Biogeosciences, 15, 703–719, 2018
https://doi.org/10.5194/bg-15-703-2018
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Biogeosciences
Open Access
EGU

Peat decomposability in managed organic soils in relation to land use, organic matter composition and temperature

Cédric Bader[1,2], **Moritz Müller**[3], **Rainer Schulin**[2], **and Jens Leifeld**[1]

[1]Agroscope, Climate and Agriculture Group, Zurich 8046, Switzerland
[2]Inst Terr Ecosyst, ETH Zürich, Zurich 8092, Switzerland
[3]School of Agricultural, Forest and Food Sciences, Bern University of Applied Sciences, Zollikofen 3052, Switzerland

Correspondence: Jens Leifeld (jens.leifeld@agroscope.admin.ch)

Received: 14 May 2017 – Discussion started: 14 June 2017
Revised: 26 October 2017 – Accepted: 13 December 2017 – Published: 5 February 2018

Abstract. Organic soils comprise a large yet fragile carbon (C) store in the global C cycle. Drainage, necessary for agriculture and forestry, triggers rapid decomposition of soil organic matter (SOM), typically increasing in the order forest < grassland < cropland. However, there is also large variation in decomposition due to differences in hydrological conditions, climate and specific management. Here we studied the role of SOM composition on peat decomposability in a variety of differently managed drained organic soils. We collected a total of 560 samples from 21 organic cropland, grassland and forest soils in Switzerland, monitored their CO_2 emission rates in lab incubation experiments over 6 months at two temperatures (10 and 20 °C) and related them to various soil characteristics, including bulk density, pH, soil organic carbon (SOC) content and elemental ratios (C / N, H / C and O / C). CO_2 release ranged from 6 to 195 mg CO_2-C g^{-1} SOC at 10 °C and from 12 to 423 mg g^{-1} at 20 °C. This variation occurring under controlled conditions suggests that besides soil water regime, weather and management, SOM composition may be an underestimated factor that determines CO_2 fluxes measured in field experiments. However, correlations between the investigated chemical SOM characteristics and CO_2 emissions were weak. The latter also did not show a dependence on land-use type, although peat under forest was decomposed the least. High CO_2 emissions in some topsoils were probably related to the accrual of labile crop residues. A comparison with published CO_2 rates from incubated mineral soils indicated no difference in SOM decomposability between these soil classes, suggesting that accumulation of recent, labile plant materials that presumably account for most of the evolved CO_2 is not systematically different between mineral and organic soils. In our data set, temperature sensitivity of decomposition (Q_{10} on average 2.57 ± 0.05) was the same for all land uses but lowest below 60 cm in croplands and grasslands. This, in turn, indicates a relative accumulation of recalcitrant peat in topsoils.

1 Introduction

Organic soils represent a major global sink for atmospheric carbon (C). Although they cover only 3 % of the earth's terrestrial surface (Tubiello et al., 2016), they store up to 30 % of the global soil organic carbon (SOC) pool (Parish et al., 2008). In Europe, more than 50 % of the former peatland area has been degraded by peat mining and conversion of land use, including drainage, to improve their suitability for agriculture or forestry (Joosten, 2010). Drainage aerates the soil so that plants of interest for agriculture and forestry can grow and make these soils manageable. The change from anaerobic to aerobic conditions, however, triggers rapid decomposition of peat that had accumulated under the conditions of waterlogging. This transforms the former C-sink into a major source of atmospheric carbon dioxide (CO_2) and makes peatlands an important contributor to global climate change (Freeman et al., 2004). Around 85 % of the global annual CO_2 emission of 915 Mt CO_2-C from drained peatlands are estimated to originate from organic soils now used as croplands (Tubiello et al., 2016). With rates of 6.5–9.4 t C ha^{-1} a^{-1} net CO_2 fluxes from organic soils now used as croplands were on average found to be higher than from organic soils under grassland, which were estimated to vary

between 1.8 and 7.3 t C ha^{-1} a^{-1} (IPCC, 2014). However, recent studies reported emission rates of 7.6 ± 2.0 t C ha^{-1} a^{-1} on organic soils managed as grassland in Germany and thus much higher rates than previously found for this type of land use (Tiemeyer et al., 2016). Drained organic soils under forest can act as both net sinks or sources of atmospheric CO_2 (Cannell et al., 1993; Minkkinen and Laine, 1998; Minkkinen et al., 1999; Wüst-Galley et al., 2016), although they are in general considered to represent a source with average net CO_2 emissions of 2.0–3.3 t C ha^{-1} a^{-1} in the temperate zone (IPCC, 2014). Temperature and soil moisture regime, which depends on drainage depth, among other factors, have the greatest influence on peat decay in drained organic soils (Hogg et al., 1992; Berglund, 1995; Scanlon and Moore, 2000; Chimner and Cooper, 2003; Couwenberg et al., 2010; Leifeld et al., 2012). However, there are substantial differences in CO_2 emissions from organic soils with similar drainage and cultivation properties. The protection of organic matter (OM) against decomposition by mechanisms such as occlusion in aggregates and binding to mineral surfaces, which are important for the stabilization of OM in mineral soils (Six et al., 2002), are of minor importance in organic soils due to the lack or low abundance of minerals (Han et al., 2016). Therefore, the intrinsic decomposability of organic matter is considered another major factor influencing the rate of peat decomposition and a major cause of substantial variation in CO_2 emissions at different sites (Chimner and Cooper, 2003; Byrne and Farrell, 2005; Höper, 2007; Wickland and Neff, 2008; Reiche et al., 2010).

Although intrinsic decomposability of SOM cannot be addressed directly, useful indicators of the latter are the relative abundances of labile and recalcitrant C moieties, which shift towards progressively higher proportions of the recalcitrant C with decomposition (Beer et al., 2008; Tfaily et al., 2014) and result in selective enrichment and depletion of specific functionalities (Leifeld et al., 2017; McAnallen et al., 2017). It is important to recognize that during peat formation, most of the net primary production contained in the initial mass of plant residues are lost due to mineralization, and only 10–20 % is transformed and accumulated as peat in the water-saturated zone of a peat bog or fen (Clymo, 1984). Although decomposition acts slowly on accumulating peat of undisturbed (i.e. water-saturated organic) soils, it is believed that primarily the most labile OM moieties are lost. Due to fresh peat layers accumulating on top of older ones, age and depletion in labile compounds increase with soil depth. Incubation studies of peat samples and carbon loss studies with undisturbed organic soils found smaller CO_2 emission rates from deeper peat layers, which was related to the absence of labile compounds i.e. a lower intrinsic decomposability of soil organic matter (SOM) (Hogg et al., 1992; Scanlon and Moore, 2000; Wang et al., 2010; Hardie et al., 2011; Leifeld et al., 2012). Using solid-state ^{13}C-NMR, DRIFT/FTIR spectroscopy and pyrolysis-GC/MS, various studies of OM composition of undisturbed peat profiles have shown a gradual change with increasing depth towards a relative enrichment of compounds that are recalcitrant against decomposition under anoxic conditions, such as lignins and polyphenols (Freeman et al., 2004), while the contents of labile oxygen-rich compounds, such as polysaccharides, were found to decrease (Leifeld et al., 2012; Biester et al., 2014; Sjögersten et al., 2016).

Elemental ratios of oxygen (O), hydrogen (H), and nitrogen (N) to carbon are widely used as indicators of the relative abundance of different groups of compounds such as phenols, lipids and polysaccharides, and proteins. Lignins and polyphenols have molar O / C ratios in the range of 0.2–0.6 and H / C ratios between 0.9 and 1.5, while the respective ratios of carbohydrates range from 0.8 to 0.9 for O / C and from 1.4 to 1.8 for H / C (Kim et al., 2003). In line with the molecular and spectroscopic analyses mentioned before, both ratios were found to decrease with increasing depth in peat (Klavins et al., 2008; Biester et al., 2014; Wüst-Galley et al., 2016). On the other hand, both fresh plant residues and undisturbed peat usually have high C / N ratios (Loisel et al., 2012). When peat becomes exposed to oxic conditions, mineralization seems to lead to relative enrichment of N, explaining why decreased C / N ratios are found in organic topsoils compared to undrained peat layers or bottom layers of drained organic soils (Malmer and Holm, 1984; Kuhry and Vitt, 1996; Krueger et al., 2015). While undisturbed organic soils have a low bulk density, drainage leads to subsidence processes and increasing bulk densities in the topsoils (Rogiers et al., 2008; Leifeld et al., 2011a, b).

