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Long-term soil organic carbon dynamics in temperate cropland-grassland systems

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ABSTRACT

Increasing soil organic carbon (SOC) in agroecosystems enables to address simultaneously multiple goals such as climate change adaptation and mitigation as well as food security. As croplands are depleted in SOC, they offer a great potential to sequester atmospheric carbon (C). Nonetheless, croplands are still losing SOC under most of the current agricultural systems. Although many factors driving SOC dynamics have already been identified, their relative importance has not been quantified yet. Using one of the densest European soil monitoring networks with 250 sites established in western Switzerland, in the present study we (i) assessed long-term (over 30 years) SOC dynamics in croplands (CR), permanent grasslands (PG) and mountain pastures (MP), and (ii) prioritized the importance of land use, soil characteristics and sites conditions in driving SOC dynamics. The SOC levels in PG and MP were similar when clay content was accounted for, whereas CR were depleted in SOC by 3.9 mg C mg^{-1} clay as compared to PG. The majority (61 %) of CR had SOC:clay ratio below 1:10, but only 16 % of PG and MP sites reached this threshold. By contrast, soil organic matter stoichiometry (C:N:Porg ratios) was similar in CR and PG for comparable SOC content. The increase of C:Porg ratio with SOC content (dilution effect) and the high total P in CR and PG (legacy effect) indicate the possibility to sequester atmospheric C at reduced nutrient sequestration costs. SOC changes ranged from -0.61 to 1.32 mg g^{-1} soil yr^{-1} and were the highest in sites that experienced land-use changes. No PG were losing SOC, while CR sites exhibited both SOC gains and losses. Because of the predominance of the initial SOC content on SOC dynamics, land-use history must be accounted for when assessing the effect of management practices. The main manageable factors driving SOC dynamics were the time under temporary or permanent grasslands along with the soil total P. As PG already are rich in SOC and total P, organic amendments should be partly redirected to CR.

1. Introduction

The increase of food production in the mid-20th century has caused serious ecosystem degradations and large greenhouse gas emissions (Bennetzen et al., 2016; Gibbs and Salmon, 2015). In particular, agricultural soils are currently strongly depleted in soil organic carbon (SOC) due to a long lasting imbalance between soil carbon (C) inputs (high biomass removal from the system and low organic amendments), soil C outputs (enhanced microbial mineralization and soil erosion) (Sanderman et al., 2017). SOC plays a central role in sustaining agricultural production by regulating many biological, chemical and physical soil functions in addition to sequestrating atmospheric C. Thus, increasing SOC in agroecosystems would address simultaneously multiple sustainable development goals, such as climate change adaptation and mitigation as well as food security (Rumpel et al., 2019).

Results from different soil monitoring networks in Europe have shown that the balance of SOC stock in agricultural soils is not positive (Gubler et al., 2019). Long-term experiments in Switzerland have also demonstrated that measures to mitigate environmental impacts of agricultural systems, which farmers implemented to receive subsidies (Proof of Ecological Performance), have not been sufficient to preserve or increase SOC, especially in croplands (Emmel et al., 2018; Keel et al., 2019; Maltas et al., 2018, 2013). Predictions of SOC evolution in temperate agricultural soils are not optimistic because SOC gained by increasing soil C inputs would be partly or totally offset by negative effects of climate change on soil C (Wiesmeier et al., 2016). Studying

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SOC dynamics can be methodologically challenging because small stock changes must be detected compared to the large size and high variability of SOC pools (Paustian et al., 2019; Smith et al., 2020). Accordingly, it is not unusual that the majority of soil monitoring networks do not show any significant change even after decades of monitoring (Capriel, 2013; Gubler et al., 2019). Despite the uncertainties in assessing SOC dynamics in the field, some sites have exhibited SOC gains, indicating that under certain conditions an increase of SOC stock is possible (Capriel, 2013; Gubler et al., 2019).

The impact of agriculture on SOC depends on land-use type. For example, croplands are usually more depleted in SOC than grasslands (Tóth et al., 2013). Grasslands provide more plant-derived C inputs, especially belowground, due to higher rate of root turnover and continuous soil cover (Poeplau and Don, 2015; Soussana et al., 2010). The continuous soil cover and the absence of disturbance from tillage limit soil C outputs due to reduced soil erosion and increased physical protection (Van Oost et al., 2007). Recent studies, however, suggest that reduced tillage does not necessarily increase SOC stocks, but likely leads to a redistribution of SOC closer to the soil surface (Haddaway et al., 2017; Powlson et al., 2014). Increasing soil cover by introducing cover crops within crop rotation can limit SOC losses in croplands (Emmel et al., 2018; Poeplau and Don, 2015). For example, higher SOC levels in croplands have been observed where higher proportions of temporary grasslands are part of a crop rotation (Stumpf et al., 2018). The higher SOC observed under permanent grasslands further supports these findings (Gubler et al., 2019; Stumpf et al., 2018). SOC stock in permanent grasslands responds favorably to several practices, such as organic amendment, mineral fertilization or improved species mix (Conant et al., 2017; Poeplau et al., 2018).

Nitrogen (N) and phosphorous (P) applications have contrasting effects on SOC stocks as it may favor biomass production and rhizodeposition, but also SOC decomposition (Rüegg et al., 2019). For example, fresh organic carbon inputs may lead to priming of the old SOC (Kuzyakov, 2010). However, nutrient applications can reduce nutrient-mining in old SOC (Chen et al., 2014) and increase microbial C use efficiency (Spohn et al., 2016). On the other hand, nutrient applications can enhance C outputs due to microbial mineralization of SOC and litter (Poeplau et al., 2016). The balance between these opposite processes is largely controlled by stoichiometry constraints resulting from imbalances in elemental ratio between soil microbes biomass, soil organic matter (SOM) and soil amendments (Bertrand et al., 2019; Spohn, 2020a). Hence, soil inputs with optimum stoichiometry will promote SOC stocks by limiting negative priming of SOC and by increasing the C retention of organic matter inputs (Finn et al., 2016; Kirkby et al., 2013; Qiao et al., 2016). The constrained range of SOM stoichiometry implies that, inevitably, C sequestration in soil will be accompanied by nutrient sequestration, hence nutrient availability to plants (Spohn, 2020b; Van Groenigen et al., 2017). Because SOM stoichiometry differs between land-use types, the nutrient requirement to sequester C varies between croplands and grasslands (Coonan et al., 2020).

The conversion of croplands to grasslands leads to a long lasting accumulation of SOC (Poeplau et al., 2011). The effect is reversible when grasslands are converted to croplands, with SOC losses occurring at faster rates than SOC gains (Soussana et al., 2004). SOC accumulation rates are negatively related to SOC content (Capriel, 2013; Gubler et al., 2019; Keel et al., 2019) indicating a diminishing return of practices designed to increase SOC at higher SOC content. Hence, it is fundamental to account for site history when assessing the effect of management practices on stock changes, because the effects of land-use changes may last for decades (Smith, 2014). Most studies on land-use changes consider a single conversion, probably because the effect of land-use change is frequently studied by a space-for-time substitution approach that compares paired plots (Don et al., 2011; Guo and Gifford, 2002; Poeplau et al., 2011). Land-use trajectories, however, are more complex and have fast changing dynamics, especially in a long-term soil monitoring network where several changes of land use may have occurred since the establishment of the network (Zhao et al., 2009). Farmers continuously adapt their crop rotations by including new crop types, changing their relative proportion, or introducing temporary grasslands before returning to crop cultivation. As a result, a field is subject to a continuous change of different trajectories within a cropland-grassland system (Stumpf et al., 2018). To account for these distinct trajectories and fast changing management practices, several studies have adapted the methodology of “sequence analysis” to characterize land-use changes by including several indices based on sequence characteristics (Mas et al., 2019; Watson et al., 2014).

Sustainable SOC management has recently gained awareness at political level also thanks to the “4 per 1000″ initiative launched at COP21 in 2015 (Soussana et al., 2019). However, linking governmental subsidies to SOC sequestration remains challenging due to the lack of accurate quantification methodologies. Empirical models such as humus-balancing methods are currently evaluated by governments as an alternative, for instance in Switzerland (FOAG, 2020). The reasons for such empirical models are that the basic factors maintaining or favoring SOC stocks are relatively well-known, such as land-use types and management practices (Basile-doelsch et al., 2020; Chenu et al., 2019; Smith et al., 2016; Paustian et al., 2019; Wiesmeier et al., 2019). However, the hierarchical and relative importance of these factors remain largely unknown and not fully incorporated in the humus-balancing methodology (Brock et al., 2013). We took advantage of one of the densest European soil-monitoring networks, the soil monitoring network of Fribourg canton in Switzerland (FRIBO), to assess *in situ* the impacts of agricultural systems on SOC dynamics and to identify the most promising approaches for increasing SOC stocks in agricultural soils. The FRIBO network includes the main agricultural systems and pedo-climatic conditions of Switzerland, and thus it is ideal for extrapolating the findings to other regions with similar climatic and agricultural systems. The specific aims of this study are to (i) quantify current SOM quantity and stoichiometry in agricultural soils under different land uses in the southwest region of Switzerland, (ii) determine SOC dynamics during the last 30 years, and (iii) quantify the relative importance of land use and soil characteristics on SOC dynamics.

