (Fig. 1A). There is no

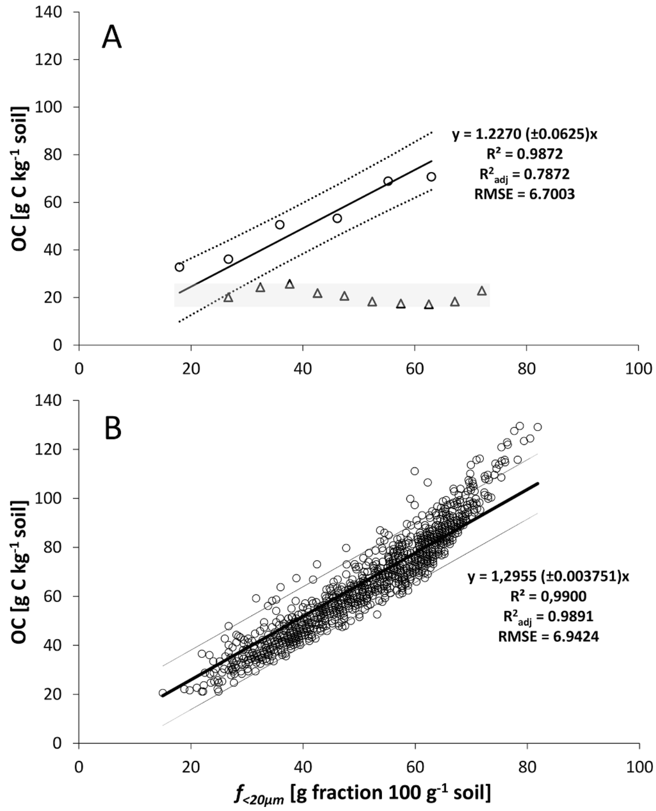

Fig. 1. Relations between the proportion of the fine fraction ($f_{<20\ \mu m}$) and maximal concentrations of mineral-protected, i.e., stabilized C (*SSOC*) in topsoils (<2mm). The plots show model estimates (open circles) and linear regression lines (solid lines) along with the prediction bands (dotted lines; α = 0.05) for the BL (Panel A) derived from Lower Austrian grassland soils (N = 400), and the OCL (Panel B) model, derived from all topsoils (N = 1219). The linear regressions are forced through the intercept, the standard error of the slope is given in parentheses. For further details regarding the models and databases used for their parameterization we refer to the methods section. The open triangles in Panel A represent the BL for cultivated topsoils (N = 1151), the shaded area indicates the upper range of *SSOC* that can be possibly obtained in cultivated soils by best management practices.

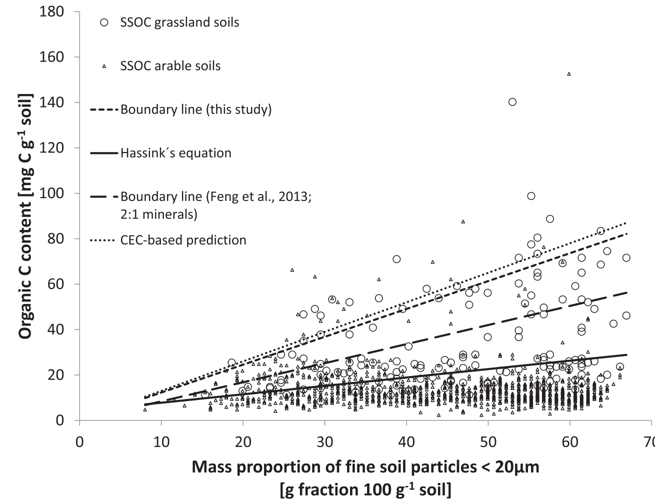

Fig. 2. Relation between *SSOC* (calculated as 85% of SOC) and the mass proportion of soil particles <20 µm in topsoils (0–20 cm) under cultivation (N = 1149, regular grid and additional samples) and grassland (N = 149, additional samples only) of the Lower Austrian soil inventory. The lines indicate the carbon saturation potentials (C_{sat}) reported by Hassink (1997)and Feng et al. (2013), and BL- and *CEC*-based predictions of C_{sat} parameterized using topsoil (0–20 cm) characteristics of the Lower Austrian soil data base.

relation between the two variables as the *SSOC* concentrations vary only moderately within 17 and 25 g C kg^{-1} across the entire fine fraction range. If we assume that the 90th percentile of *SSOC* represents the best available management, the shaded area in Fig. 1A can be interpreted as upper limit of realistic sequestration potentials of cultivated soils. We find no relation to C_{sat} derived from grassland soils, indicating that *SOC* accumulation under cultivation is limited by the efficacy of management options rather than the protective potential of the soil mineral phase.