The temperature sensitivity of peat mineralization, as expressed by its Q_{10} value, is a useful parameter for characterizing the intrinsic decomposability of SOM (Hogg et al., 1992; Biasi et al., 2005; Davidson and Janssens, 2006; Conant et al., 2008; Boddy et al., 2008; Karhu et al., 2010; Hilasvuori et al., 2013). In line with the biochemical and elemental evidence reviewed above, it was reported to increase with increasing resistance of peat soils against OM decomposition (Scanlon and Moore, 2000), soil depth and peat age (Hardie et al., 2011; Hilasvuori et al., 2013).

Despite its likely important role in determining future C losses from drained peatland, the influence of SOM composition on peat decomposition in managed organic soils is not well studied. While decomposition rates seem to decline with increasing peat age, i.e. profile depth, the oxic conditions, occurring after drainage onset, lead to fast SOM decomposition. As for undisturbed organic soils, we expect that post drainage decomposition primarily acts on the most labile OM moieties. However, the much faster decomposition of labile SOM might alter the depth interaction found in undisturbed peat soils. Further, recent inputs from plant residues may supply the topsoils with labile OM. Around 20 % of carbon in organic soils under agriculture is derived from crop residues and thus decomposes more rapidly (Bader et al., 2017). The fractions of OM derived from peat and recent inputs and their decomposability in drained organic soils may, however, sub-

stantially vary with land use, site conditions and time since land use conversion. Schulze et al. (2009) reported that inputs of fresh organic matter residues were smaller in croplands than in grasslands or forests, suggesting that SOM might be on average more aged and thus less decomposable. In situ measurements of CO_2 fluxes from managed organic soils reveal slower decomposition of peat under forest (IPCC, 2014). Together, smaller peat loss rates and higher residue input make us expect that SOM decomposition rates under controlled conditions are fastest in forest topsoils.

In this study, we analysed the relationship between SOM properties, specific decomposition rates (CO_2-C mg^{-1} SOC) and their temperature sensitivities to peat samples taken from depths between 0 and 200 cm of 21 drained organic soils in Switzerland managed as cropland, perennial grassland or forest. These sites embody three major uses for drained peatlands as they occur in Europe (Joosten, 2010) and are also representative of the situation in Switzerland where most former peatlands are drained and managed (Wüst-Galley et al., 2015). We measured decomposition rates in incubation experiments under standardized lab conditions and interpreted the current decomposition status of peat using SOM properties such as (i) carbon stocks, bulk densities and the elemental ratios O / C, H / C and C / N as well as (ii) the temperature sensitivity towards decomposition, expecting that

1. specific decomposition rates of SOM decline with depth,

2. specific decomposition rates of SOM in managed organic soils correlate with its composition and are inversely related to the temperature sensitivity of decomposition,

3. specific decomposition rates of topsoil SOM are largest in the forest and smallest in the croplands.

2 Methods

2.1 Sampling sites

The soil samples used for this study were taken from organic soils distributed across Switzerland that were identified using the map of Wüst et al. (2015). Apart from current land use (grassland, cropland, forest), they differed in the type of drainage system (ditches in forest, pipes in crop- and grassland), time since drainage onset and drainage intensity, altitude (MASL) and climate (Table 1). All sites were classified as fens, although we found bog-derived peat layers within the top 30 and 40 cm of the soil profiles at two sites (SK_FL, K_FL). Cropland management comprised rotations typical for Switzerland with maize, winter wheat, ley and rapeseed as major crops. Sites were conventionally tilled. Grasslands were used for cutting and haymaking, not grazing, and are fertilized according to the Swiss Fertilization Recommendations (Flisch et al., 2009). Forest sites were managed and their vegetation was not peat-forming.

2.2 Soil sampling

Between October 2013 and June 2015, we sampled a total of 84 peat cores from all 21 sites (4 cores per site). All cores were taken to a maximum depth of 1 m. If the underlying mineral layer was reached before 1 m depth, coring was discontinued. We used a Belarusian peat corer (cuts a half-cylindrical undisturbed core of diameter 4 cm) for soils with low bulk densities and a motorized Humax corer (cuts a cylindrical core of diameter 5 cm) for denser soils. The samples were stored at 4 °C for up to 2 months until analysis. We applied the method of Rogiers et al. (2008) to account for soil compaction during sampling for any sample and divided the cores into segments corresponding to 5–10 cm depth increments. This corresponded, depending on the type of soil corer used and length of the increment, to sample volumes of between 31 and 196 cm^3 per segment. In total this resulted in 1605 soil samples. Some cores had interlayers of mineral sediment identified by a different colour (grey), a high bulk density and their SOC content was lower than 150 g kg^{-1}. These interlayers were excluded from the analysis. The soil of one site (BI_FL) had no limnic layer and therefore was classified as a murshic Histosol; all others were classified as murshic limnic Histosols (WRB, 2014).

2.3 Soil analysis

Soil pH was measured for two to three samples of each core (307 samples in total) using a flat surface electrode (pH 100, Extech Instruments, USA) calibrated at pH 7.00 and pH 4.01. Aliquots of fresh soil (10 g dry matter) were diluted in distilled water (2.5 parts water to 1 part material by mass), shaken, left for 20 h and shaken again, before the pH measurements were carried out.

Prior to further chemical analysis, the samples were oven-dried at 105 °C and weighed to determine bulk density (g cm^{-3}). The dried samples were ground for 2 min at 25 rotations s^{-1} in a ball mill (Retsch MM400) and subsampled to determine total carbon (Ctot), SOC, hydrogen (H), nitrogen (N) and oxygen (O) contents. Ctot, H and N were analysed after dry combustion of ground subsamples in an elemental analyser (Hekatech, Germany). To determine SOC, we hydrolysed ground aliquots with 36 % HCl (acid fumigation) in a desiccator to remove any carbonates before the samples were analysed in the elemental analyser. A third set of ground subsamples were used to determine the O contents by means of the same analyser after pyrolysis at 1000 °C. We corrected O contents for inorganic O, assuming that all inorganic O was present in form of $CaCO_3$. The O / C and H / C ratios given in this paper represent mole ratios, whereas the C / N ratios represent mass ratios. For analysis O / C ratios and H / C ratios of samples having a SOC content lower than

Table 1. List of sampling locations, including information on the land-use type, peat thickness, approximate time since drainage onset, elevation (MASL), mean annual temperature (MAT) and mean annual precipitation (MAP) of each site.

Location name	Abbreviation of location with land use (CL: cropland, GL: grassland, FL: forest)	Co-ordinates; WGS (1984)	MASL (m)	Area size (ha)	Peat thickness[1] (cm)	Drainage history[2]	Drainage class[3]	MAT[4] (°C)	MAP[5] (mm)
Birmensdorf	BI_FL	8.454, 47.357	560	2.6	95	Unclear; peat excavation nearby	s	9.2	1122
Brüttelen	B_CL	7.175, 47.033	438	3.1	290	Drained by 1864	d	9.9	1003
Chreienriet	F_FL	8.486, 47.434	440	6.9	330	Unclear; peat excavation site nearby until 1940	d	9.4	1040
Cressier	C_CL, C_GL	7.047, 47.041	430	1.6, 1.6	120	Drained by 1864	d	10.0	1145
Gals	G_CL, G_GL, G_FL	7.065, 47.040	430	1.2, 0.8, 1.0	< 100	Drained by 1864	d	10.0	1145
Im Moos	IM_CL	9.573, 47.379	414	5.6	400	Drained by 1860; intensive drainage between 1942 and 1962	d	10.1	1297
Katzensee	K_FL	8.495, 47.433	440	1.9	230	Unclear; peat excavation site nearby until 1940	s	9.4	1040
Kirchenthurnen	K_GL	7.523, 46.821	540	9.9	302	Drained after 1860	d	8.9	1136
Lüchingen	L_CL	9.574, 47.378	414	4.4	400	Drained by 1860; intensive drainage between 1942 and 1962	d	10.1	1297
Mühlethurnen	M_CL, M_GL	7.523, 46.821 7.523, 46.817	540	8.2, 7.6	400	Drained after 1860; intensive drainage in 1942	d	8.9	1136
Rüthi	R_GL	9.536, 47.283	435	13.3	> 700	Drained by 1970	d	10.1	1533
Staatswald 1 + 2	SW1_FL, SW2_FL	7.092, 46.984	431	30.0, 48.4	142	drained by 1864; intensive drainage in 1942	d	10.1	990
Summerigchopf	SK_GL, SK_FL	9.399, 47.212	1300	11.5, 2.2	147–202	Drain established between 1935 and 1960	s	6.0	1731
Treiten	T_CL	7.145, 47.010	439	29.3	238	Drained by 1864	d	9.9	1033
Vorderwengi	VW_GL, VW_FL	9.098, 47.196	1070	1.1, 0.9	100–146	grassland drained by 1935	s	6.2	2240

[1] Peat thickness was determined by excavation of an additional peat core down to the underlying sediment layer. [2] Information on drainage was gained by viewing Siegfried topographical maps (1870–1949), considering information on Swiss organic soils by Lüdi (1935) as well as aerial photographs. [3] Shallow drainage < 0.5 m s, deep drainage > 0.5 m d. [4] MAT is the average for the years 1981–2010. [5] MAP is the average of the years 1971–1991 derived from original data of MeteoSchweiz.