2. Materials and methods

2.1. Study sites

A soil monitoring network named FRIBO was established in 1987 by the Agricultural Institute of the Fribourg Canton (Switzerland) with the aim of monitoring agricultural soil quality (Levasseur et al., 2019; Blanchet et al., 2017; Roger et al., 2014; Frau et al., 2020). The network is composed of 250 sites within the Fribourg Canton (167,000 ha) which is located north of the Alps, in the western part of Switzerland (46° - 47 °N, 7 °E). Lying on the Swiss Midland between the piedmont of the Jura Mountains (NW) and the foothills of the Western Alps (SE), the Canton presents a NW-SE gradient of soils, elevation and climate. The Fribourg Midland is characterized by a gentle slope from NW to SE, which gradually increases from 429 m a.s.l. to 800−900 m a.s.l., towards a mountainous landscape on the Alps foothills and then becomes much steeper (< 2389 m a.s.l.). Its geology is composed of Tertiary molasse (sandstone, marl) partly covered by moraine deposits with a smooth hilly topography. The lowest part of the Midland in the NW is flatter and partly covered by a lake and alluvial sediments. Soils are dominated by Cambisols, Gleysols and Fluvisols. The Alps foothills have a limestone origin and are dominated by Gleysols, Leptosols and Regosols (Frau et al., 2020). The climate is temperate continental (MAT: 8.9 °C; MAP: 1075 mm) in the Midland. In the foothills, for every 100 m change in elevation, precipitation increases by about 80 mm and temperature decreases by about 0.5 °C (Dumas, 2013; Sevruk, 1997).

Agricultural land (AL) covered 75,516 ha in 2017, representing 7.2 % of the total agricultural area of Switzerland (DIAF, 2019). During summer, an additional surface of 20,000 ha is used as mountain pastures (MP). Animal production, especially milk production, is important and represents 9.3 % of the Swiss agriculture production. Permanent (PG) and temporary (TG) grasslands occupy 67 % of the agricultural land. TG are artificial grasslands sown as part of crop rotations to produce forage. However, according to Swiss legislation (OTerm 910.91), TG are considered as PG if left for at least 6 years. TG are usually more productive than PG because of the selected species mixture. Annual crops covered 30 % of the AL in 2017, half of which was composed of cereals (CER). About one quarter of the annual crops are managed under reduced or no-till practices.

Alpine pastures are usually not fertilized. Cropland and grassland fertilization must follow the official Swiss guidline for fertilization (Sinaj and Richner, 2017) and farmer must establish a nutrient balance plan that account for nutrient input and outputs for each land-use type (Sinaj and Richner, 2017; Herzog et al., 2008). In the Fribourg canton, the main source of fertilization for croplands, temporary and permanent grasslands is cattle manure and cattle slurry (Spiess, 2011). Following the guidelines, extensive grasslands are not fertilized, but intensive grasslands (i.e. productivity of about 13 Mg of dry matter ha^{-1}) are fertilized with up to 170 kg N ha^{-1} yr^{-1}, 47 kg P ha^{-1} yr^{-1}, 236 kg K ha^{-1} yr^{-1} and 33 kg of Mg ha^{-1} yr^{-1} (Sinaj and Richner, 2017). Additional sulphur (S) fertilization is usually not necessary as organic amendments contain sufficient amounts of S. Croplands and to a lesser degree grasslands are limed 1–2 times per decade if soil acidification occurs (Sinaj and Richner, 2017).

The FRIBO network is based on a 2 × 2 km grid, similar to the LUCAS Topsoil Survey at European level (Tóth et al., 2013). However, the FRIBO network is more than one order of magnitude denser than the LUCAS Topsoil Survey. The site selection was based on current land use, land-use history, topography and representativeness of the surrounding conditions (Julien and Morand, 1995). As the main goal of the network was to monitor soil degradation, the selection of sites was intentionally biased towards croplands. Out of the 250 sites, 6 sites classified as Histosols were excluded from this study because of their limited extent and because the factors affecting SOC dynamics in organic and mineral soils are different. Gleysols were removed from our analysis and included only in the presentation of land-use characteristics (Table 1) because of the impact of waterlogging on SOC and nutrient dynamics (Maranguit et al., 2017) and their low occurrence (28 sites). Similarly, Leptosols (3 sites), vineyards (3 sites) and 1 site recently converted from forest into cropland were also excluded from this study, but they wereonly included in the presentation of land-use characteristics (Table 1). The remaining dataset included 107 croplands, 77 permanent grasslands and 25 mountain pastures for a total of 209 sites according to the last sampling campaign.

Table 1
Main soil characteristics (0−20 cm) in relation to three current land-use types for the last sampling campaign. All soil types are included, with the exception of Histosols. Letters indicate significant differences between land-use types ($p < 0.05$) as determined by Kruskal-Wallis test. Data are mean ± standard deviation (min-max), n is the number of sites for each land-use type.

	Croplands n = 117 (114)[a]		Permanent grasslands n = 83 (80)		Mountain pastures n = 44 (40)	
SOC (mg g^{-1})	17 ± 7 (7–50)	a	31 ± 10 (13–59)	b	45 ± 11 (25–68)	c
Ntot (mg g^{-1})	2.1 ± 0.9 (1–5)	a	4.0 ± 1.2 (2–7)	b	4.8 ± 1.5 (2–10)	b
Ptot (mg g^{-1})	0.87 ± 0.23 (0.4–1.6)	a	1.11 ± 0.29 (0.6–2.3)	b	0.93 ± 0.34 (0.3–1.7)	a
Porg (mg g^{-1})	0.40 ± 0.16 (0.2–1.0)	a	0.68 ± 0.21 (0.4–1.3)	b	0.66 ± 0.29 (0.2–1.4)	b
C:N	9.6 ± 1.4 (6–21)	b	9.2 ± 0.7 (6–11)	a	10.6 ± 1.7 (8–17)	c
C:Ptot	22 ± 6 (11–38)	a	33 ± 8 (16–54)	b	59 ± 23 (27–120)	c
C:Porg	50 ± 13 (30–109)	a	54 ± 12 (29–101)	b	83 ± 31 (39–184)	c
N:Ptot	2.3 ± 0.7 (0.9–4.0)	a	3.6 ± 0.8 (1.7–5.9)	b	5.6 ± 1.7 (2.8–9.8)	c
N:Porg	5.3 ± 1.3 (3.2–10.0)	a	5.9 ± 1.3 (3.2–10.5)	b	7.9 ± 2.3 (4.3–13.9)	c
C:N:Porg[b]	56 : 5 : 1		54 : 6 : 1		86 : 8 : 1	
pH_{H20}	6.6 ± 0.7 (5.2–8.1)	c	6.2 ± 0.6 (5.1–7.6)	b	5.9 ± 0.7 (4.6–7.4)	a
CEC (cmol kg^{-1})	13 ± 5 (6–38)	a	19 ± 8 (9–52)	b	24 ± 10 (12–50)	c
Sat_{CEC} (%)	57 ± 18 (24–100)	a	60 ± 12 (32–92)	a	57 ± 16 (18–87)	a
Clay (mg g^{-1})	178 ± 7 (87–581)	a	238 ± 11 (113–752)	b	343 ± 11 (190–654)	c
SOC:clay (%)	9.8 ± 3.3 (5.4–34.8)	a	13.7 ± 3.6 (4.9–23.1)	b	14.1 ± 5.3 (7.0–35.7)	b
Elevation (m.a.s.l)	606 ± 114 (430–865)	a	771 ± 125 (460–1015)	b	1280 ± 180 (880–1590)	c

SOC: soil organic carbon; Ntot: total nitrogen; Ptot: total phosphorus; Porg: organic phosphorus; CEC: cation exchange capacity; Sat_{CEC}: saturated cation exchange capacity.

[a] number of samples for Ptot and Porg analysis.

[b] C:Porg ratios were first calculated for each samples by multiplying C:N by N:Porg and then averaged over all samples.