Fig. 2 shows a plot of *SSOC*, calculated as 85% of measured *SOC* concentrations in Lower Austrian topsoils under cultivation and grassland, versus $f_{<20\mu m}$. The lines represent C_{sat} predicted by the BL and OCL method as a function of $f_{<20\mu m}$. For comparison, we also show the BL for 2:1 mineral soils of Feng et al. (2013) and Hassinḱs equation. Whereas the own parameterizations of the BL and OCL models agree well, C_{sat} estimated by Fenǵs BL equation is by about one third lower. While the relation of Feng et al. (2013) was obtained by regressing measured *SOC* concentrations in the fine fraction to $f_{<20\mu m}$, for our BL parameterization we estimated *SSOC* from *SOC* of the <2 mm fraction using a constant factor of 0.85 (Angers et al., 2011). McNally et al. (2017) found a *SSOC* share close to 83% across the entire range of $f_{<20\mu m}$, indicating that a constant factor as used in our study may be appropriate. A share of *SSOC* close to 85% is also supported by various other studies, including *meta*-analysis of larger datasets (Balesdent et al., 1998; Jolivet et al., 2003; Gregorich et al., 2006). Beare et al. (2014) report an intercept of 1.16 for a BL parameterized with non-allophanic long-term pasture topsoils (0–15 cm) from New Zealand which is relatively close to the intercept of 1.227 obtained for the BL parameterization in our study (Fig. 1). Further, their fits of a BL regression (90th percentile) to the plot of *SSOC* (assumed as 85% of measured SOC) against *SSA* (estimated from the residual moisture content of air-dried soil samples using Parfitt́s equation) supports a maximum C load of ~1 mg C m^{-2} soil as assumed in our OCL calculations. In a subsequent study on New Zealand soils this was confirmed by a measured share of 83% of *SSOC*, and a maximum C loading rate of 0.9 mg C m^{-2} soil (McNally et al., 2017).

The results of our OCL parameterization also exceed those of the OCL reported by Feng et al. (2013), who assumed a constant SSA of 80 m^2 g^{-1} soil, while, using measured *CEC* as predictor of *SSA*, we were able to account for the variation among soils. Our *SSA* estimates appear to be conservative compared to the relation published by Schulte et al. (1992). Applying their pedotransfer function to our *CEC* data results on average in ~25% larger values of *SSA* and subsequently C_{sat}.

These considerations, along with the notably strong agreement between the two independent approaches (BL vs. OCL) to predict C_{sat} (Fig. 2) suggest that our regional parameterizations represent the currently best estimates of C_{sat} for Lower Austrian soils. Our estimates agree well with those for New Zealand soils (Beare et al., 2014) but indicate larger protective capacity in Lower Austrian soils as compared to reports from other regions (Feng et al., 2013; Six et al., 2002; Sparrow et al., 2006; Wiesmeier et al., 2014). Given the close agreement of the two prediction approaches, *CEC* may provide a useful alternative to soil texture ($f_{<20\mu m}$) for predicting C_{sat}. McNally et al. (2017) identified *SSA* as being superior to other soil characteristics in predicting C_{sat} in New Zealand soils.