150 g kg^{-1} were excluded from analysis. Soil carbon stocks (t C ha^{-1}) refer to the organic horizons summed over each profile and thus do not include sediment layers that interspersed the profiles.

2.4 Incubation experiment

We selected at least two soil segments of each soil core from depths between 0–30, 30–60 and 60–100 cm for incubation to determine SOM decomposability. Each segment was divided in two subsegments, whereas one subsegment was incubated at 10 °C and the other subsegment at 20 °C for between 6 and 13 months. From the one location (M_CL) where we had taken cores of > 100 cm length, we selected six additional samples from the depth below 100 cm for incubation, resulting in a total of 560 incubated samples. Prior to incubation, we thoroughly mixed every segment, removed visible roots and adjusted the water potential to −10 kPa using a hanging water column. The sample weight was 53.9 ± 0.7 g (mean ± standard error) at −10 kPa. Following the method of Chapman (1971), we measured CO_2 emission rates by means of a Respicond VII analyser (Nordgren Innovation, Sweden) over three to four measurement cycles of several weeks between November 2013 and March 2016. The measurement principle is based on the change in electrical conductivity of the NaOH solution with increasing uptake of CO_2. In each cycle, we vented the alkali CO_2 traps (NaOH 0.6 M) of the analyser regularly after 50–60 mg of CO_2 had been emitted to prevent O_2 deficiency. In addition, we exchanged the NaOH solution while the traps were vented. Between measurement cycles, we kept the soil samples at the same temperature and moisture level as during the cycles.

2.5 Data analysis

We only used CO_2 data taken after the first 3 days of each measurement cycle for data analysis to avoid artefacts that might have resulted from moving the samples and adjusting their water content. Furthermore, we excluded all negative emission rate values (0.45 % out of 1700 CO_2 measurements taken on average per sample). Data gaps (83 % of the timeline) between measurement cycles were filled by means of interpolation using a robust linear regression on the log-transformed data. The specific amount of SOC which was emitted from a sample as CO_2 during 10 000 h of incubation at 10 or 20 °C [mg CO_2-C g^{-1} SOC], L, was calculated as

$$L = \frac{(CO_{2\ \mathrm{sample}} - CO_{2\ \mathrm{blank}}) \times \frac{12.01}{44.01}}{SOC_{\mathrm{sample}} \times m_{\mathrm{sample}}}, \quad (1)$$

where $CO_{2\ \mathrm{sample}}$ is the amount of CO_2 emitted from the sample over 10 000 h of incubation [mg CO_2-C g^{-1} SOC], $CO_{2\ \mathrm{blank}}$ is the median of ambient CO_2 accumulation collected in six blank vessels over more than 6 months and extrapolated to 10 000 h (on average 27 mg), SOC_{sample} is the SOC content of the sample [g kg^{-1}], and m_{sample} is the mass of the soil sample [kg].

To determine Q_{10} values we applied the method used by others (e.g. Hogg et al., 1992; Scanlon and Moore, 2000; Wang et al., 2010; Wetterstedt et al., 2010; Hardie et al., 2011), dividing the 10 000 h length of the incubation period at 10 °C by the time span over which samples incubated at 20 °C emitted the same amount of CO_2-C per mg SOC as those incubated at 10 °C emitted during 10 000 h. Given that the same amount of SOC is lost at both temperatures, changes in OM composition during incubation are also assumed to be the same and thus differences in the rates are assumed to reflect only the influence of temperature and not that of differences in composition. Q_{10} values are known to depend on incubation temperatures. In order to compare our results with those of other studies we calculated the activation energy (Ea in kJ mol^{-1}) required for decomposition of SOC using Q_{10} values.

While R is the gas constant (8.314 J K^{-1} mol^{-1}) and T is the temperature used for incubation (K).

$$\mathrm{Ea} = \frac{R \times \frac{\ln(Q10)}{(\frac{1}{T1} - \frac{1}{T2})}}{1000} \quad (2)$$

Mixed linear models were used to analyse the effects of the various soil parameters on SOM mineralization and their interactions with land use. The function lmer from the package lme4 (Bates et al., 2015) was implemented using the software R (R core Team, 2015) to run mixed linear models. Heteroscedasticity or departure from normality was assessed graphically. In order to avoid heteroscedasticity, we log-transformed topsoil C stocks and bulk density data. We tested whether the factor "land use" had a significant influence on the variation of each of the analysed variables ($\alpha = 0.05$). To do this, the following two mixed models, 2 and 3, were run for each dependent variable and compared using an ANOVA.

$$\text{variable} \sim \text{land.use} + \text{random effects} \quad (3)$$

$$\text{variable} \sim \text{random effects} \quad (4)$$

Sampling depth, sampling location and site repetition were included as random effects to account for the dependence among segments of the same core and among cores from the same sampling location, respectively. In addition, we included bulk density, SOC, nitrogen, hydrogen and oxygen contents as well as the emitted CO_2 as further random effects, given that there was no collinearity with the tested variable and that the Akaikes criterion (AIC) of the models revealed smaller scores with additional random effects. The additional random effects used for each model are given in Tables 2 and 3. Further, we determined the significance of land-use-specific differences (CL vs. FL, CL vs. GL, FL vs. GL) using a least square means test for linear models (lsmeans package).