2.2. Soil sampling and analysis

Every year since 1987, 50 sites have been monitored on 10 × 10 m wide plots. Thus, a sampling cycle lasts 5 years before the same site is resampled. The sixth and last completed sampling cycle started in 2012 and finished in 2016. Soils were sampled between 0−20 cm using an Edelman soil auger. A composite soil sample for each site was obtained by mixing 25 soil cores collected systematically on a 2 × 2 m grid within the 10 × 10 m plot. Plant residues were removed and soil samples were air-dried and sieved at 2 mm before further analysis. SOC (Walkley-Black method), total nitrogen (Ntot, Kjeldahl method), pH-H_2O (1:2.5 w:v), particle-size analysis (pipette method) and cation-exchange capacity ($BaCl_2$ extraction) were measured following the Swiss standard methods (Agroscope, 2011). Total phosphorous (Ptot) was measured using the molybdate colorimetric method (Murphy and Riley, 1962) on an extraction of 0.25 g of soil with 5 ml of hydrofluoric acid (40 %) and 1.5 ml of $HClO_4$ (65 %) (NFX 31–147). Organic phosphorous (Porg) was measured following Saunders and Williams (1955). The ratios of organic carbon (hereafter C) to total N (C:N), total P (C:Ptot), and organic P (C:Porg) as well as total N to total P (N:Ptot) and organic P (N:Porg) are expressed as mass ratios. Soil organic matter (SOM) stoichiometry refers to C:N:Porg ratios and soil stoichiometry to C:N:Ptot ratios.

2.3. Land-use types characterization

For the purpose of this study, two different land-use classifications have been subsequently used. First, sites were classified in three common land-use types, i.e. croplands (CR), permanent grasslands (PG), and mountain pastures (MP), according to the land-use present at the sampling time of the 6^{th} cycle to enable comparison with literature. Within the CR, crops were divided into three subgroups following Gubler et al. (2019) in order to investigate the effect of crop type on SOC dynamics. The three subgroups were cereals (CER: wheat, barley, triticale, rye and oat), hoe crops (HOE: maize, beets, potatoes and rapeseed) and others (OTH: e.g. vegetables, tobacco).

A second classification was adopted to account for land-use changes between PG and CR (no land-use change in MP) and the diversity of crop rotations (within CR) on SOC dynamics. The main crop cultivated each year was recorded *a posteriori* every 5 years from farmer declaration. The proportion of land-use type varied over the 30 years of monitoring (Fig. S1). The PG sites that remained PG over the 30 years of monitoring are called PGr category to avoid confusion with the first classification based on the current land-use types. The remaining sites cropped for at least 1 year (n = 121) were divided into four categories by cluster analysis according to their land-use trajectories (see Statistical analysis section). The first two categories corresponded to sites that always remained under cropland (CRr), and subsequently were renamed intensive (CR_{int}) and extensive (CR_{ext}) croplands. They differed mainly by the proportion and the number of successive years under temporary grasslands (TG) within the crop rotation (Table S1). The last two categories included sites that experienced land-use change between CR and PG (LUC) and were subsequently named intensive (LUC_{int}) or extensive land-use change (LUC_{ext}) depending on the land-use change regime (Table S1).

Sites have been assigned to the four categories by cluster analysis based on their land-use trajectories. Land-use trajectory analysis was done with the package *TraMineR* following the methodology of Mas et al. (2019). Analysis of sites hierarchical clustering was performed using Ward's method. The approach uses the function *hclust* on pairwise dissimilarities between land-use sequences. The pairwise dissimilarities were calculated based on the longest common subsequence (LCS) between two sequences *x* and *y*. The distance (δ) was calculated following Eq. 1:

$$\delta\,(x,y) = A(x,x) + A(y,y) - 2A(x,y) \tag{1}$$

where *A(x,y)* is the length of the longest sequence common to *x* and *y*, and *A(x,x)* and *A(y,y)* are the length of *x* and *y*, respectively.

To represent the complexity of land-use sequences, the entropy and turbulence indices were calculated using the land-use sequence at each site (Mas et al., 2019). The entropy for a site is higher for a higher number and equal proportions of land use types. The entropy (E) corresponds to the Shannon index and was calculated following Eq. 2:

$$E = \; -\sum_{i=1}^{l} p_i \log(p_i) \tag{2}$$

where *pi* is the proportion of the land-use category *i* and *l* is the total number of land-use categories (i.e. 5; CER, HOE, OTH, TG and PG). Additionally, *E* was normalized to the theoretical maximum.

The turbulence accounts for the regularity of land use, land-use rotation or land-use conversion within a sequence. It is based on the number and the length of distinct subsequences *s* within the total sequence *S* of land use for a site. All successive land uses s_i of *s* subsequences must appear in *S* in the same order. The turbulence (T) was calculated following Eq. 3:

$$T = log_2\left(\varphi(x)\frac{s^2_{t,max}(x) + 1}{s^2_t(x) + 1}\right) \tag{3}$$

where $\varphi(x)$ is the number of different subsequences, $s^2_t(x)$ and $s^2_{t,max}(x)$ the variance of the consecutive times spent in the distinct state and its maximum value given the length of the sequence *S*, respectively. Finally, to determine if sites had experienced an increase or a decrease in time spent under grassland over the 30 years ($Grass_{ratio}$), we developed an index ($Grass_{dyn}$) corresponding to the $Grass_{ratio}$ after the 3rd cycle minus $Grass_{ratio}$ before the 3rd cycle. The 3rd cycle was selected as the midpoint.

Since the 6th sampling cycle started in 2012, this year corresponded to the last land-use data recorded for the 50 sites sampled in 2012. Because there was no land-use data prior to 1987, there was only one land use data for the first cycle of the 50 sites sampled in 1987 and 5 for the sites sampled in 1991. Consequently, the proportion of land use each year across the 209 sites was calculated only until 2012 to avoid bias in proportion due to the sampling strategy. For the same reasons, the proportion of land use and crop type in each site ($Grass_{ratio}$, CER, HOE, OTH) was calculated using data from the 2nd to the 6th cycle.

Sequence analysis was not possible for 11 sites because of an important number of missing annual land use data (> 20 %). For the remaining sites, only 84 annual land-use data out of the 7004-recorded data (1.2 % of the records) were missing, distributed across 22 sites. Gaps were filled using the rotation scheme followed by the farmer if the rotation did not change. However, for changing rotations, missing data were filled equally with the land uses preceding and following the gap. Since it was not possible to determine the crop identity, gaps were filled either as temporary grassland or as *other* crops.

2.4. Statistical analysis

All statistical analyses were performed using R software 3.6.1 (R Core Team, 2019). Effects of land use categories on soil parameters were tested by *Kruskal-Wallis* non-parametric test because data were not normally distributed even after log-transformation. Significant differences between land uses were assessed by Dunn tests (function *dunnTest* in *FSA* package), which account for ties. Effects on soil response variables of land-use types with co-variables such as SOC and clay contents were assessed by ANCOVA models (function *lm*). If the interaction between factors was significant, differences in slopes between land-use were assessed by the function *emtrends* (*emmeans* package). If not significant, the interaction term was removed from the model and differences between the three lands uses were assessed by the function *emmeans* (*emmeans* package). If data were not normally distributed, even after log-transformation, a Monte Carlo simulation with 10,000 replicates was performed (function *PermTest* in *pgirmess* package) on the ANCOVA model. Variance partitioning between factors and interaction was assessed by omega squared (ω^2) computed using *anova_stat* function (*sjstats* package). SOC accumulation rate was calculated for each site by linear regression between the SOC content and the sampling year. As SOC dynamics may not be linear, sites with significant SOC changes were determined as such if p-values were significant either in a linear regression model or in a spearman correlation test between SOC and sampling cycle. The hierarchy of factors affecting SOC change rates was determined by stepwise multiple regressions (function *stepAIC*) on standardized data and the corresponding semi-partial coefficient of determination. The model was run again with non-transformed data to determine the absolute effect of each factor on SOC rates. Minimal detectable effect size at a power of 0.8 was computed with the software G*Power 3.1.9.3 using linear bivariate regression with two tails (Faul et al., 2009). To estimate the minimum detectable SOC change in subsequent future cycles (7–13), data points were modeled assuming that the standard deviation of the residuals of the linear model fitted for the 6 first cycles remained constant for the subsequent future cycles. Random SOC data were computed for each new cycle using the function *rnorm*. The mean value was the SOC value predicted by the model (intercept and slope) for the year of the respective cycle. The standard deviation of the model residuals was calculated as the square roots of the mean square of the residuals given by the model. Linear regressions were run with the modeled data points to determine if SOC changes would be significant. Finally, the standard deviation of SOC including the modeled data points was used in G*power to determine the new minimum detectable SOC change after each subsequent cycle. If not specified, data are presented as mean ± standard error (SE) and discussed differences are significant at least at P-value < 0.05.