Most of the cultivated topsoils are clearly below the lowest estimate of C_{sat}, the regression line of Hassink (1997), except in coarse textured soils with $f_{<20\mu m}$ <30 g 100 g^{-1}, confirming the findings of other studies (Angers et al., 2011; Wiesmeier et al., 2014). Consistent with findings in other regions (Beare et al., 2014; Wiesmeier et al., 2014), the *SSOC* concentrations in grassland topsoils vary largely, with a relevant share even exceeding the BL of our study. In contrast, the majority of measured current C concentrations in the $f_{<20\mu m}$ fraction of Bavarian topsoils (0–10 cm) under grassland falls even below Hassinḱs LSR line (Wiesmeier et al., 2014). The large variability shows that grassland soils are not generally saturated, supporting the use of OCL or BL rather than LSR to predict C_{sat}.

The C_{sat} stocks in Lower Austrian topsoils (0–20 cm) were calculated by accounting for bulk density and the average rock fragment abundance (Table 3), yielding medians of 178 and 143 Mg ha^{-1} for cultivated and grassland soils, respectively. The medians of C_{sat} stocks in grassland topsoils are by ~19% smaller than in cultivated soils which can be explained primarily by the larger rock fragment abundance in grassland soils whereas the average mass fraction of silt and clay particles is even slightly smaller in cultivated soil (Table 1).

Using the estimates of *SSOC* and C_{sat} for average rock fragment abundance, we obtained the medians of the C_{def} stocks in topsoils under cultivation (143 Mg ha^{-1}) and grassland regime (88 Mg ha^{-1}). The lower deficits in grassland topsoils are in line with our expectation.

We showed that C_{def} in Lower Austrian topsoils is generally larger than previously reported for other regions McNally et al. (2017), Wiesmeier et al. (2014). Note that C_{sat} values obtained in our study are similar to those of New Zealand soils (McNally et al., 2017) but Lower Austrian soils are more depleted of *SSOC*.

3.4. Variation of stocks by WRB soil group and regions

We calculated stocks of *SOC*, C_{sat} and C_{def} for major soil groups according to the World Reference Base for Soil Resources (IUSS Working Group WRB, 2014). The results are shown for cultivated soils in Fig. 3. The largest *SOC* stocks are observed in Chernozems, followed by Gleysols and Phaeozems. This is consistent with thick A horizons in Chernozems and Phaeozems that formed in the sub-continental (Pannonian) climate of the lowlands, and the preservation of *SOM* in the reducing environments of Gleysols. Among hydromorphic soils, those influenced by groundwater (Gleysols) store more C than episodically flooded soils (Fluvisols) and Stagnosols, which is consistent with results of Wiesmeier et al. (2012) for Bavarian soils. The smallest *SOC* stocks occur in Regosols which typically show limited soil depth and higher abundance of rock fragments, and Fluvisols, which, as a consequence of flooding, remain in an early stage of soil development.

The variation of the stocks of C_{sat} and C_{def} among WRB soil groups is closely related (Fig. 3). The smallest C_{sat} and C_{def} stocks are observed in Umbrisols (89 and 55 Mg C ha^{-1}, respectively), followed by Cambisols. All other soil groups exhibit considerably larger C_{sat} and C_{def} stocks exceeding ~150 Mg C ha^{-1}. Similarly, McNally et al. (2017) found larger C_{def} in Gleys compared to soils of weak (recent soils) to moderate (brown soils) development.

Regional differences of the C status of Lower Austrian soils are reflected by the topsoil (0–20 cm) stocks of C_{sat}, *SOC*, and C_{def} (Fig. 4). Average C_{sat} stocks vary by a factor of ~2, ranging between ~100 (non-calcareous igneous and metamorphic rocks of the north-western parts and highlands of the Bohemian massif) and 200 Mg C ha^{-1} (central and north-eastern lowlands covered by fine quaternary and tertiary sediments). The average *SOC* stocks vary between ~35 and ~75 Mg ha^{-1} and are related to C_{sat} and predominant land use of the regions. The largest *SOC* stocks are observed in the carbonate Alps dominated by soils with high C_{sat} under grassland. The regions with the lowest *SOC* stocks include the north-eastern and central lowlands and the central Bohemian massif, whereas the C_{sat} stocks are clearly smaller in the latter region. While in the central Bohemian massif the low *SOC* stocks are related to rather low C_{sat}, in the lowland regions they are associated with intense cultivation. Average C_{def} stocks in Lower Austrian regions vary considerably (factor ~5) between ~30 and 170 Mg ha^{-1} (Fig. 4). The largest deficits are associated with intensively cultivated soils on fine tertiary and quaternary sediments of the lowlands, the smallest deficits are found in the regions with low C_{sat} and predominant grasslands (highlands and north-western part of the Bohemian massif followed by the silicate Alps).