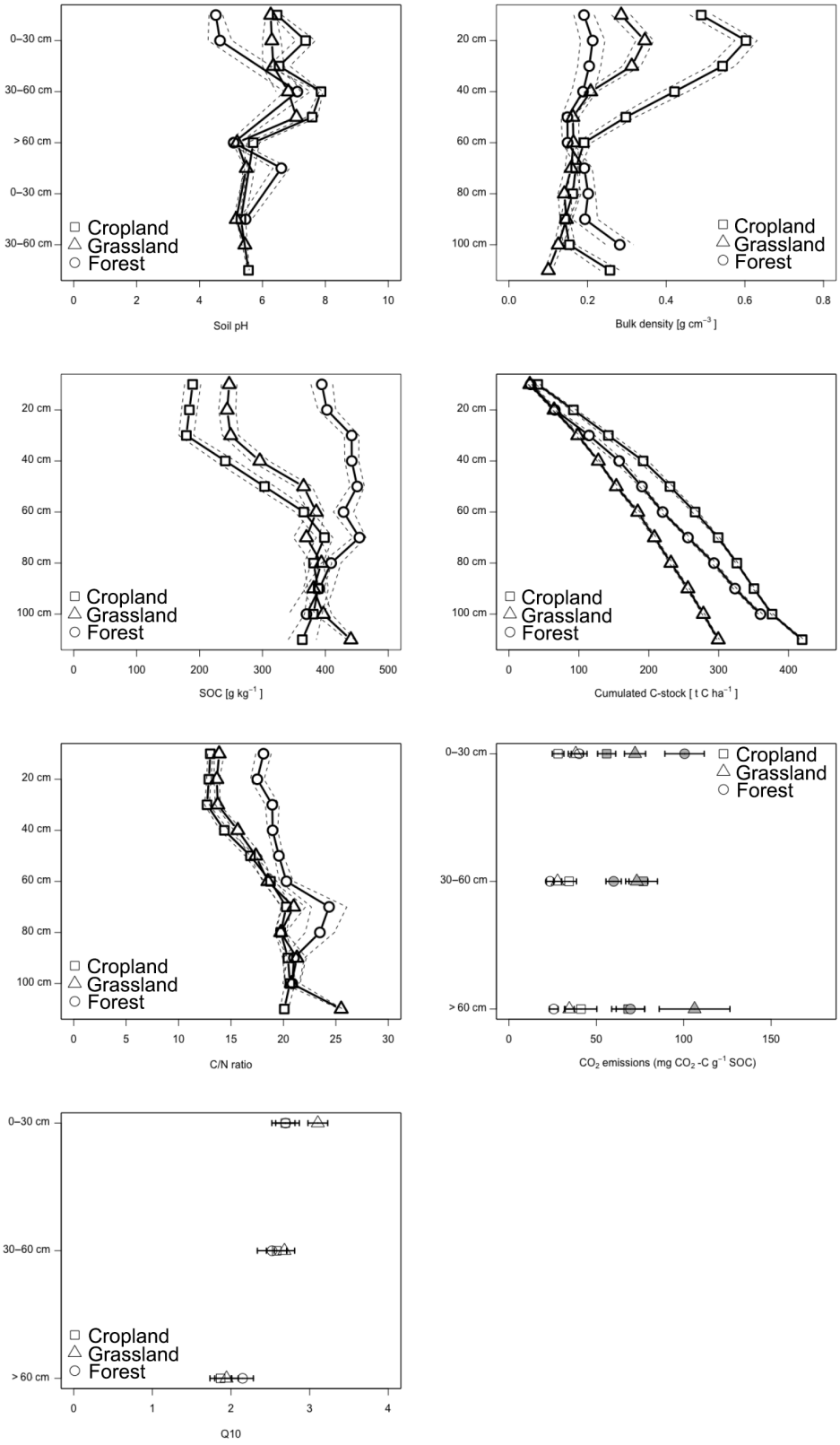

Figure 1. Soil pH, bulk density, SOC content, cumulated C stocks, C / N ratios, CO_2 emissions and temperature sensitivity (Q_{10}) displayed for the three land-use types (cropland, grassland and forest) in relation to the profile depth (cm, *y* axis). CO_2 emissions are displayed at 10 (open symbols) and 20 °C (black symbols), while the area between dashed lines and error bars represents the standard errors of the mean.

Table 2. Results of land-use effect analysis for the whole soil profile as well as specifically in the topsoil (0–30 cm) and bottom layers (> 30 cm), displayed for SOC concentration, C stocks, bulk density C / N, H / C and O / C ratios, CO_2 emissions at 10 and 20 °C and the resulting Q_{10} values.

	Land-use interaction[1]		*P* values between specific land-uses[2]		
Attribute	χ^2 value	*P* value	CL vs. FL	CL vs. GL	FL vs. GL
Soil pH	$\chi^2(2) = 3.7$	0.16			
Soil pH (0–30 cm)	$\chi^2(2) = 14.9$	0.0006	0.0003	0.9	0.0021
Soil pH (> 30 cm)	$\chi^2(2) = 0.7$	0.7			
SOC	$\chi^2(2) = 10.7$	0.005	0.14	0.28	0.002
SOC (0–30 cm)	$\chi^2(2) = 14.5$	0.0001	0.0001	0.5	0.009
SOC (> 30 cm)	$\chi^2(2) = 3.0$	0.2			
Cumulative C stock					
C stock (0–30 cm)	$\chi^2(2) = 5.4$	0.07	0.06	0.4	0.6
C stock (0–100 cm)	$\chi^2(2) = 5.4$	0.06	0.2	0.06	0.8
Bulk density	$\chi^2(2) = 3.4$	0.2			
Bulk density (0–30 cm)	$\chi^2(2) = 10.3$	0.06	0.02	0.09	0.4
Bulk density (> 30 cm)	$\chi^2(2) = 2.0$	0.4			
C / N ratio	$\chi^2(2) = 5.9$	0.05	0.06	0.9	0.1
C / N ratio (0–30 cm)	$\chi^2(2) = 15.0$	0.0005	0.0002	0.8	0.003
C / N ratio (> 30 cm)	$\chi^2(2) = 2.2$	0.3			
H / C ratio	$\chi^2(2) = 6.7$	0.04	0.5	0.4	0.02
H / C ratio (0–30 cm)	$\chi^2(2) = 6.3$	0.04	0.6	0.4	0.03
H / C ratio (> 30 cm)	$\chi^2(2) = 3.5$	0.2			
O / C ratio	$\chi^2(2) = 11.5$	0.003	0.4	0.7	0.001
O / C ratio (0–30 cm)	$\chi^2(2) = 10.5$	0.005	0.06	0.7	0.003
O / C ratio (> 30 cm)	$\chi^2(2) = 8.5$	0.014	0.008	1.0	0.003
CO_2 10°C	$\chi^2(2) = 2.4$	0.3			
CO_2 10 °C (0–30 cm)	$\chi^2(2) = 2.9$	0.2			
CO_2 10 °C (30–60 cm)	$\chi^2(2) = 7.17$	0.03	0.023	0.38	0.34
CO_2 10 °C (> 60 cm)	$\chi^2(2) = 1.6$	0.4			
CO_2 20**°C**	$\chi^2(2) = 1.4$	0.5			
CO_2 20 °C (0–30 cm)	$\chi^2(2) = 6.5$	0.04	0.03	0.2	0.7
CO_2 20 °C (30–60 cm)	$\chi^2(2) = 1.7$	0.4			
CO_2 20 °C (> 60 cm)	$\chi^2(2) = 1.2$	0.5			
Q_{10}	$\chi^2(2) = 3.5$	0.2			
Q_{10} (0–30 cm)	$\chi^2(2) = 0.4$	0.8			
Q_{10} (30–60 cm)	$\chi^2(2) = 1.1$	0.6			
Q_{10} (> 60 cm)	$\chi^2(2) = 1.0$	0.6			

[1] *P* value of ANOVA comparing linear mixed models with and without the factor "land-use type".
[2] *P* value emitted using least square means between land-use types.

We used the same approach to test the influence of the factor "soil depth" on the target variables with interactions between the three sampling depths (0–30, 30–60, > 60 cm) using the model

$$\text{variable} \sim \text{depth.interval} + \text{random effects} \quad (5)$$

in addition to Eq. (3). To determine the significance of depth-specific differences, we used a least square means test as mentioned before.

Table 3. Results of the depth influence analysis displayed for Q_{10} values, CO_2 emissions at 10 and 20 °C, SOC contents, bulk densities, C / N ratios, H / C ratios and O / C ratios. Ea values (not shown) had similar significance to Q_{10} values.