3. Results

3.1. Current soil physico-chemical characteristics

SOC content was the lowest in cropland (CR, 17.1 ± 0.7 mg C g^{-1} soil), almost two times higher in permanent grasslands (PG), and about 2.5 times higher in mountain pastures (MP) (Table 1). MP showed the highest C:N, C:Ptot and C:Porg ratios, the highest CEC and the lowest pH. Base saturation was similar in all land-use types. Nonetheless, site conditions, such as clay content or elevation varied between land-use types (Table 1). Compared to PG and CR, MP sites, which were located at higher elevations, contained more clay and had a higher proportion of soils experiencing waterlogging (16 Gleysols in MP *vs.* 6 in CR and 6 in PG). Though elevation and clay ranges overlapped for CR and PG, their distribution for PG was slightly skewed toward higher values.

Clay content strongly influenced SOC in the three land-use types (Fig. 1a). The model explained a large proportion of the data variability (R^2_{adj}: 0.79, n = 209). More SOC variation was explained by clay (ω^2: 0.36) than by land-use type (ω^2: 0.26). The interaction between factors was significant, but the effect size was small (ω^2: 0.01). The relationship between clay and SOC was not linear (Fig. 1b), exhibiting SOC saturation at high clay content (approx. > 40 % clay). At similar clay content, PG and MP had similar SOC content, whereas CR was depleted in SOC with a SOC:clay ratio of 9.8 ± 3.3 % (Table 1). The slope of the relationship between SOC and clay was steeper in CR than in PG and MP (Fig. 1a), suggesting greater SOC depletion and thus greater SOC sequestration potential for CR sites with low clay content. SOC:clay ratios were below 1:10 in 61 % of CR, whereas this limit was reached in only 16 % of PG and MP sites (Fig. 1b). In CR, 23 % of sites had a ratio even below 1:13, compared to only 5 % and 4 % of PG and MP, respectively. All except one PG and all MP sites with ratio below 1:13 had very high clay contents (≥ 56 %).

Soil organic matter stoichiometry was affected by both land use and SOC content (Fig. 2). The relationship between C:Porg and SOC in MP differed strongly from the relationship in CR and PG (Fig. 2c). While C:Porg increased with increasing SOC, the opposite occurred in MP. C:Porg in MP was higher and much more variable than in CR and PG. Hence, C:Porg data had to be log-transformed to meet normality assumption. This

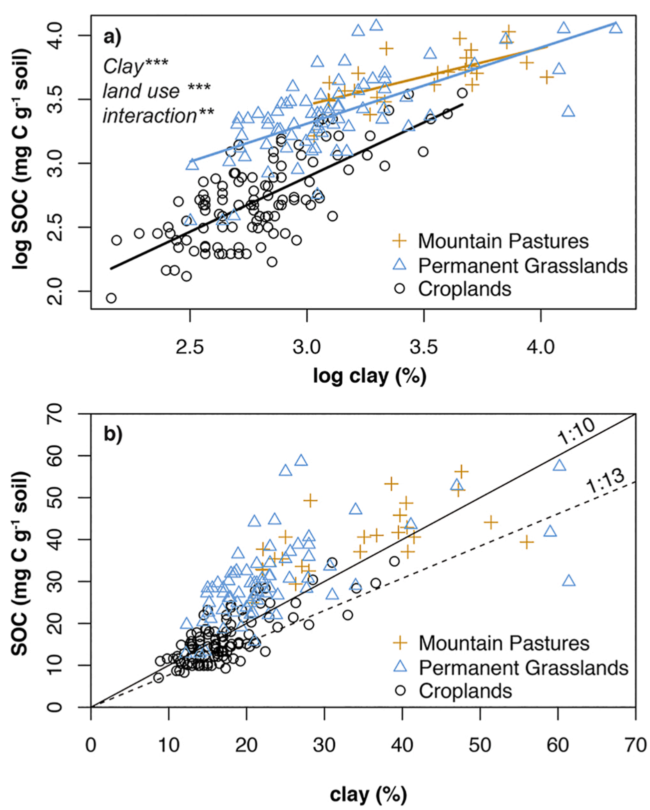

Fig. 1. Relationship between SOC and clay content (n = 209) for the three major land-use types (log-transformed values in panel a, and absolute values in panel b). The significance effect of clay content, land-use type and their interaction are indicated by asterisks (** = p < 0.01, *** = p < 0.001) based on ANCOVA. Solid line 1:10 represents the level at which SOC:clay ratio (mg mg^{-1}) should be improved, whereas the dashed line 1:13 is the level of the SOC:clay ratio that is considered unsuitable for soil structure according to Johannes et al. (2017a).

was not necessary when only CR and PG were included in the model. In this case, the land-use type (CR and PG) had no effect on the C:Porg (n = 184, p-values > 0.85) and the ratio increased by 0.38 ± 0.10 unit per 1 mg g^{-1} soil increase. The effects of land-use and SOC on N:Porg were the same as for C:Porg, with an increase in CR and PG of 0.048 ± 0.011 unit per increase of 1 mg g^{-1} soil (Fig. 2d). The main difference between C:Porg and C:Ptot was that the relationship of C:Ptot with SOC was different in CR and PG (Fig. 2b). The slope was 0.22 ± 0.11 higher in CR than in PG. Moreover, the model for C:Ptot (R^2adj: 0.60) explained much more variation in CR and PG than the model for C:Porg (R^2adj: 0.10). When considering PG and CR, C:Porg variability between land-use was small and the variability within land-use was high, while the opposite occurred for C:Ptot (Table 1). C:N ratio was affected by land-use, but not by SOC content in PG and MP (Fig. 2a). However, SOC content had an effect in CR where C:N increased with decreasing SOC (slope: -0.045 ± 0.014).

3.2. *SOC dynamics*

SOC changes during the 30-year long monitoring were significant in 33 sites, i.e. 16 % of the selected 209 sites. Six of them were significant only with Spearman correlation tests, suggesting that changes in those sites were not linear. All significant SOC changes in PG were positive (0.37 ± 0.04 mg C g^{-1} soil yr^{-1}), whereas 64 % of significant changes in CR were negative (-0.11 ± 0.06 mg C g^{-1} soil yr^{-1}) (Fig. S2). A sensitivity analysis was carried out to determine the minimum detectable rate of SOC changes at a power of 0.8 for sites where no significant changes of SOC over the 30 years had been detected. The median minimum detectable rate was 0.17 mg g^{-1} soil yr^{-1}, corresponding to a standard deviation (SD) of SOC in the 6 sampling cycles of 1.9 mg g^{-1} soil (Fig. 3a). The minimum detectable rates ranged from 0.03 to 1.47 mg g^{-1} soil yr^{-1}, but 92 % of them were below 0.5 mg g^{-1} soil yr^{-1}. Assuming linear changes of SOC with time and no change of the model with time (i.e. constant slope, intercept and SD of residuals), the minimum detectable change in SOC shows approximatively a logarithmic decrease with time, reaching 0.09 mg g^{-1} soil yr^{-1} at the 13[th] cycle in 2052 (Fig. 3b). The number of sites exhibiting a significant linear change of SOC would strongly increase to 77 % of the well-drained sites. Change in SOC would not be detected in 24 % of MP and CR sites, but only in 16 % of PG sites.

3.3. *Temporal changes of land-use type*

The proportion of sites as grasslands, i.e. temporary grasslands within crop rotations (TG) and permanent grasslands (PG) combined, ranged from 51 % in 1994 to 67 % in 2010, with an increasing trend over-time (Fig. S1). Only 53 out of 77 sites classified as PG in the 6[th] cycle remained as such over the 30 years of the monitoring and, subsequently, were named PGr sites. Thirty-four sites during this period experienced at least one conversion between cropland and permanent grassland and, subsequently, were named LUC sites (Fig. 4). Intensive LUC sites (LUC_{int}) experienced at least one conversion, but remained cropland for most of the time. Nonetheless, the proportion of TG in the crop rotation was high (69 %), often with long TG sequences of several years (Table S1). LUC_{int} was the land-use type with the highest increase of $Grass_{ratio}$ between the first and the second half of the monitoring ($Grass_{dyn}$: 21 %), but also the one with the most extreme positive and negative value of $Grass_{dyn}$. The category extensive LUC (LUC_{ext}) included sites that had been cropped once for one or several consecutive years before returning to PG or that had been only recently converted to CR. They were on average 88 % of the time under grasslands, but $Grass_{ratio}$ remained fairly constant ($Grass_{dyn}$: 1 %). The two cropland categories which included sites that had always remained croplands (CRr) differed mostly in the proportion and the length of TG within the crop rotation (Table S1). The average $Grass_{ratio}$ reached 41 % in the category CR_{ext} and slightly increased in the second half of the monitoring period, whereas the average $Grass_{ratio}$ was only 8 % in CR_{int} and

