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Subarctic soil carbon losses after deforestation for agriculture depend on permafrost abundance

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Abstract

The northern circumpolar permafrost region is experiencing considerable warming due to climate change, which is allowing agricultural production to expand into regions of discontinuous and continuous permafrost. The conversion of forests to arable land might further enhance permafrost thaw and affect soil organic carbon (SOC) that had previously been protected by frozen ground. The interactive effect of permafrost abundance and deforestation on SOC stocks has hardly been studied. In this study, soils were sampled on 18 farms across the Yukon on permafrost and non-permafrost soils to quantify the impact of land-use change from forest to cropland and grassland on SOC stocks. Furthermore, the soils were physically and chemically fractionated to assess the impact of land-use change on different functional pools of SOC. On average, permafrost-affected forest soils lost 15.6 ± 21.3% of SOC when converted to cropland and 23.0 ± 13.0% when converted to grassland. No permafrost was detected in the deforested soils, indicating that land-use change strongly enhanced warming and subsequent thawing. In contrast, the change in SOC at sites without permafrost was not significant but had a slight tendency to be positive. SOC stocks were generally lower at sites without permafrost under forest. Furthermore, land-use change increased mineral-associated SOC, while the fate of particulate organic matter (POM) after land-use change depended on permafrost occurrence. Permafrost soils showed significant POM losses after land-use change, while grassland sites without permafrost gained POM in the topsoil. The results showed that the fate of SOC after land-use change greatly depended on the abundance of permafrost in the pristine forest, which was driven by climatic conditions more than by soil properties. It can be concluded that in regions of discontinuous permafrost in particular, initial conditions in forest soils should be considered before deforestation to minimize its climate impact.

KEYWORDS
Canada, chronosequence, climate change, fractionation, land-use change, soil organic matter, Yukon

1 | INTRODUCTION

With the increase in temperatures linked to climate change, agriculture is expected to shift poleward (Franke et al., 2021; Tchebakova et al., 2011). The cold and dry climate with a short vegetation period and widespread permanently frozen soils has hampered the development of a strong agricultural sector in the subarctic, leaving wide areas of subarctic forests so far untouched. However, especially in regions with sporadic to discontinuous permafrost, agricultural development might become more likely in the near future, although the environmental impacts of this are highly uncertain (Poeplau et al., 2019). Within the global carbon (C) cycle, soils of the northern permafrost region are very important due to the large amounts of preserved C they contain (Dutta et al., 2006; Tarnocai et al., 2009). The permafrost has protected soil organic matter from microbial breakdown since the last ice age (Dutta et al., 2006). Climate change is predicted to be most pronounced at high latitudes, where stronger warming than the global average is expected (Pepin et al., 2015), inevitably leading to the thawing of permafrost (Biskaborn et al., 2019) and a climate–carbon feedback (Davidson & Janssens, 2006; Heimann & Reichstein, 2008). Furthermore, deforestation promotes permafrost thaw by changing the microclimate because vegetation and the litter layer in natural forests act as insulating layers between the atmosphere and the soil. After deforestation, forced soil warming in summer can enhance permafrost thaw (Runyan et al., 2012). Therefore, land-use change from forest to agricultural land in the subarctic may accelerate the thawing of permafrost.

Globally, across ecosystems, conversion of forests to agricultural land leads to the depletion of soil organic carbon (SOC), with the highest losses per area in regions with high SOC stocks (Guo & Gifford, 2002; Smith, 2008). Moreover, conversion of forest to cropland has mostly been observed to decrease SOC stocks (Deng et al., 2016; Grünzweig et al., 2015; Guo & Gifford, 2002; Poeplau et al., 2011), while conversion of forest to grassland has been observed to increase or not change SOC stocks (Deng et al., 2016; Guo & Gifford, 2002). Specifically at high latitudes, there is some empirical evidence for SOC losses after conversion of forests to grassland (Grünzweig et al., 2004), which might be related to subsequent permafrost thaw. In regions of discontinuous permafrost, only a certain proportion of the soil under native vegetation is affected by shallow permafrost. It has been suggested that such initial conditions might strongly influence the direction and magnitude of SOC stock change after deforestation (Grünzweig et al., 2015). However, systematic and comprehensive studies on the interactive effect of permafrost abundance and land-use change on SOC stocks are lacking. Increased net primary productivity following land-use change may enhance the input of C into the soil (Köchy et al., 2015), as microclimate changes after deforestation and the fertilization of agricultural soils encourage plant growth. In contrast, permafrost loss as a consequence of a changed microclimate (Murton, 2021) may reduce SOC stocks at greater levels than can be offset by increased N