	Depth interaction		*P* values between specific depth classes		
Attributes	χ^2 values	*P* value	0–30 vs. 30–60	0–30 vs. > 60	30–60 vs. > 60
Q_{10} values	$\chi^2(2) = 46.2$	9.56×10^{-11}	0.05	< 0.0001	< 0.0001
Q_{10} cropland	$\chi^2(2) = 16.1$	0.0003	0.8	0.0002	0.002
Q_{10} forest	$\chi^2(2) = 5.2$	0.08			
Q_{10} grassland	$\chi^2(2) = 29.5$	3.9×10^{-7}	0.06	< 0.0001	0.009
CO_2 emission (10 °C)	$\chi^2(2) = 6.1$	< 0.05	0.03	0.7	0.2
Cropland (10 °C)	$\chi^2(2) = 1.5$	0.5			
Forest (10 °C)	$\chi^2(2) = 17.3$	0.0001	0.0001	0.01	0.5
Grassland (10 °C)	$\chi^2(2) = 7.9$	0.02	0.01	0.3	0.5
CO_2 emission (20 °C)	$\chi^2(2) = 0.9$	0.6			
Cropland (20 °C)	$\chi^2(2) = 8.4$	0.015	0.02	1.0	0.09
Forest (20 °C)	$\chi^2(2) = 13.2$	0.0001	0.0007	< 0.05	0.6
Grassland (20 °C)	$\chi^2(2) = 3.5$	0.17			
pH	$\chi^2(2) = 6.0$				
Cropland	$\chi^2(2) = 19.4$	6.2×10^{-5}	< 0.02	< 0.0001	0.09
Forest	$\chi^2(2) = 36.8$	1×10^{-8}	0.004	< 0.0001	0.001
Grassland	$\chi^2(2) = 27.4$	1.1×10^{-6}	0.0001	< 0.0001	0.9
SOC	$\chi^2(2) = 157.7$	$< 2.2 \times 10^{-16}$	< 0.0001	< 0.0001	0.0001
Cropland	$\chi^2(2) = 158.2$	$< 2.2 \times 10^{-16}$	< 0.0001	< 0.0001	< 0.0001
Forest	$\chi^2(2) = 3.8$	0.15	0.2	1.0	0.3
Grassland	$\chi^2(2) = 143.2$	$< 2.2 \times 10^{-16}$	< 0.0001	< 0.0001	< 0.0001
Bulk density	$\chi^2(2) = 57.6$	$< 3.1 \times 10^{-13}$	< 0.0001	< 0.0001	0.9
Cropland	$\chi^2(2) = 312.6$	$< 2.2 \times 10^{-16}$	< 0.0001	< 0.0001	< 0.0001
Forest	$\chi^2(2) = 31.6$	1.4×10^{-7}	0.7	< 0.0001	< 0.0001
Grassland	$\chi^2(2) = 49.9$	1.4×10^{-11}	< 0.0001	< 0.0001	0.08
C / N ratio	$\chi^2(2) = 325.5$	$< 2.2 \times 10^{-16}$	< 0.0001	< 0.001	< 0.0001
Cropland	$\chi^2(2) = 199$	$< 2.2 \times 10^{-16}$	< 0.0001	< 0.001	< 0.0001
Forest	$\chi^2(2) = 41.2$	1.5×10^{-9}	0.4	< 0.001	< 0.0001
Grassland	$\chi^2(2) = 152.8$	$< 2.2 \times 10^{-16}$	< 0.0001	< 0.001	< 0.0001
H / C ratio	$\chi^2(2) = 19.9$	$< 4.7 \times 10^{-5}$	< 0.0001	< 0.0002	0.7
Cropland	$\chi^2(2) = 46.7$	7.3×10^{-11}	< 0.0001	< 0.0001	0.3
Forest	$\chi^2(2) = 1.9$	0.38			
Grassland	$\chi^2(2) = 52.6$	3.7×10^{-12}	< 0.0001	< 0.0001	0.07
O / C ratio	$\chi^2(2) = 22.0$	0.0005	< 0.06	0.03	0.9
Cropland	$\chi^2(2) = 0.01$	1.0			
Forest	$\chi^2(2) = 2.6$	0.3			
Grassland	$\chi^2(2) = 6.3$	0.04	0.03	0.5	0.5

Table 4. Coefficients of determination and correlation for CO_2 emissions measured at 20 °C and different soil attributes as explanatory variables (profile depth, SOC content, bulk density, C / N, O / C and H / C ratio). Ea values (not shown) behaved similarly to Q_{10} values.

Attribute	0–30 cm				30–100 cm			
CO_2 at 20 °C	Intercept	cor	P value	R^2	Intercept	cor	P value	R^2
Depth (cm)	$< 2.0 \times 10^{-16}$	−0.23	0.01	0.05	0.0001	0.11	0.2	0.01
SOC ($g\,kg^{-1}$)	8.06×10^{-6}	0.31	0.001	0.09	7.6×10^{-6}	−0.01	0.9	4.8×10^{-5}
Bulk density ($g\,kg^{-1}$)	$< 2.0 \times 10^{-16}$	−0.27	0.003	0.07	$< 2 \times 10^{-16}$	−0.19	0.02	0.04
C / N ratio	0.002	0.28	0.002	0.08	3.5×10^{-6}	−0.1	0.5	0.004
O / C ratio	1.33×10^{-6}	−0.02	0.9	0.0002	2.1×10^{-5}	0.06	0.4	0.01
H / C ratio	0.046	0.03	0.8	0.0005	0.292	0.07	0.4	0.01
pH	2.3×10^{-8}	−0.29	0.001	0.09	2.3×10^{-8}	−0.25	0.003	0.06
CO_2 at 10 °C								
Depth (cm)	1.7×10^{-13}	0.19	0.2	0.02	0.0002	0.10	0.2	0.02
SOC ($g\,kg^{-1}$)	1.7×10^{-5}	−0.13	0.04	0.03	4.8×10^{-16}	−0.24	0.002	0.06
Bulk density ($g\,kg^{-1}$)	$< 2.0 \times 10^{-16}$	−0.22	0.02	0.04	$< 2.0 \times 10^{-16}$	−0.01	0.9	9.1×10^{-5}
C / N ratio	0.0007	0.31	0.0007	0.09	1.05×10^{-5}	−0.03	0.7	0.0008
O / C ratio	3.8×10^{-15}	0.01	1.0	3.4×10^{-5}	3.4×10^{-5}	0.10	0.2	0.01
H / C ratio	0.5	0.14	0.1	0.2	0.3	0.24	0.002	0.06
pH	6.6×10^{-5}	−0.14	0.1	0.02	0.0001	−0.09	0.3	0.007
Q_{10} values								
Depth (cm)	$< 2.0 \times 10^{-16}$	0.18	0.12	0.01	0.0001	−0.30	0.0001	0.08
SOC ($g\,kg^{-1}$)	$< 2.0 \times 10^{-16}$	−0.16	0.03	0.03	$< 2.0 \times 10^{-16}$	0.12	0.06	0.02
Bulk density ($g\,kg^{-1}$)	$< 2.0 \times 10^{-16}$	−0.02	0.6	−0.006	$< 2.0 \times 10^{-16}$	−0.02	0.4	−0.002
C / N ratio	8.1×10^{-16}	−0.10	0.15	0.01	1.2×10^{-15}	−0.08	0.4	−0.002
O / C ratio	5.3×10^{-15}	0.20	0.07	0.02	$< 2.0 \times 10^{-16}$	−0.13	0.3	0.0002
H / C ratio	0.0001	0.06	0.2	0.004	2.9×10^{-8}	−0.13	0.06	0.02
pH	1.02×10^{-9}	−0.03	0.8	0.0006	2.7×10^{-8}	−0.02	0.8	0.0003

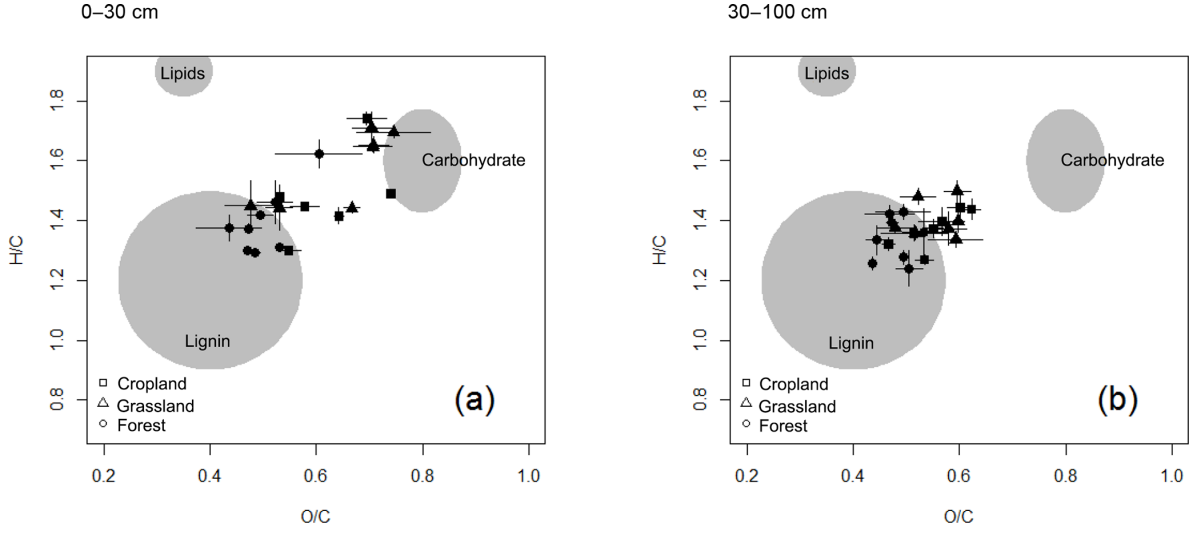

Figure 2. Van Krevelen plots of samples from **(a)** the upper 30 cm and **(b)** depths below 30 cm. Symbols represent averages for relevant core segments from each site, black bars represent the standard error of the mean, grey surfaces represent the range of O / C and H / C for lignin, carbohydrates and lipids, adapted from Preston and Schmidt (2006).