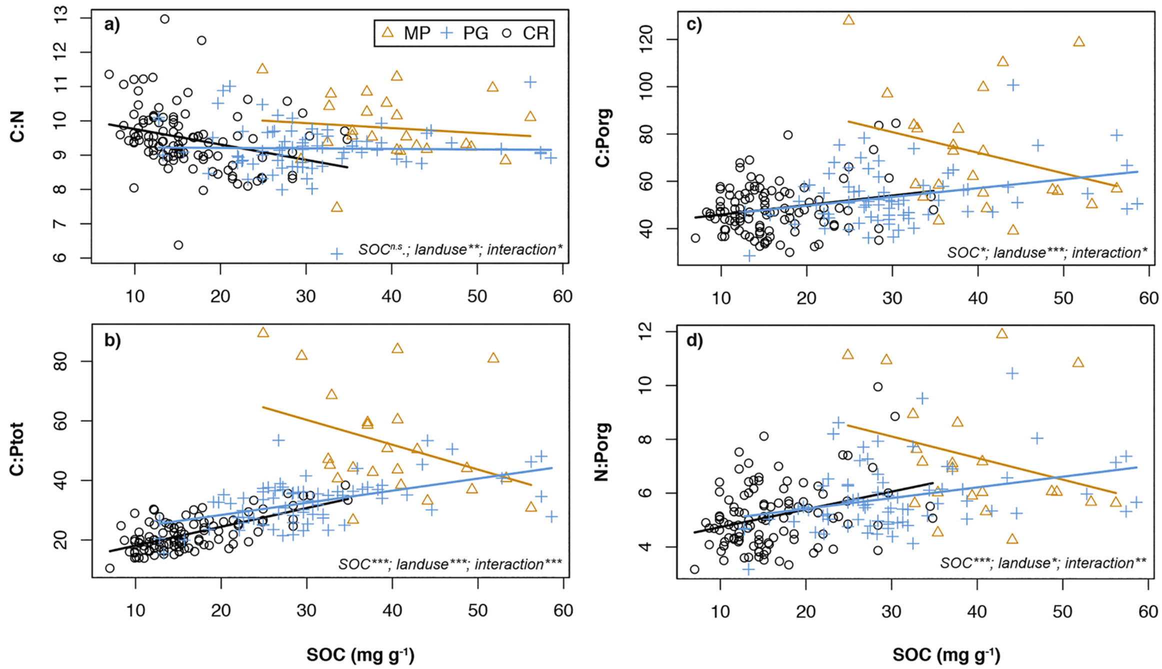

Fig. 2. Effect of land use and SOC content on SOM stoichiometry. Differences between croplands (CR), permanent grasslands (PG) and mountain pastures for a) C:N, b) C:Ptot, c) C:Porg, and d) N:Porg ratios were assessed by ANCOVA (n = 209) on log-transformed data (C:Porg and C:Ptot) or by permutation tests (N:Porg and C:N). Asterisks at bottom right corner of each panel indicate significance levels at p-values < 0.001 (***), < 0.01 (**), < 0.05 (*).

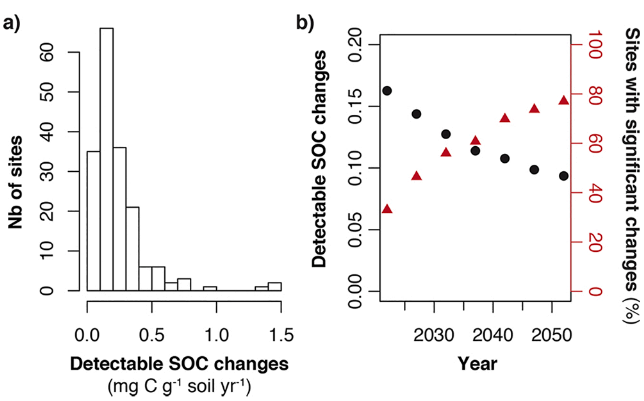

Fig. 3. Sensitivity analysis of minimum detectable rates of SOC change. a) Minimum detectable SOC changes for the 6th sampling cycle for sites that did not show significant SOC changes over 30 years (n = 179). b) Projection for the subsequent soil sampling cycles (7–13) of the median minimum detectable rates (dots) and of the proportion of sites (n = 209) with significant SOC changes (triangles), assuming that the parameters of the linear model determined with cycles 1–6 remain constant in subsequent cycles.

slightly decreased in the second half. The two CRr categories could not be distinguished by the entropy nor by the turbulence. However, these two indices were different between the two LUC categories, with lower values in LUC_{ext}.

3.4. Factors influencing SOC dynamics

SOC contents and accumulation rates differed between land-use types (Fig. 5). In the 6 categories (CR_{int} and CR_{ext}, LUC_{int}, LUC_{ext}, PGr and MP), minimum accumulation rates ranged from -0.61 to -0.21 mg g^{-1} soil yr^{-1} and maximum accumulation rates ranged from 0.14 to 1.32 mg g^{-1} soil yr^{-1}. While SOC content showed a gradient from CR_{int} to MP similar to that of $Grass_{ratio}$ (Fig. 5a), accumulation rates in LUC_{ext} were low, with a slightly negative median similar to CR_{int} (- 0.02 C g^{-1} soil yr^{-1}; Fig. 5b). By contrast, most sites showed a positive SOC accumulation in the other land-use types. Median rates corresponded to changes of the initial SOC content by -1.5, 1.7, 6.8, -0.2, 9.1 and 7.7‰ yr^{-1} in CR_{int}, CR_{ext}, LUC_{int}, LUC_{ext}, PGr and MP, respectively. Nonetheless, each land-use type included sites with lower or higher rates than zero and most rates were not significantly different from zero (Fig. 5). The proportion of sites that would still not show significant SOC changes by 2050 is similar in CR_{int}, CR_{ext} and MP, with 28, 26 and 24 %, respectively. About half of the LUC_{ext} (45 %) would not change, whereas SOC changes would be detected in 91 % of LUC_{int} and 89 % of PGr sites by the middle of this century.

In order to understand which factors mostly explained the rate variability under different land-use types, we applied a multiple regression using soil, site and land use explanatory variables that were not highly correlated (r < 0.70). Model fits (R^2) reached 0.50 for PGr, 0.66 for CRr + lUC categories (CR_{int} + CR_{ext} + LUC_{int} + LUC_{ext}) and up to 0.78 for MP (Table 2). The initial SOC content ($SOC_{initial}$), i.e. SOC measured during the 1st cycle, was the variable that explained most of the rate variability for all categories (Table 2; Fig. 6). It was also the only variable, except for the slope of MP that negatively affected SOC accumulation rates. $SOC_{initial}$ was by far the main explanatory variable for sites that had been cropped for at least 1 year (CRr + LUC) and for sites that remained croplands over the 30 years (CRr), explaining respectively 2.7 and 2.4 times more than the second most explanatory variable. Although the importance of $SOC_{initial}$ in explaining SOC accumulation rates varied among categories, the effect of $SOC_{initial}$ remained constant, between -0.014 to -0.015 mg g^{-1} soil yr^{-1} per increase of 1 mg g^{-1} soil, in PGr, LUC and CRr.

The proportion of grassland ($Grass_{ratio}$) and the increase of grassland overtime ($Grass_{dyn}$) were important factors explaining SOC dynamics in cropped sites. The SOC accumulation rates increased at least by 0.015 mg g^{-1} soil yr^{-1} for 10 % increase of temporary grasslands that were incorporated in the crop rotation (Table 2; Fig. S3). Land use-related variables had a large influence on SOC dynamics in LUC sites, while in CRr soil variables like Ptot and clay had a large influence on SOC dynamics. For the three crop models, the crop type (CER and HOE), the turbulence and the entropy had no significant impacts on SOC