Soil organic carbon is a complex matter, consisting of many compounds with different chemical and physical properties. It is crucial to understand how these compounds react to environmental changes because positive and negative feedbacks between SOC and environment can reinforce or buffer such changes (Lavallee et al., 2020). A wide range of studies have evaluated the impact of either land-use change (Guimarães et al., 2013; Poeplau & Don, 2013; Wei et al., 2014) or permafrost thaw (Schuur et al., 2015; Xu et al., 2009) on SOC stocks and fractions. However, little is known about the interactions between land-use change and permafrost loss since many studies on permafrost focus on pristine ecosystems, free of direct anthropogenic impacts. Permafrost soils store large proportions of SOC in various forms of particulate organic matter (POM) (Höfle et al., 2013; Xu et al., 2009). Particulate organic matter is plant-derived organic matter with a lower density and greater susceptibility to microbial breakdown than mineral-associated organic matter (Lützow et al., 2007). Due to the low temperatures and often wet and anoxic conditions in the soil, the labile POM is well protected against microbial breakdown, but is quickly decomposed once the soil has been thawed (Ping et al., 2015). Effects of land-use change on SOC can vary greatly between fractions. Poeplau and Don (2013) observed that POM is most sensitive to land-use change, compared with total SOC and other SOC fractions. Accordingly, mineral-associated SOC fractions, such as the fraction attached to silt and clay particles, appear to be less sensitive to land-use change than total SOC. It was therefore hypothesized that the remaining SOC under converted land consists of relatively more silt- and clay-attached SOC, as POM is mostly removed by deforestation or quickly decomposed after the introduction of the new land use and loss of permafrost.

The aim of this study was to investigate the effect of deforestation on SOC stocks and fractions in the subarctic and how the presence of shallow permafrost drives SOC dynamics after land-use change. It is important to understand whether the abundance of permafrost plays a significant role in the response of SOC to land-use change in order to inform land-use strategies as well as earth system models. Based on the scarce available literature on permafrost agriculture, it was hypothesized that (i) land-use change leads to greater SOC losses in permafrost soils than in soils that are not affected by permafrost, (ii) initial losses of SOC due to deforestation might be offset in the long run, and (iii) SOC under converted land consists of relatively more carbon stored in the fine mineral fraction than SOC under native forest.

2 | MATERIAL AND METHODS

2.1 | Research area and farms

To test these hypotheses, the Yukon Territory in northwest Canada was chosen as a study area that is typical for land-use change in the subarctic. Yukon is located at the transition zone between continuous and discontinuous permafrost and is greatly affected by climate change (IPCC, 2013). Due to the gold rush at the end of the 19th century, the Yukon has some exceptionally old farms at this latitude in North America, while a young agricultural sector is also expanding due to growing demand for locally produced food (Yukon Agriculture Branch, 2020). Therefore, this area provides unique conditions for comparing land-use change on permafrost (here defined as soils that have detectable ice in the upper 80cm of the soil profile during sampling in midsummer) and non-permafrost soils. The existence of both old (>100 years) and fairly new (<30 years) farms allows land-use change effects to be assessed in a quasi-chronosequential, paired-plot approach (Poeplau et al., 2011). Most farms in the Yukon are located along river banks; therefore, there are few topographic or pedogenetic differences between the sampling points. In July of 2019, 18 farms located between the cities of Whitehorse and Dawson were selected for sampling (Figure 1). The farms' age (i.e. time since the forest was cleared), management and size cover a broad range of Yukon's agricultural sector. Croplands were small fields with vegetables, greens and herbs grown for local markets. Grasslands were used as pasture for livestock grazing (cattle, horses) or meadows for hay production. The common practice for preparing the land for agriculture was to cut down trees, pile up stumps and roots, and burn it all. Irrigation and application of locally produced (on-farm and at nearby farms) organic fertilizer (compost and manure) were common practices at most sites, with mineral fertilizer applied in only a few cases. Croplands were tilled occasionally using a rototiller to a depth of between 10 and 30cm (Table S1).