3.5. Effect of climate on SOC status

Apart from the protective effect of the soil mineral phase, storage of

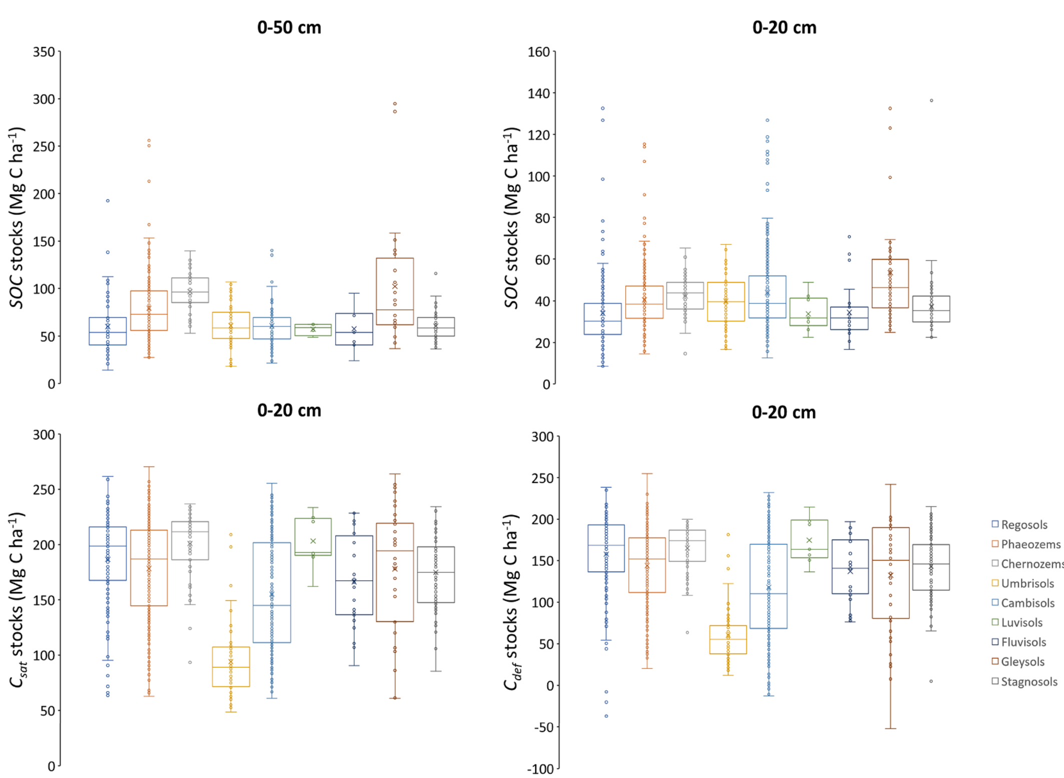

Fig. 3. Boxplots of soil organic carbon (*SOC* stocks) to 50 and 20 cm depth, and carbon saturation potentials (C_{sat}) and deficits (C_{def}) to 20 cm depth in cultivated soils for soil groups according to IUSS Working Group WRB (2014). The *SOC* stock calculations for 0–50 cm depth are based on soil profiles from the regular grid, all other data are based on samples from the regular grid nodes and from the grid centres.