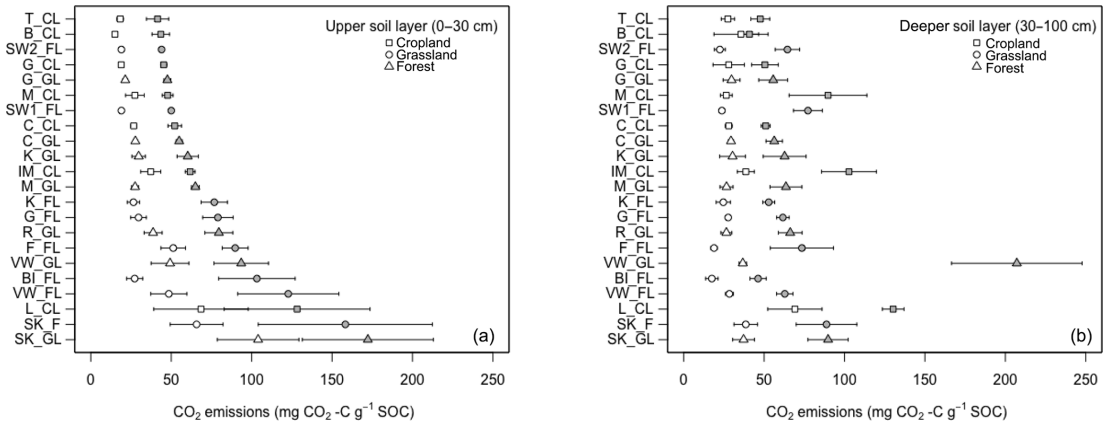

Figure 3. CO_2 emissions at 10 (open symbols) and 20 °C (grey symbols) displayed for upper soil layers (0–30 cm) and bottom layers (30–100 cm) of all sampling locations. Sampling locations are sorted from the lowest to highest CO_2 emissions in **(a)**. Same order of sites was taken for **(b)**. Error bars represent the standard error of the mean. If a symbol lacks error bars, the standard error was smaller than the symbol size or, as in the cases of upper soil layers SW1 and SW2, $n = 2$.

3 Results

3.1 SOM characteristics

Soil pH, SOC content, C / N ratio and bulk density showed significant land-use effects (Fig. 1, Table 2, Table S1). The lowest soil pH values were found in the forest topsoil samples, whereas SOC content and C / N ratio were the highest in these samples. Bulk density was highest in the cropland topsoils. Below 30 cm depth, soil pH, SOC content, C / N ratio and bulk density showed no land-use effect.

In the forest soil profiles, soil pH overall increased with depth, whereas it decreased in the grassland and cropland soils (Table 3). Also bulk density decreased with depth in the grassland and cropland soils, while SOC content and C / N ratio increased. In the forest soils, SOC content, bulk density and C / N ratio did not differ between topsoil (0–30 cm depth) and subsoil samples (30–60 cm depth); however, below 60 cm depth SOC was slightly lower than above, while bulk density and C / N ratio were higher than above 60 cm depth (Fig. 1, Table 3). The cumulated topsoil C stocks showed no land-use effects but tended to be larger in cropland and forest than in grassland soils over the entire profile (Fig. 1, Table 2).

The molar H / C and O / C ratios of the organic matter fell between the typical values of the ratios for carbohydrates and lignin, which is displayed in a Van Krevelen plot (Fig. 2). The lowest values of both ratios were found in the forest soils, the highest in the grassland and cropland topsoils. Both ratios were lower in the topsoils than in the subsoils of the cropland and grassland sites, while there was no difference between the two depths in the forest soils (Table 3). At depths below 30 cm, the O / C ratio was lower in the forest soils than in the other soils but without a land-use effect in the H / C ratio.

3.2 CO_2 emissions and Q_{10}

The samples incubated at 10 °C emitted 32.56 ± 1.39 mg CO_2-C g^{-1} SOC, while samples incubated at 20 °C emitted 74.06 ± 2.98 mg CO_2-C g^{-1} SOC (Fig. 1). At 10 °C we did not observe a land-use effect on CO_2 emission (Fig. 1, Table 2), but at 20 °C the topsoil samples from croplands emitted less CO_2 than those from forests. This effect occurred due to extraordinarily high emissions of the samples from two grassland and two forest sites (VW_GL, VW_F, SK_GL, SK_F) (Fig. 3). Those four sites experienced the least intensive drainage. Furthermore, these sampling sites were situated at high altitude in a pre-alpine environment with lower mean annual temperatures and higher precipitation than at the other sites (Table 1). In pairwise comparisons between adjacent sites of different land use (i.e. VW_GL vs. VW_FL, SK_GL vs. SK_FL, C_CL vs. C_GL and G_CL vs. G_GL and G_FL; Table 1), land-use effects were only found for the last site (Fig. 3).

At 10 °C, CO_2 emissions of the topsoil samples from all sites together were higher than from samples taken at 30 to 60 cm depth, independent of land use (Table 3). When analysing the influence of depth separately by land-use type, this effect was found to only manifest in grassland and forest but not in cropland soils. We found no overall depth effect at 20 °C, but CO_2 emissions of topsoil samples from forests were higher than those of samples taken at lower depths, whereas we found the opposite case for the cropland soils. Despite these depth effects, the general relationship between emissions and soil depth was weak and not consistent in its sign (Table 4).

Over the course of the incubation, CO_2 emissions increased for 40 % of the samples, as revealed in Table S1 by positive slopes of the regression lines. These increases were independent of land use. In total, the CO_2 emissions from

Table 5. Incubation studies with organic and mineral soils at different moisture levels, soil depths and temperatures. If moisture level stays moist, samples were incubated directly after being retrieved from the field, while saturated samples were incubated under wetter, i.e. anaerobic conditions. Similar stands for samples whose water content was similar to our samples.

	Soil	Region	Moisture level	°C	Days	C / N	Depths (cm)	CO_2 emissions ($mg\,CO_2\text{-}C\,g^{-1}\,SOC\,d^{-1}$)	Q_{10}	Ea ($kJ\,mol^{-1}$)
This study	Drained Fens	Switzerland	−10 kPa	10	416	17.7	5–150	0.078	2.57	69.5
				20				0.18		
Chapman and Thurlow (1998)	Drained/undrained	UK (Scotland)	Moist	10	unclear		0–20	0.051*	3.2	80.0
	Bogs			20				0.030*		
Grover and Baldock (2012)	Bog	Australia	Moist	20	38	15–25	5–110	0.13–0.78		
Hahn-Schöfl et al. (2011)	Fen	Germany	Saturated	20	346	15.3		0.013		
Hardie et al. (2011)	Bog	UK	Drier	5	6	30	0-30	0.027	3.66	86.4
				10				0.049		
				15				0.093		
Hartley and Ineson (2008)	Mineral soil	UK	Drier	10	124		unclear	0.046	3.25	81.3
				20				0.074		
Hilasvuori et al. (2013)	Bog	Finland	Moist	10	short	83	0–44	0.016*	2	22.7
				20				0.061*		
Hogg et al. (1992)	Fen	Canada	Similar	8	120	40.6	5–40	0.083	1.9–2.2	62.0
				16				0.282		
				24				0.381		
Karhu et al. (2014)	Organic soil	UK (Scotland)	Similar	11.4	174	28.6	0–10	0.065		
	Organic soil	UK (Scotland)		7.6	174	36.5	0-10	0.105		
	Organic soil	UK		13.3	174	18.7	0–10	0.201		
	Mineral soil	UK		11.4	174	13.3	0–10	0.101		
	Mineral soil	Spain		21.5	174	14.3	0–10	0.293		
	Mineral soil	Spain		19	174	13.0	0–10	0.345		
	Mineral soil	Spain		19	174	18.6	0–10	0.448		
	Mineral soil	Italy		18.4	174	13.2	0–10	0.129		
Karhu et al. (2010)	Mineral soil	Finland		8–25	540	–	0–30	–	3.0	45.0
Koch et al. (2007)	Organic soil	Austria	Moist	0–30	25	21.6	0–5	–	2.0	31.9
Leifeld and Fuhrer (2005)	Mineral soil	Switzerland	Similar	25	707	7.85	5–35	0.12	4.6	110.8
Neff and Hooper (2002)	Organic soil	USA (Alaska)	Unclear	10	352	34.6	0–10	0.32	1.9	22.9
				30				0.75		
Plante et al. (2010)	Mineral soil	USA	Similar	15	56		0–20	0.28	1.36–1.79	31.7
Reiche et al. (2010)	Fen	Germany	Saturated	15	31	30.1	0–40	0.0022		
Reichstein et al. (2000)	Mineral soil	Switzerland	Similar	5	104	30.3		0.05	2.5–2.7	65.9
				15				0.14		
				25				0.22		
Scanlon and Moore (2000)	Fen	Canada	Moist	4	12	43	5–45	0.227	2.0	45.8
				14				0.109		
Wang et al. (2010)	Organic soil	China	Similar	5–20	40	28.5	10–30	0.31	2.2	53.3
Wickland and Neff (2008)	Organic soil	Canada	Similar	10	57	24.7	2–30	0.35	1.7	36.6
				20				0.79		
Yavitt et al. (2000)	Bog	Canada	Moist	12–22	2		0–54	–	1.4	32.5

* Study authors are not specific about the SOC content of peat; therefore we assumed it to be $400\,g\,kg^{-1}$, according to the findings of undisturbed bog peat (Loisel et al., 2014).

these samples were almost 50 % higher than those from the other samples that instead showed a trend of decreasing emissions.