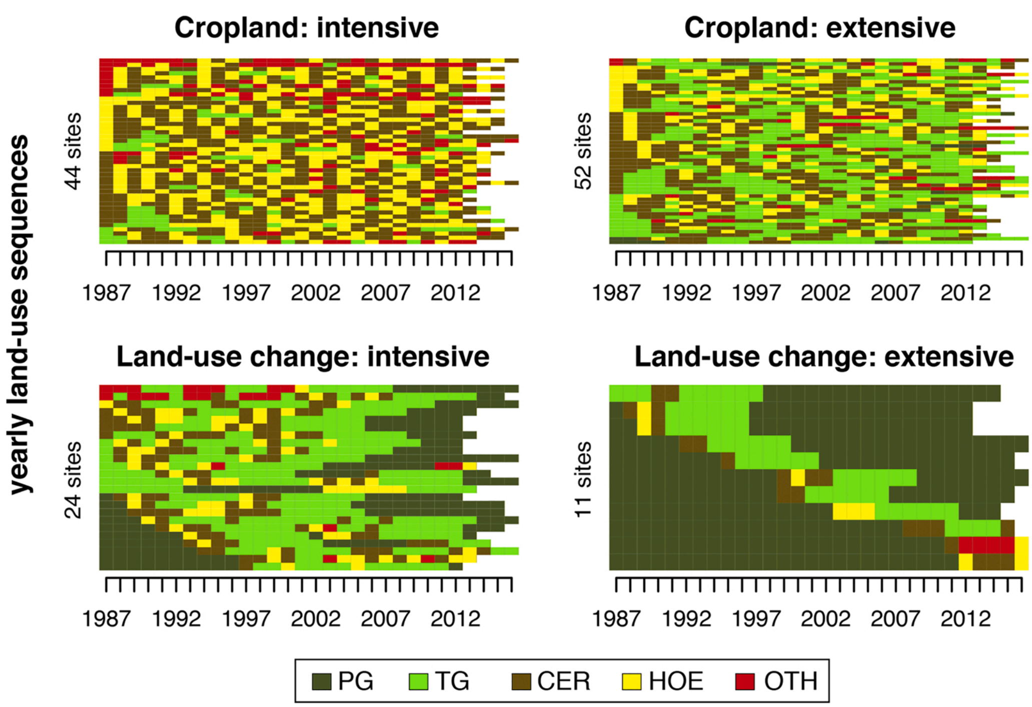

Fig. 4. Land use trajectories of the 131 sites that have been cropped for at least 1 year between 1987 and the 6th sampling campaign, excluding vineyards. Clustering was done using distance matrix based on the longest common sequence of land uses. Clusters 2 and 4 (bottom) represent sites with land-use change from cropland to permanent grassland or vice versa, while clusters 1 and 3 (top) represent croplands with long or short sequences of temporary grassland in the crop rotation, respectively.

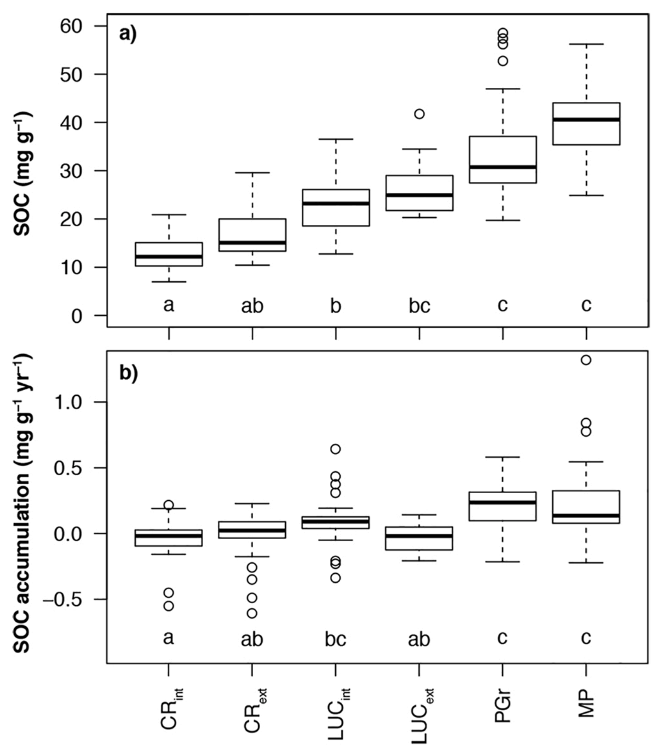

Fig. 5. Impacts of land-use categories on a) SOC content, and b) SOC accumulation rate. The four first categories from the left-hand side are sites that have been cultivated for at least 1 year and have been grouped by clustering (see Fig. 4). PGr includes sites that always remained permanent grasslands. Letters indicate significant differences ($p < 0.05$) determined by Kruskal-Wallis test.

accumulation rates. Site conditions such as terrain slope and elevation had almost no impact on the SOC dynamics in the cropland-grassland system, except for a small effect of elevation in PGr.

SOC dynamics in PGr, were similarly driven by Ptot and $SOC_{initial}$ and slightly less by pH. The effect of Ptot in PGr was similar to the one in CRr (0.18 ± 0.4 vs. 0.15 ± 0.5 C g^{-1} soil yr^{-1} per increase of 1 mg P g^{-1} soil). By contrast, SOC dynamics in MP, besides $SOC_{initial}$, were also driven by variables inherent to site conditions, such as clay, terrain slope and elevation.

4. Discussion

4.1. Current soil organic carbon content

The concentrations of SOC that we found in croplands (17 ± 7 mg g^{-1}) and permanent grasslands (31 ± 10 mg g^{-1}) were similar to the levels reported for corresponding land use types under comparable climate despite different scales and sampling designs (Capriel, 2013; Leifeld et al., 2005; Tóth et al., 2013). The ratio SOC:clay has been identified as a good indicator of soil physical quality (Johannes et al., 2017a). Thresholds of SOC:clay ratio have been determined by another study conducted in the same study region for Cambisols in agricultural fields of the Swiss Midland, including a subset of FRIBO sites (Johannes et al., 2017a). The majority of grasslands in our study area had a sufficient level of SOC (SOC:clay ratio > 1:10) to enable good soil structure (Fig. 1b). By contrast, SOC content in the majority of croplands was low with SOC:clay ratios < 1:10. Below this threshold, structure degradation may reduce structural porosity and hydro-structural stability of aggregates, affecting water content and oxygen diffusion (Johannes et al., 2019, 2017b). As temporary hypoxia is likely to occur when SOC:clay ratio is below < 1:13, plant productivity might be limited by soil structure degradation, at least in the most severe cases (25 % of croplands sites), due to the negative impacts on roots and soil microorganisms (Drew, 1997).

The evolution of SOC between 1987 and 2016 in the FRIBO network, however, contrasts with the general declining trend in agricultural soils observed in other soil monitoring networks (Capriel, 2013; Heikkinen et al., 2013), long-term experiments (Emmel et al., 2018; Keel et al., 2019; Maltas et al., 2018), and modeling studies (Wiesmeier et al., 2016). For example, average SOC losses in Finnish croplands (4‰ yr^{-1} of the SOC content) were higher than the highest average losses found in the FRIBO cropland categories (Heikkinen et al., 2013). A previous study

Table 2
Factors influencing SOC accumulation rates in various land-use categories determined by multiple stepwise regressions.

		R^2	Estimates[a]	p-values
Cropped sites (CRr + LUC) n = 121	*Model*	*0.66*	*$-2.7 \pm 1.5 \times 10^{-1}$*	*< 0.001*
	$SOC_{initial}$[b]	0.54	$-2.2 \pm 0.2 \times 10^{-2}$	< 0.001
	$Grass_{ratio}$	0.20	$2.5 \pm 0.5 \times 10^{-3}$	< 0.001
	$Grass_{dyn}$	0.16	$1.8 \pm 0.4 \times 10^{-3}$	< 0.001
	Ptot	0.14	$2.2 \pm 0.5 \times 10^{-1}$	< 0.001
	Clay	0.06	$5.5 \pm 2.2 \times 10^{-3}$	< 0.05
	Elevation	0.03	$1.9 \pm 1.1 \times 10^{-4}$	< 0.10
	pH	0.02	$2.7 \pm 1.7 \times 10^{-2}$	< 0.10
	entropy / slope	removed during the stepwise regression		
Croplands (CRr) n = 81	*Model*	0.42	$-1.4 \pm 0.5 \times 10^{-1}$	< 0.001
	$SOC_{initial}$	0.29	$-1.5 \pm 0.3 \times 10^{-2}$	< 0.001
	Ptot	0.12	$1.5 \pm 0.5 \times 10^{-1}$	< 0.01
	$Grass_{ratio}$	0.10	$1.5 \pm 0.5 \times 10^{-3}$	< 0.01
	Clay	0.09	$2.3 \pm 0.6 \times 10^{-3}$	< 0.01
	Elevation	0.04	$1.7 \pm 0.9 \times 10^{-4}$	< 0.10
	$Grass_{dyn}$	0.03	$6.6 \pm 4.4 \times 10^{-4}$	< 0.15
	entropy / CER / pH / slope / turbulence	removed during the stepwise regression		
Land-use change (LUC) n = 34	*Model*	0.56	$-4.1 \pm 1.0 \times 10^{-1}$	< 0.001
	$SOC_{initial}$	0.36	$-1.4 \pm 0.3 \times 10^{-2}$	< 0.001
	$Grass_{dyn}$	0.36	$3.1 \pm 0.8 \times 10^{-3}$	< 0.001
	CER	0.12	$-6.0 \pm 3.0 \times 10^{-3}$	< 0.10
	Elevation / $Grass_{ratio}$ / HOE / pH /slope	removed during the stepwise regression		
Permanent grasslands (PGr) n = 53	*Model*	*0.50*	*-1.3 ± 0.4*	*< 0.001*
	$SOC_{initial}$	0.38	$-1.5 \pm 0.3 \times 10^{-2}$	< 0.001
	Ptot	0.35	$1.8 \pm 0.4 \times 10^{-1}$	< 0.001
	pH	0.28	$3.7 \pm 0.8 \times 10^{-1}$	< 0.001
	Elevation	0.13	$5.6 \pm 2.1 \times 10^{-4}$	< 0.05
	Clay / Slope	removed during the stepwise regression		
Mountain pastures (MP) n = 22	*Model*	*0.78*	*$7.9 \pm 3.0 \times 10^{-1}$*	*< 0.001*
	$SOC_{initial}$	0.77	$-6.1 \pm 0.8 \times 10^{-2}$	< 0.001
	Clay	0.58	$2.7 \pm 0.6 \times 10^{-2}$	< 0.001
	Slope	0.32	$-7.9 \pm 2.9 \times 10^{-3}$	< 0.05
	Elevation	0.24	$5.4 \pm 2.4 \times 10^{-4}$	< 0.05
	Ptot	0.11	$2.1 \pm 1.5 \times 10^{-1}$	< 0.20
	pH	removed during the stepwise regression		