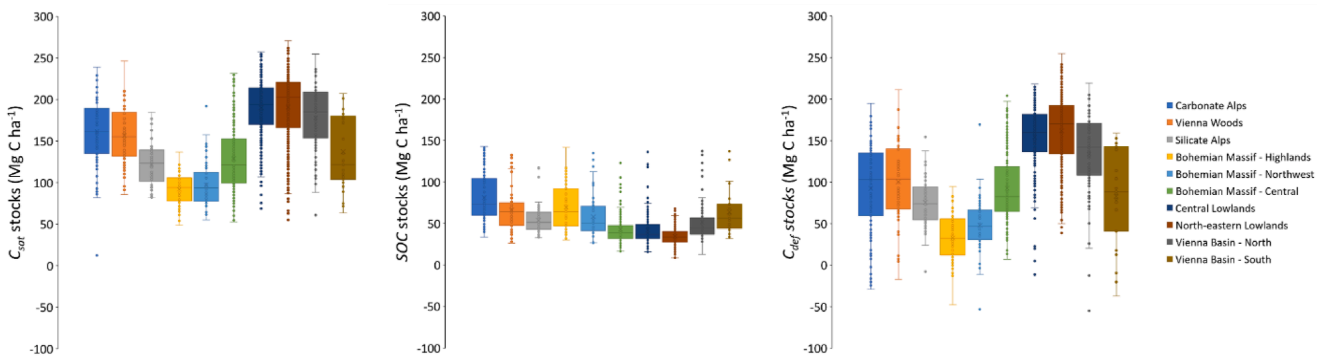

Fig. 4. Stocks of the carbon saturation potential (C_{sat}), soil organic carbon (*SOC*) and the carbon deficit (C_{def}) in topsoils (0–20 cm) of different regions in Lower Austria. According to a Kruskal-Wallis test, there are significant differences ($p < 0.005$) among medians of the stocks of C_{sat}, *SOC* and C_{def}, respectively.

SOM is expected to be controlled by site-specific factors, in particular climate (Wiesmeier et al., 2014, 2019)(;). As the regions differ considerably in climatic conditions (compare section 2.1), we tested how mean annual temperature, precipitation and elevation (as an integrative indicator of both) modify *SOC* concentrations in topsoils. The relations were investigated separately for cultivated and grassland topoils (0–20 cm) using multiple linear regression analysis. For grassland soils, mean annual temperature and precipitation together have little explanatory value ($R^2 = 0.08$, $p < 0.05$, $n = 147$), whereas including $f_{<20\mu m}$ and carbonate content increases the multiple correlation coefficient to $R^2 = 0.33$ ($p < 0.01$). For cultivated soils, the corresponding coefficients of determination are even smaller ($R^2 = 0.14$, $p < 0.01$, $n = 1147$) if all variables are considered. These findings indicate that compared to soil characteristics ($f_{<20\mu m}$ and carbonate) the effect of climate on carbon storage in Lower Austria is small. This agrees with findings of Tan et al. (2004) for Ohio soils who report relevant effects of soil texture but, for non-forest soils, neglectable influence of climate indicated by elevation. Apart from other environmental factors, the large unexplained share of the variation may be due to differences in soil management, however, data are not available to explore this further.

3.6. SOC accumulation in topsoils between 1985/2000 and 2015/2020

The data presented in the previous sections reflect the *SOC* status of the early 1990ies. During the period 2015 – 2020 we re-sampled topsoils from Lower Austrian locations where soil profiles had been collected between 1985 and 2000 for the Austrian soil mapping and the Lower Austrian soil inventory. Table 4 compiles the descriptive statistics of *SOC* concentrations in archived and re-sampled topsoils for the entire dataset, and subsets grouped by regions and land use categories. According to the Wilcoxon signed-rank test (Table 5) the medians of *SOC* concentrations increased significantly ($p < 0.05$) in all regions presented in Table 4 and also in both land use categories. As indicated by the *r* values (Table 5), the effects were moderate to large and thus relevant. Overall, *SOC* concentrations in Lower Austria increased by 14.7% relative to the initial values. The medians of *SOC* in the ecological regions increased by 12.9 – 25.0%, indicating substantial variation of carbon accumulation during the past three decades among regions.

We observe also a distinct difference of C accumulation between the land use categories. The median of *SOC* in the cultivated topsoils increased by 17.1% from 12.7 to 14.8 g kg^{-1} which is still below the proposed upper limit of SOC accumulation (17 – 25 g C kg^{-1}; Fig. 1A) derived from the 90st percentile of *SOC* at the beginning of the monitoring period. Starting from a considerably higher level of 39.4 g kg^{-1}, *SOC* in grassland topsoils increased by 29.7% to 51.1 g kg^{-1} (Table 4).