Mean Q_{10} values were 2.57 ± 0.05. The Q_{10} did not differ between the three land use types. It was lower below 60 cm depth in the cropland and grassland but not in the forest soils (Fig. 1, Table 3). Activation energies (Ea) calculated from Q_{10} values ranged around 48.1 and 123.5 kJ mol^{-1} and like Q_{10} values decreased with depth. There were significant relationships between CO_2 emission and SOC content, bulk density and C / N ratio in general but they were weak (Table 4). The Q_{10} values showed similar relationships to these soil variables as CO_2 emission.

4 Discussion

4.1 SOM characteristics

The SOC contents, bulk densities and C / N ratios found in the deeper parts of soil profiles presented here were close to values that are typical for undisturbed peat (Grover and Baldock, 2012; Loisel et al., 2014). They also indicate that soils of our study sites were characteristic of European fens and resembled typical properties of managed organic soils (Berglund, 1995; Kechavarzi et al., 2010; Eickenscheidt et al., 2015; Krueger et al., 2015; Wüst-Galley et al., 2016; Brouns et al., 2016). Several studies assume that deeper layer peat of managed organic soils is less decomposed (Ewing and Vepraskas, 2006; Rogiers et al., 2008; Leifeld et al., 2011a, b; Krueger et al., 2015; Wüst-Galley et al., 2016). We therefore interpret the different SOM characteristics found in the topsoils of our samples as indicators of advanced decomposition triggered by drainage.

The land-use-specific differences manifested in different topsoil SOC contents and C / N ratios (highest under forest) and topsoil bulk densities (lowest under forest). The higher forest C / N ratios might be explained by absence of the use of N fertilizers and lower bulk densities by lower traffic with field machinery. In addition, differences in C / N between land-use types may also suggest that peat decomposition was less advanced in forests compared to croplands and grasslands. Further, depth effects are lowest in forest soils, indicating a lower impact of soil management that could also result in a lower decomposition of forest topsoils. The relatively high carbon stocks found in cropland top soils are most likely the result of subsidence after drainage and compaction from field traffic, leading to increased soil bulk density in the uppermost layers. This effect, with respect to C stocks, overrides the overall much smaller C concentration in agriculturally managed organic soils.

The H / C and O / C ratios in the deeper layers of the studied soils were similar to those found in undisturbed bogs and drained bogs used for forestry in Switzerland (Zaccone et al., 2007; Wüst-Galley et al., 2016). They indicate an enrichment of polyphenols and aromatic carbon with depth, which is in line with the current understanding of peat development in peatlands (Cocozza et al., 2003; Zaccone et al., 2007; Klavins et al., 2008; Delarue et al., 2011; Leifeld et al., 2012, 2017). The increased H / C and O / C ratios in the grass- and cropland topsoils can be attributed to inputs of fresh plant litter to the topsoil via above- and belowground residues, as such residues are rich in carbohydrates (Koegel-Knabner, 2002). In a previous study, in which we used stable and radiocarbon isotopes to label the SOC of two of the studied soils (C_CL and C_GL in Table 1), at least 20 % of topsoil organic matter was not peat but derived from recent plant litter (Bader et al., 2017). The results further indicated that the OM derived from these fresh plant residues was a source of labile C that contributed more to decomposition than the old, peat-derived SOM.

The H / C and O / C ratios reflect the mixing ratio of these two SOM sources. The H / C and O / C ratios in forest topsoils were lower than of those under cropland and grassland and did not change with depth. Interpreting these lower H / C and O / C ratios in the forest topsoils as indicators of more advanced peat decomposition (Klavins et al., 2008; Leifeld et al., 2012; Biester et al., 2014; Wüst-Galley et al., 2016) would be in contradiction to our conjecture that land management effects on peat decomposition, revealed by SOC, bulk density and C / N ratio, are less pronounced for forests. We rather argue that the reason for the low H / C and O / C ratio in the forest soils is a higher abundance of lignin rich (wood derived) plant residues. A second mechanism for comparably higher O / C and H / C ratios in cropland and grassland soils could be that peat loss in the uppermost layers was higher under agriculture than under forest, resulting in a relatively higher share of SOM from recent inputs. Considering all the available evidence of SOM characteristics, we conclude that peat decomposition is less advanced in forest soils than in agricultural soils and also in line with field flux measurements on managed organic soils that typically show faster decomposition in croplands and grasslands than in forests (IPCC, 2014).

4.2 CO_2 emissions and temperature sensitivity of decomposition

The studied soils emitted, on average, ca. 5–10 % of their SOC (20 °C) as CO_2, calculated for an incubation duration of 1 year. The advanced decomposition state of many of the samples might give reason to expect that CO_2 rates are below that of more intact peat or mineral topsoils that contain a larger fraction of recent plant residues. To understand whether SOM in the studied organic soils is particularly stable, we compared its average daily carbon loss with data from studies that used undisturbed to extensively managed organic soils or mineral soils (Table 5, Fig. 4). Indeed, our values are on average below those from other organic soil studies. However, their range overlaps with the uncertainty

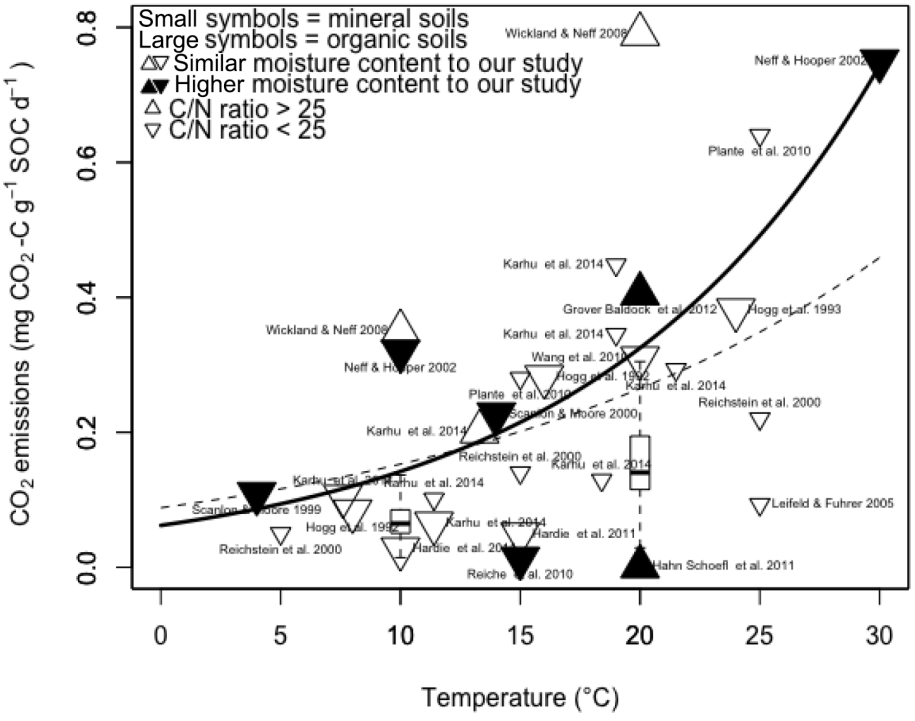

Figure 4. Comparison of daily CO_2 emission rates from this study (box plots) with rates found during other incubation studies (organic soils and mineral soils). The curves represent the modelled CO_2 emission rates for organic soils from other studies (solid thick line) rate $= 0.06 \times 10^{0.08t}$ and mineral soils (dashed thin line) rate $= 0.09 \times 10^{0.05t}$ for temperatures between 0 and 30 °C.

of the regression line that is plotted through results from studies from other, mostly intact or little degraded organic soils. Hence, the pronounced oxidative decomposition after long periods of drainage might result in a relatively smaller labile SOC pool, but the large variability between experimental set-ups, incubation lengths and water contents among incubation studies prevents a stronger line of interpretation. Interestingly, the regression lines modelled for organic and mineral soils did not deviate significantly from each other. Therefore, the pools size of labile carbon, indicated by the decomposition rates, seem not to differ between these soil classes. This comparison suggests that accumulation of recent, labile plant materials that presumably account for most of the evolved CO_2 is not systematically different between mineral and organic soils.