[a] Estimates correspond to the slope for variables and to the intercept for the model. Models were performed on standardized data, but estimates are presented as non-transformed data.

[b] ($SOC_{initial}$; mg g-1), temporary grassland proportion in the rotation ($Grass_{ratio}$; %) and its increase between the 1st and 2nd half of the monitoring period ($Grass_{dyn}$; %), total P (Ptot; mg g-1), clay (mg g-1), elevation (m), pH, entropy, slope (degrees), turbulence, proportion of cereals in the rotation (CER, %) and hoe crops (HOE, %).

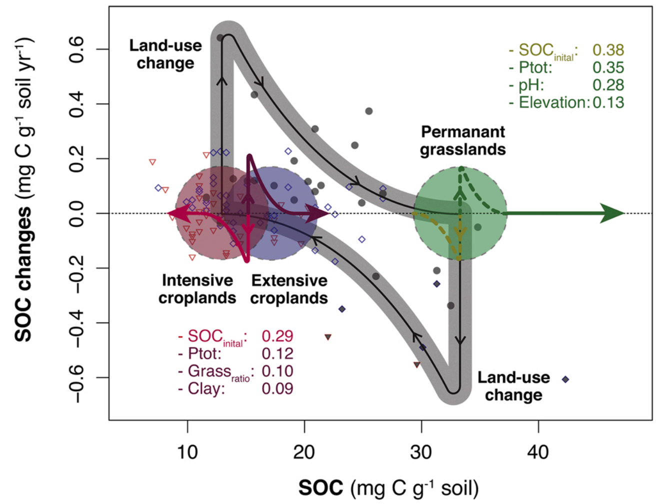

Fig. 6. Conceptual figure representing the relative effect (dominance) of land-use change over management effect on SOC dynamics in a temperate cropland-grassland agricultural system. A change of land-use (LUC_{int} sites; black arrows) or management (CRr and PGr; colored arrows) induces a relatively high initial change in SOC rate that decreases over time approaching a new equilibrium. Vertical arrow heights in land-use change sites (LUC_{int}) and permanent cropland sites (CRr) represent the rate change over time observed under those categories. The permanent grasslands (PGr) is not included due to high uncertainties related with the legacy of past land-use change effects on rates. Symbols represent study sites initial SOC content ($SOC_{initial}$) and the rates of SOC changes for the intensive land-use change sites (LUC_{int}, grey-filled circles), intensive cropland sites (CR_{int}, red triangles) and extensive cropland site (CR_{ext}, blue diamonds). Filled triangles and diamonds shape symbols represent the six-permanent cropland CRr sites that exhibited land-use change characteristics. The average SOC content at the 6[th] sampling cycle for CR_{int}, CR_{ext} and PGr are represented by the center of the large colored circles (circle diameters have no meaning). Colored arrows show the main direction of SOC changes for each category. The rate of SOC changes represented by the arrow is proportional to the median rate of the respective category. The main factors affecting SOC change rates in croplands (CRr) and grasslands (PGr) are displayed with their partial eta-square values. Ptot: total P (mg g^{-1}), $Grass_{ratio}$: the percentage of temporary grassland within the crop rotation. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

on 30 croplands in Switzerland reported significant SOC changes from -12 to +11‰ yr^{-1} (Gubler et al., 2019), a range much narrower, especially for positive rates, than the one observed in the FRIBO network (-20 to +50‰ yr^{-1}). The highest loss in our study occurred in a site that did not include temporary grassland in the rotation since the 1990′s and had a high initial SOC:clay ratio (> 1:7), while the highest gain occurred in a site that was initially strongly depleted in SOC (SOC:clay ratio > 1:15), but turned into a grassland in 2000. One of the reasons explaining our observed data is the role of animal husbandry in Swiss agriculture, especially in the Fribourg Canton, that can provide high amounts of organic manure. The high manure availability (72 % of the farmes include animal production) represents an important source of nutrients in croplands and grasslands (Sinaj and Richner, 2017; Spiess, 2011). As a consequence, soil C inputs are likely higher than in regions dominated by crop production with little return of plant residues (through organic manure) to the soil.

4.2. Effect of land use and land-use change

Land-use change was responsible for most of the highest increase or decrease in SOC content. The LUC categories included sites that had low SOC content and high SOC accumulation rates when converted to grassland and sites with high SOC content and high SOC loss rates when converted to CR (Fig. 6). This explains the negative relationship between the initial SOC content and SOC changes. The predominance of the initial SOC content over other factors in sites that have been cropped for at least 1 year highlights the general strong effect of cropland-grassland conversion on SOC dynamics. Rates of SOC gain or loss are the highest immediately after conversion and tend to slow down over time until they reach a new equilibrium (Chenu et al., 2019). Thus, management-associated effects were not detectable in LUC sites (Table 2) as SOC dynamics was fully driven by variables related to land-use change ($SOC_{initial}$ and $Grass_{dyn}$). Some studies have found higher SOC losses than gains after conversion in the grassland-cropland system (Poeplau and Don, 2013). In our study, cropland sites that were converted to grasslands showed higher accumulation rates compared to loss rates when grassland was converted to cropland (Fig. 6). Sites with the fastest rates of SOC loss were found in CRr (up to 0.6 mg g^{-1} soil yr^{-1}), but they had been likely converted from grassland shortly prior to monitoring as they exhibit the characteristics of this land-use change, i.e. high initial SOC and fast SOC loss after the conversion.

SOC dynamic in permanent grasslands in Great Britain (Bellamy et al., 2005), Belgium (Lettens et al., 2005), Bavaria (Capriel, 2013) and New Zealand (Schipper et al., 2010) was highly variable with higher and/or lower gains or losses depending on sites. However, almost all PG in our study showed increasing SOC trends. Thus, 85 % of them may experience a significant SOC gain by 2050, while only 4 % may experience significant loss. The fast SOC accumulation rates observed in several European grasslands (Soussana et al., 2007) were attributed to the legacy effects from past land-use or management changes (Smith, 2014). In our study, past land-use changes, if any, occurred more than 30 years ago. Because small SOC accumulation can still occur up to a century after land-use change (Poeplau et al., 2011; Smith, 2014), land-use change could slightly influence SOC dynamics in permanent grasslands, but could not be the main driver for the observed strong SOC accumulation, even in sites with high SOC content and SOC:clay ratios (Fig. S3). As Ptot and pH together explained more variability than the initial SOC content, the SOC accumulation appears to be mainly driven by fertilization practices (Jeangros and Sinaj, 2018a, 2018b; Sinaj and Richner, 2017). Beside favoring plant productivity and, in turn, C input to the soil, P availability increases the growth of legume species in grasslands (Coonan et al., 2019), leading to narrower SOM C:N ratio and favoring soil C input stabilization (Poeplau et al., 2018). Accordingly, the high nutrient surplus from the past that resulted in an important P enrichment in PG (Table 1) could have a legacy effect on the current SOC accumulation in PG (Spiess, 2011). For example, Spohn (2020b) reviewed the interaction of P and SOM and suggested that this interaction is complex because the storage of SOC, particularly in mineral soils, may require large amounts of organic P. Other factors, however, must drive SOC dynamics in PGr since 50 % of the variability was not explained by the model. The mix of species, which is known to affect SOC accumulation (Conant et al., 2017), is not directly manipulated by farmers because these grasslands are not sown, but they might have been modified by changes in fertilization practices and grazing pressure over time (Soussana et al., 2010). Finally, grasslands might have moved away from steady-state by the effect of climate change on the balance between plant productivity and SOC mineralization (Buttler et al., 2019).