Our observations indicate relevant *SOC* accumulation in topsoils of both land use categories. As shown in the previous sections, a large share of cultivated as well as grassland topsoils had been considerably depleted of *SOC* at the time of the initial sampling which is indicated by large values of C_{def} (Table 3, Fig. 3, Fig. 4). It is expected that C-depleted

Table 4
Descriptive statistics for organic carbon concentrations (*SOC*) in archived and re-sampled topsoils of ecological regions and for land use categories.

	Minimum	Maximum	Mean	SD[a]	Median	IQR[b]	N[c]
				Archived soils			
All regions and land use categories	3.19	187	24.9	22.0	15.5	20.4	754
Northeastern and central lowlands	4.59	37.0	12.9	4.47	11.9	4.04	443
Vienna Woods-east	13.6	69.6	30.0	14.2	26.2	13.3	23
Silicate Alps	14.3	65.7	32.5	12.5	31.6	19.2	17
Carbonate Alps	12.5	142	48.6	25.2	40.2	27.7	178
Vienna Basin-north	3.19	187	34.4	18.4	29.3	17.4	93
All regions-grassland soils	12.5	143	48.4	26.5	39.4	28.9	195
All regions-cultivated soils	3.19	187	16.8	12.3	12.7	8.27	559
				Re-sampled soils			
All regions	3.57	185	29.0	24.2	17.8	23.4	754
Northeastern and central lowlands	3.57	39.6	15.2	4.07	14.2	2.57	443
Vienna Woods-east	15.4	98.3	35.5	21.3	26.2	24.6	23
Silicate Alps	12.6	94.9	38.4	22.2	32.0	12.2	17
Carbonate Alps	12.7	172	55.8	25.7	50.3	29.9	178
Vienna Basin-north	4.71	185	40.0	27.4	33.1	24.1	93
All regions-grassland soils	4.71	172	56.4	27.1	51.1	31.7	195
All regions-cultivated soils	3.57	185	19.4	13.3	14.8	7.52	559

[a] Standard deviation.
[b] Interqualtile range.
[c] Number of observations.

Table 5
Statistical parameters of the Wilcoxon signed-rank test for the evaluation of significant ($p < 0.05$) differences between medians of *SOC* in archived and re-sampled soils. Differences are considered as significant at IzI >1.96.

	N	T(+)	T(-)	W	μ_W	z	r
All regions and land use categories	754	223,667	−60968	60,968	142,318	−13.6	0.50
Northeastern and central lowlands	443	89,089	−9257	9257	49,173	−14.8	0.70
Carbonate Alps	178	11,351	−4580	4580	7966	−4.91	0.37
Vienna Basin-north	93	3419	−952	952	2186	−4.73	0.49
All regions-grassland soils	195	13,924	−5186	5186	9555	−5.54	0.40
All regions-cultivated soils	559	130,996	−25524	25,524	78,260	−13.8	0.58

soils are generally more responsive to measures for carbon sequestration.

During the monitoring period, management of cultivated soils had been adopted towards more protective systems, supported financially by the Austrian Agro-Environmental Programme (ÖPUL) under the umbrella of the European Common Agricultural Policy (CAP). Measures considered as relevant for the built-up of *SOC* stocks that are funded through this programme include greening, catch crops, diversified crop rotation, reduced tillage, carbon input with manures and compost, and organic farming. The carbon accumulation potential of these management options has been recently evaluated based on a metaanalysis of published data (Tiefenbacher et al., 2021). On average, the potential carbon accumulation rates of most measures for topsoils range between 0.2 and 0.4 Mg C ha^{-1} y^{-1} (Tiefenbacher et al., 2021).