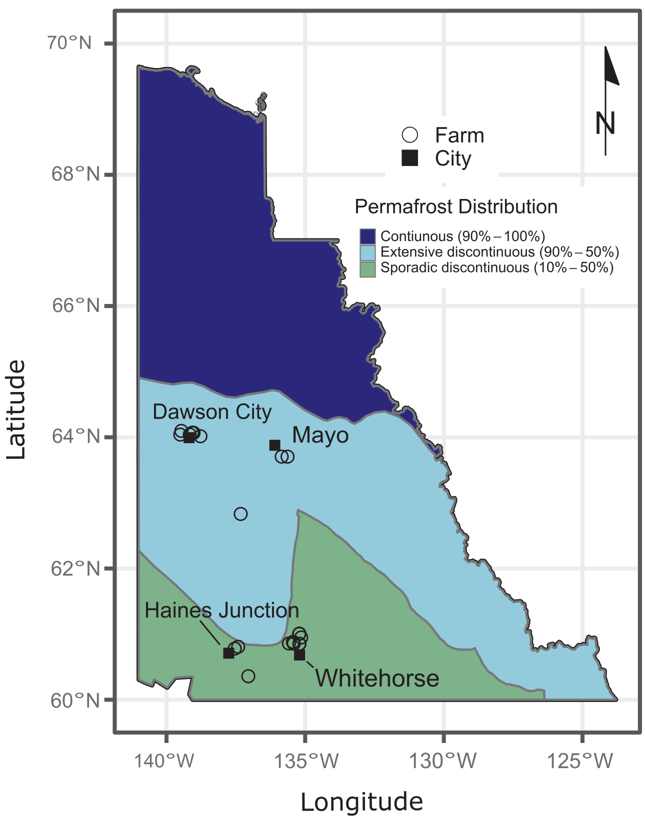


FIGURE 1 Yukon territory: Overview showing sample farms (circles), major cities (squares) and permafrost occurrence (colours) (modified from Heginbottom et al., 1995).

On each farm, a paired-plot design was established for sampling. Each pair of plots consisted of a forest as the reference and an adjacent cropland or grassland site or both. Forests were usually mixed wood forests of the boreal cordillera ecoregion with black (*Picea mariana*) and white spruce (*Picea glauca*), subalpine fir (*Abies lasiocarpa*), lodgepole pine (*Pinus contorta*), trembling aspen (*Populus tremuloides*), balsam poplar (*Populus balsamifera*) and paper birch (*Betula papyrifera*) (Smith et al., 2004). The shares of individual species were not determined at each farm, but the dominating tree type has been recorded (Table S1). For plot selection, an auger-based pre-assessment of the soil was undertaken, following the advice of the farmers which area of the farms might be most suitable for comparative sampling. To ensure comparability between forest and agricultural land, the focus of the pre-assessment was on soil texture and visible properties. Furthermore, the plots were selected within a maximum distance between forest and agricultural land of about 500m, and in many cases, the forest was directly adjacent to the agricultural field, making distances between the plots <50m. The third criterion for plot selection was a similar elevation and flat terrain at both forest and agricultural land to avoid effects of the relief, which could potentially lead to geomorphological related differences in SOC (Schiedung et al., 2022). At each plot, a slide hammer-driven soil corer with a diameter of 7 cm and a sample length of 20cm was used to sample five soil cores to a maximum depth of 80cm below the surface of the mineral soil (Figure 2). The soil cores were divided into five increments: 0–10 cm, 10–20cm, 20–40cm, 40–60cm and 60–80cm. Additionally, the litter layer of the forest floor was sampled using a metal ring 10 cm in diameter. In the centre of every forest plot, a soil profile was dug to a depth of 80cm, or as deep as possible if the permafrost or bedrock was at a shallower depth. Despite the fact that digging a soil profile to 80cm was not possible at every site, sampling with the soil corer could be done down to 80cm at every site, except for NB, where the bedrock was hit at 50cm. The permafrost depth at the date of sampling was determined visually (abundance of visible or tangible ice) in the soil pit. At 11 out of the 18 sites, permafrost was found in the soil cores of the forest, with an average active layer depth of 50cm. Seven out of the 18 sites had no permafrost within the first 80cm in the forest. The agricultural land plots generally had no permafrost within the uppermost 80cm, which was a strong indicator that land-use change encouraged the deepening of the active layer. Further indicators of permafrost loss upon land-use change, such as thermokarst, have been observed at one particular site. However, due to the relatively low ice content of the permafrost in this semiarid area, cryogenic soil or landscape features were scarce. At sites with permafrost, six grassland and nine cropland plots were sampled. At sites without permafrost, a total of six grassland and four cropland plots were sampled. A detailed overview with general site parameters can be found in Table 1. Management information regarding clearing, tilling, fertilization, irrigation and crop rotation (Table S1) was assessed by means of a short questionnaire, which was completed by 14 of the 18 farmers.