Samples showing an increase in CO_2 emission rate over time were predominantly of subsoil origin, where SOC contents and C / N ratios indicate a lower decomposition than in the topsoil. Furthermore, based on the information we have on land use and drainage depths, it appears that most of these samples were taken from soil layers that were protected from intensive decomposition by water saturation. The long incubation period in our study might have given aerobic decomposer communities time to develop and grow, whereas time might not have been sufficient in shorter studies.

Like other studies on extensively managed or undisturbed organic soils that investigated depth interaction of decomposition rates in the top 30 to 50 cm (Hogg et al., 1992; Scanlon and Moore, 2000; Wang et al., 2010; Hardie et al., 2011), we found a decrease of specific CO_2 release with depth. However, the relationship between CO_2 emissions and depth was rather weak in our case and not consistent for both incubation temperatures and the different land uses. Compared to the studies on unmanaged organic soils, reporting declines of a factor 2 to 30 (Hogg et al., 1992; Scanlon and Moore, 2000; Wang et al., 2010; Hardie et al., 2011), our differences were substantially smaller. Drainage and decadal agricultural use of the studied soils led to more intense decomposition processes in the topsoil, resulting in little depth interaction or, for croplands, sometimes maybe even a reversal of decomposability. Further, the abundance and decomposability of crop residues have to be considered as a substantial CO_2 source. For two topsoils (C_GL and C_CL), Bader et al. (2017) showed that at least 20 % of the SOM is crop residue derived and responsible for 40 % of the emitted CO_2. Assuming that the abundance of crop and plant residues is highest in topsoils, it might be possible that decomposability of peat-derived SOM either does not depend on depth or topsoil peat decomposes at smaller rates. Therefore, we cannot confirm our first hypothesis of decreasing decomposition rates with depth.

As Table 5 shows, the Q_{10} values found in our study (2.74 ± 0.06) were higher than Q_{10} values found elsewhere for similar sampling depths but in unmanaged organic soils

(Chapman and Thurlow, 1998; Hamdi et al., 2013; Hardie et al., 2011; Hilasvuori et al., 2013; Hogg et al., 1992; Scanlon and Moore, 2000; Yavitt et al., 2000). Also the temperature-independent Ea was higher in the studied samples ($69.4 \pm 3\,\mathrm{kJ\,mol^{-1}}$) than in most other studies on undisturbed organic soils ($47.4 \pm 7.2\,\mathrm{kJ\,mol^{-1}}$) (Table 5). However, three studies (Chapman, 1971; Hardie et al., 2011; Hogg et al., 1992) found similar or higher Ea values in northern organic soils. In the case of Chapman and Thurlow (1998) they were also managed as grassland or forest, whereas the other studies used peat from undisturbed organic soils. Nevertheless, the high Ea of the studied samples might reflect the change in chemical peat composition with decomposition after drainage towards higher recalcitrance. In contrast to other studies on unmanaged organic soils reporting no trend or increasing Q_{10} values with depth (Scanlon and Moore, 2000; Wang et al., 2010; Hardie et al., 2011; Hilasvuori et al., 2013), the cropland and grassland profiles in our study had a lower Q_{10} below the 60 cm depth. Various studies on SOM decomposition used Q_{10} values as an indicator of SOM recalcitrance (Hogg et al., 1992; Biasi et al., 2005; Davidson and Janssens, 2006; Conant et al., 2008, 2011; Hartley and Ineson, 2008; Hilasvuori et al., 2013). Considering that the presence of labile crop residues would decrease Q_{10} in the topsoil rather than in the subsoil, the higher topsoil Q_{10} may be explained by an extended accumulation of recalcitrant moieties. This proposed a high abundance of recalcitrant moieties in topsoils of degrading organic soils is in line with the reported differences in SOM composition in different layers as well with the pattern of CO_2 emissions. The latter show no substantial difference with depth and indicate that a higher fraction of recent and labile plant residues in topsoils is counterbalanced by a high recalcitrance of the highly degraded peat. Comparing radiocarbon concentrations in SOC and emitted CO_2 of two sites also used for this study (C_CL, C_GL), Bader et al. (2017) estimated that SOC from plant residue inputs is more labile than peat. The measured radiocarbon contents for SOC were 75 to 80 pMC and therefore indicated that peat of the topsoil must have experienced a substantial decomposition.

It is remarkable that despite the controlled conditions in our incubation experiment the variation in cumulative loss of initial SOC of between 0.6 and 42.3 % (Fig. 4) was similar to or even larger than that observed in field flux measurements (IPCC, 2014). This large variability suggests that the composition of SOM is of similar importance to drainage, climate and other site factors in controlling CO_2 emissions from drained organic soils. Nevertheless, the relationships between the measured SOM parameters used to assess the biochemical decomposability, CO_2 emissions and Q_{10} values were rather weak and thus do not support our second hypothesis. This stands in contrast to other studies which concluded that chemical composition is a major factor of SOM decomposability in organic soils (Scanlon and Moore, 2000; Koch et al., 2007; Reiche et al., 2010; Hardie et al., 2011; Leifeld et al., 2012). However, these studies focused mainly on single profiles of undisturbed or extensively used organic soils. A recent study investigated relationships between SOM parameters and decomposition rates of German organic soils under controlled conditions (Säurich et al., 2017). These authors mostly studied strongly disturbed fens with similar properties to the soils in our study. Besides SOC contents, soil pH and C / N ratios, Säurich et al. (2017) focused on other soil nutrients, stable isotopes and microbial biomass. In line with our results, they could not identify strong proxies for SOC decomposition by means of simple chemical attributes.

In order to explain the weak relationships between SOM composition and CO_2 release it should be considered that, in our case, the emitted CO_2 comprised, on average, only 3.2–7.4 % of the total SOC, while the analysed SOM parameters in this and other studies represent bulk SOM. Our methods allowed us to gain a broad overview of the chemical composition of SOM, while decomposition might more tightly be bound to the abundance of specific OM moieties.

Although land-use-affected SOM characteristics, such as elemental contents and their ratios, the amount of CO_2 emitted from the soils did not differ among the three types of land use. We therefore have to also reject our third hypothesis of a higher SOM decomposition rate in forest topsoils. We assume that long-lasting drainage and management might have resulted in an equivalent decomposition of most of the labile OM, along with its intrinsic decomposability.

5 Conclusions

Chemical characteristics of SOM indicated advanced peat decomposition in the uppermost layers of drained organic soils used as cropland or grasslands. Under controlled moisture and temperature conditions, CO_2 emissions from peat samples had a similar variability, as found for in situ CO_2 flux experiments on drained organic soils. Therefore, carbon loss from drained organic soils cannot be explained entirely by climate or drainage depth. However, simple chemical characteristics of SOM, as used in this study, were not specific enough to explain the variability in CO_2 emissions or the temperature sensitivity of decomposition under controlled conditions. Despite CO_2 emissions being occasionally higher in topsoils, probably derived from accrual of labile plant residues, the remarkable decrease of Q_{10} values with depth suggested that the relative content of recalcitrant peat-derived SOM was high in topsoils of managed organic soils, indicating advanced degradation in these uppermost layers. It is therefore necessary to quantify the fraction of peat-derived SOM throughout a drained soil profile in order to verify this assumption. Yet we understand from the similar magnitude of CO_2 emission rates found above and below 30 cm depth that future peat loss will occur at similar or even faster rates, assuming an increasing mean annual temperature.

Data availability. The underlying research data are provided in the supplement.

The Supplement related to this article is available online at https://doi.org/10.5194/bg-15-703-2018-supplement.

Competing interests. The authors declare that they have no conflict of interest.

Acknowledgements. This study was funded by the Swiss National Science Foundation in the framework of the National Research Programme "Sustainable Use of Soil as a Resource", grant number 406840_143145. We are grateful to Chloé Wüst-Galley for help with site selection and her statistical support, and to Robin Giger for analysing the soil samples.

Edited by: Jens-Arne Subke
Reviewed by: three anonymous referees

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