Indicators of improved management of cultivated soils during the monitoring period are compiled in Fig. 5. In 1993, shortly after the first soil sampling, prohibition of open biomass incineration was implemented by Austrian legislation (BGBl. Nr. 405/1993). Before prohibition, burning of straw on fields after harvest had been a common practice of agricultural management, especially in the lowlands with limited water availability for microbial degradation. In 1995, the ÖPUL programme was implemented, with relevant measures for *SOC* accumulation in cultivated soils. The one considered as most important is greening which results in C input that can be accounted for as sequestration (Fig. 5). There are no data available before ÖPUL had been initiated, however, it is known that greening was not a common practice at that time. After implementation of ÖPUL, the share of greening on the total cultivated area of Lower Austria (~700000 ha) has been consistently >35%. Based on long-term field experiments in the region, Freudenschuß et al. (2010) calculated the possible contribution of the greening measures to amount to 0.20 – 0.25 Mg C ha y^{-1}, which is close to the estimated carbon accumulation rate of 0.20 Mg C ha^{-1} y^{-1} observed in the cultivated topsoils (0–20 cm; average bulk density 1390 kg m^{-2}; rock fragment content 5%) during the monitoring period in this study. Along with a steady increase of the proportion of organically-farmed area (Fig. 5), this provides evidence for improved soil management being a main driver of the observed *SOC* accumulation in cultivated soils of Lower Austria since the initial sampling ~30 years ago.

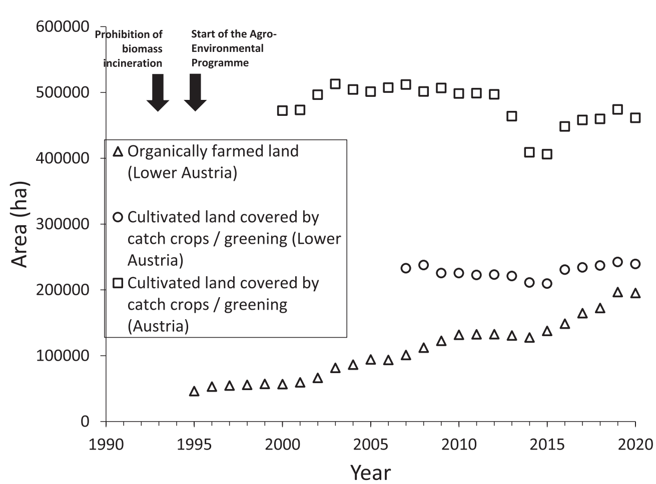

Fig. 5. Indicators of agricultural management likely contributing to *SOC* accumulation during the monitoring period. Data obtained from www.gruenerbericht.at.

The ecological regions dominated by cultivated soils include the central and eastern lowlands and the northern Vienna Basin. Whereas the relative increase of the *SOC* medians is larger in the central and eastern lowlands (19.6%), in absolute terms it was more pronounced in the topsoils of the Vienna basin (Table 4), even though this region was re-sampled after a shorter period (on average after ~23 y, all other regions after ~29 y). Accordingly, the annual accumulation rate in the topsoils of the Vienna basin (0.43 Mg ha^{-1} y^{-1}) exceeds that of the eastern and central lowlands (0.21 Mg ha^{-1} y^{-1}) by a factor of ~2. This difference is not related to C_{def} as the latter is even smaller in the Vienna basin (Fig. 4).

Opposite to our expectation, the observed increases of *SOC* concentration in grassland topsoils were even more pronounced than in the cultivated soils. Note that we only included permanent grassland sites in our calculations. The medians of *SOC* increased by ~30% from 39.4 to 51.1 g kg^{-1} (Table 4), corresponding to an annual accumulation rate of 0.87 Mg ha^{-1} y^{-1} which was calculated by using the average bulk density of Lower Austrian grassland soils (1290 kg m^{-3}) and the average content of rock fragments (16%). This large accumulation rate is surprising as grassland soils are generally closer to saturation than cultivated soils (Fig. 2). According to Six et al. (2002) soils that are closer to saturation should be less responsive to C inputs than soils exhibiting large C_{def}. However, Fig. 2 also shows that at the beginning of the monitoring period a large proportion of the grassland soils had *SOC* concentrations as low as those typical for cultivated soils. Furthermore, owing to the limited availability of archived samples in other regions, the re-sampled grassland soils are mostly derived from the Carbonate Alps with rather large protective capacity (C_{sat}) (Fig. 4), which is related to the accumulation of residual clays from carbonate weathering. Yet, C_{def} is clearly below that of the cultivated soils of the lowland regions (Fig. 4). There is no obvious link of the observed *SOC* accumulation in the grassland soils to changes of management during the monitoring period. Between 2012 and 2017, grassland management in the Carbonate Alps shifted from two harvests (-19.5%) towards permanent grazing (+14.2%), and to a lesser extent towards one or three harvests (Suske et al., 2019). However, the spatial extent of these changes is limited.