For the cropland sites with no record of land-use change, the SOC content still responds to management practices. The increase of fertilization, as suggested by Ptot, and the inclusion of temporary grasslands within crop rotation favors SOC accumulation, especially for soils depleted in SOC (Table 2). These two management practices are more efficient especially on clay soils with low SOC content. The positive effect of P in croplands and grasslands, may be also partly and indirectly due to the supply of organic amendments, which are an important C source for agricultural soils in Switzerland (Maltas et al., 2018). The discrepancies between the SOC dynamics in croplands as observed in our study and SOC losses as reported in most of the long-term Swiss experiments (Emmel et al., 2018; Keel et al., 2019) could be explained by the low amount of temporary grasslands included in long-term experiments within the intensive croplands category (Table S1). In our study the positive effect on SOC dynamics was observed when the site, in relation to the crop rotation, was under temporary grassland for more than 30 % of the time (Fig. S3). The fact that crop identity (cereals and hoe crops) did not significantly affect SOC rates does not mean that improved cultivation practices (reduced-tillage, cover crops, residues left on site, species mix in TG, etc.) cannot result in SOC gain. Unfortunately, cultivation practices in our study were not recorded. Crop identity has been reported to be a factor in previous studies (Gubler et al., 2019). The absence of crop type effect in our study may rather suggest that the cultivation systems (agroecology, conservation agriculture, organic farming, etc.) can potentially overcome the differences resulting from crop identity (plant-derived soil C inputs, specific management related to the species, etc.). The model explained less than half of the variability in CRr, indicating that some factors were not identified, or properly represented, by the selected indicators.

4.3. Soil stoichiometry in cropland-grassland systems

Carbon-to-nutrient mass ratios (C:N:P) in FRIBO cropland (22:2:1) were comparable to ratios found in cropland (25:2:1) by meta-analysis, but FRIBO grasslands (33:4:1) were much richer in Ptot than the reviewed grasslands (55:5:1) (Xu et al., 2013). The increase of C:Porg and N:Porg ratios with SOC content, especially for CR and PG, demonstrates a dilution effect of P in SOM when SOC increases (Fig. 2). This means that the P requirement for increasing C sequestration in soils decreases with increasing SOC content. By contrast, increases in C sequestration would require a proportional amount of N increase at high SOC content (c. > 2 %) and even more N at low SOC content, especially in croplands (Fig. 2a). The high C:N ratio at low SOC content contradicts, at first glance, the general trend that C:N ratio decreases with increase of SOM decomposition (Tipping et al., 2016). The fresh soil C input from a crop depends on the yield rather than on the SOC content (Keel et al., 2017). Thus, within croplands, fresh soil C inputs that have higher C:N ratio might contribute by higher proportion to SOC pool relative to the mineral-associated pool that has lower C:N ratio (Kirkby et al., 2013) in soils more depleted in SOC.

Frossard et al. (2016) suggested that soil stoichiometry was not controlled by soil input stoichiometry, but rather by site conditions and soil properties. This is in line with the findings that land-use in the grassland-cropland system did not affect SOM stoichiometry (C:Porg and N:Porg), but only C:Ptot ratio (Fig. 2). The lower C:Ptot ratio in croplands means that extra C can be sequestered without additional P inputs if the excess of mineral P can be mobilized and integrated into SOM pools by microorganisms directly or via plant biomass. This would partly mitigate the P fixation required to sequester C, based on SOM stoichiometric constrains (Spohn, 2020b).

4.4. Increasing SOC in grassland-cropland systems

The strong SOC content increase in the top 20 cm of croplands converted to grasslands or to permanent grasslands does not necessarily translate to a comparable increase rate of SOC stock along the soil profile because of SOC redistribution and changes in soil bulk density (Don et al., 2009). On one hand, the increase of SOC in the top soil of grassland might be at the expense of the subsoil because of changes in rooting systems and tillage (Don et al., 2009). On the other hand, SOC content in grassland might be slightly underestimated because they are generally more compacted (Lee et al., 2009). The differences in SOC content in the top soil between croplands and grassands were significant to confirm the SOC trajectory, however, an accurate quantification of the exact amount of SOC gains would require accounting for changes in SOC redistribution to deeper soil layers and differences in soil bulk density. While more data on SOC redistribution with depth and changes in soil bulk density are needed to evaluate the SOC sequestration potential, the increase of SOC content after cropland conversion to grassland is undeniably positive for soil quality and resilience to climate change.

An efficient measure to trigger SOC accumulation in the cropland-grassland system is to increase the proportion of temporary grassland during the crop rotation as the main manageable factor affecting SOC dynamics. While this measure would favor climate change mitigation by increasing soil C sequestration, it would decrease agricultural production in favor of animal production, which needs more surface area and other resources to produce the same amount of calories (Godfray et al., 2018). It could also affect the P balance in soils (Spohn, 2020b). Hence, there are tradeoffs between climate change mitigation and food security objectives for large-scale increase of temporary or permanent grasslands in a grassland-cropland system.

To overcome the effects of such tradeoffs, the most readily applicable measure is to increase SOC in croplands by re-balancing organic amendments between permanent grassland and croplands. Permanent grasslands have already adequate levels of SOC to enable good structure quality, favoring soil fertility and climate change adaptation. They are very rich in P, as well as other nutrients such as K and Mg that are abundant in manure and slurry, and they risk nutrient leakage (Blanchet et al., 2017; Frau et al., 2020; Jeangros and Sinaj, 2018a; Roger et al., 2014). Accordingly, the objective for permanent grasslands would be to preserve the high SOC already sequestered (Smith, 2014) rather than increasing SOC by applying organic amendments in excess of nutrient demands. As such, redirecting the surplus of organic amendments from permanent grasslands to croplands would favor climate change mitigation and adaptation as well as food security without consuming additional resources.

The monitoring of SOC dynamics and the ability to validate the pertinence of management practices to increase C sequestration remains a crucial question (Paustian et al., 2019; Smith et al., 2020). The lessons learned from the FRIBO monitoring network indicate a temporal mismatch between the sensitivity of such networks to detect SOC changes (several decades) and the evaluation of farmers' practices by tracking SOC changes (yearly to a decade). Nonetheless, the FRIBO network was sufficiently sensitive to determine factors affecting SOC dynamics.

In the light of our study, linking governmental subsidies to practices favoring overall soil C sequestration is more realistic and rewarding than linking the subsidies to actual SOC changes quantified by direct SOC measurements. Thus, the humus balancing methodology suggested in Switzerland (FOAG, 2020) needs to be updated as several driving factors are not included (e.g. soil nutrients), nor adequately weighted (e.g. percentage grassland vs hoe crops, pH), (Neyroud et al., 1997). The quantification and the prioritization of factors driving SOC dynamics as determined by this study, as well as the large body of literature on this topic produced in the last 25 years, should enable a strong improvement of the accuracy of the humus balancing methods and create incentives for farmers to sustainably manage SOC resources.

5. Conclusion

Soil organic matter dynamics and stoichiometry responded to land use changes and agricultural practices (crop rotation, fertilization). The response, whether positive or negative, was more pronounced after land-use change. The initial SOC content was the main predictor of SOC accumulation rates. When approaching a new steady-state condition, SOC dynamics responded to agricultural practices such as temporary grasslands or fertilization. Site conditions played an important role on the SOC dynamics only for the less disturbed agroecosystems such as mountain pastures, but were determinant (e.g. clay content) for the SOC level in steady-states conditions. The role of land-use change and agricultural practices prior to the establishment of our soil sampling network, as well as the role of climate change, were not analyzed in our study, but they could partly explain the continuous SOC gains in PG despite their high SOC levels. As SOC levels in PG are already sufficient to ensure good soil quality (high SOC:clay ratio), the most efficient way to avoid negative tradeoffs between soil C sequestration and agricultural production goals is to redirect organic amendments from permanent grasslands to croplands. Until faster and more accurate measurements of SOC dynamics become available, humus balancing/budgeting, like nutrient budgets, can be implemented for governmental subsidy schemas to encourage soil C sequestration. In the long-term, a more detailed inventory of management practices combined with additional and more responsive soil properties and indicators, especially at field-level scales, should be studied for further improving SOC accounting.

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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Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:https://doi.org/10.1016/j.agee.2020.107184.

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