Overall, our data indicate that C accumulation during the past three decades was not related to the initial saturation deficit, i.e., not limited by the extent of saturation. This is in contrast to our initial expectation derived from the saturation model of Six et al. (2002) but may be explained by the generally large C_{def} as compared to reports from other regions (McNally et al., 2017; Wiesmeier et al., 2014) and related linear response to C inputs (Kong et al., 2005; Stewart et al., 2008).

It appears that changes in management and land use intensity were generally more effective at *SOC* accumulation in grasslands. Similarly, in regions dominated by cultivation, *SOC* accumulation was larger at smaller C_{def} (Vienna Basin) as compared to the central and north-eastern lowlands, but there is no obvious explanation for this finding.

3.7. Implications for CO_2 mitigation

Based on our data, C_{def} for topsoils of the entire agricultural area of Lower Austria (850113 ha in 2016) is estimated to 115 Tg (own BL equation). The annual total CO_2 emissions of Lower Austria currently amount to ~15 Tg y^{-1} (Anderl et al., 2017), containing ~4 Tg C. The theoretical capacity (C_{def}) of Lower Austrian topsoils to fully compensate for these emissions would last for 29 years. This compares to a measured accumulation rate of ~0.28 Tg C y^{-1} in agricultural topsoils (0–20 cm) of Lower Austria during the past 29 years. Grassland soils contributed ~0.15 Tg C y^{-1} (167626 ha; median $\rho = 1290$ Mg m^{-3}; average rock fragment content of 16%), cultivated soils ~0.13 Tg C y^{-1} (682847 ha; median $\rho = 1390$ Mg m^{-3}; average rock fragment content of 5%). These figures translate to 7% of the annual C emissions being stored in agricultural soils of Lower Austria. However, C *sequestration* rates, i.e., elimination of CO_2 from the atmosphere are likely to be lower as contributions of organic soil amendments such as manures and compost, or biochar, are not included in the definition of C sequestration (Olson et al., 2014).

The observed C accumulation during the past three decades is still small (~7%) compared to the C_{def}, with limited scope for substantial increase without principal changes of land use regimes (Freibauer et al., 2004; Tiefenbacher et al., 2021). However, given the large C_{sat} estimates obtained in our study, the protective capacity of Lower Austrian soils may be larger than expected from published work, which may provide opportunities to set off CO_2 emissions for a limited period until emissions are reduced to acceptable/targeted level.

4. Conclusions

Exploring the data of the Lower Austrian soil inventory and related datasets confirmed previous findings that grassland soils typically contain larger C stocks and lower C_{def} as compared to cultivated soils, even though also many grassland topsoils display large carbon gaps. A detailed, analysis revealed further pedogenetic and land use-related regional pattern of C storage and saturation. Our data show that C_{def} is largely determined by soil texture land use, but less by climatic factors. Accordingly, soil texture and land use category could be used as simple indicators of C_{def} (Wiesmeier et al., 2019). We further show that the observed, substantial *SOC* accumulation during the past three decades, owing to the large *SOC* gap, was not related to C_{def} but rather reflects recent changes in land use intensity and management, such as those implemented through the Austrian Agro-environmental Programme and legislation to prohibit biomass burning on fields. Further monitoring including other Lower Austrian regions with different C_{def} – land use pattern is required to obtain a more comprehensive picture.

Even though reporting on the *SOC* status of a single province of Austria, the results of this study have implications beyond the region as they challenge some of the common concepts of *SOC* research. In contrast to expectations (Six et al., 2002; Stewart et al., 2007; Stewart et al., 2008), it appears that rates of *SOC* accumulation are not necessarily governed by the saturation deficit, probably because most soils were far from saturation. Finally, our monitoring data show relevant *SOC* accumulation which might reflect the situation in various other European regions with similar ecological conditions, landuse pattern and management.

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.

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