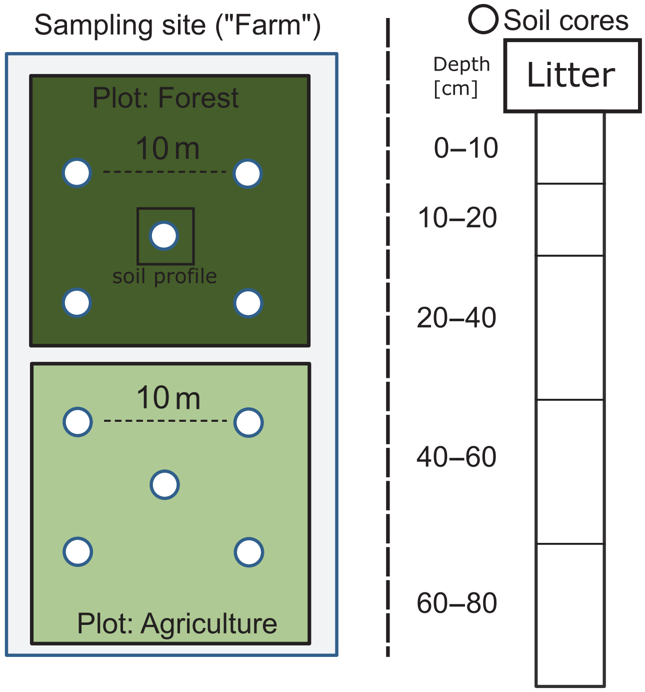


FIGURE 2 Sampling scheme. Five soil cores were sampled in each plot. The soil core consisted of five depth increments. The litter layer was also sampled with a metal ring (10 cm diameter).

2.2 | Laboratory analyses

2.2.1 | Sample preparation

Directly after sampling, soil was air-dried, weighed, sieved to ≤2 mm and weighed again to calculate the fine soil mass, stone and root content, and bulk density from the mass proportions and core volume. An aliquot of fine soil was dried at 105°C to correct for residual water content in the samples. All subsequent analyses were performed on the ≤2 mm sieved fine soil samples. Aliquots of the five field replicates of each plot and depth ($n = 1077$) were pooled to a mixed sample ($n = 216$). Carbon (C) and nitrogen (N) content were analysed in all samples (individual field replicates and mixed samples), while all the other analyses were performed on the mixed samples only.

2.2.2 | Fractionation

Fractionation was performed on mixed samples from the depth of 10–20 cm (referred to as topsoil) and from 40 to 60 cm (referred to as subsoil). The second depth increment was selected rather than the first increment due to the acknowledged difficulties in determining the exact border between litter layer and mineral soil in the forest soils. Soil carbon fractions were isolated according to the method of Zimmermann et al. (2007), modified by Poeplau et al. (2018). In brief, 30 g of the bulk soil was dispersed with an ultrasonic probe at 22 J and then wet-sieved with 2.2 L deionized water through a 63 μm sieve in order to separate the coarse POM and sand and aggregate fractions (S+A) from the fine mineral fraction. After sieving, the fine fraction was centrifuged and the supernatant fluid was filtered through a 0.45 μm filter and analysed for water-extractable carbon (here defined as dissolved organic carbon [DOC]). The remaining silt and clay fraction (S+C) was dried until weight constancy at 50°C and analysed for C and N content. From the S+C fraction, a 1 g aliquot was used to determine resistant soil organic carbon (rSOC) with a 6% sodium hypochlorite solution (NaOCl). In this step, the 1 g sample was stored in a 50 mL centrifuge tube that was filled to 45 ml with NaOCl. After shaking, the tubes were left open for 16 h to ensure optimal oxidation and prevent the tubes from bursting due to gas produced by the ongoing oxidation process. Afterwards, the tubes were centrifuged, decanted, washed twice with deionized water and refilled with NaOCl. After three repetitions, the washed sample was dried at 50°C until weight constancy and the remaining material was analysed for C and N content.

The POM and S+A fraction was dried until weight constancy at 50°C. This fraction was then mixed with a sodium polytungstate solution, which was adjusted to a density of 1.8 g/cm^3. After mixing and centrifuging, the POM floating on the sodium polytungstate was decanted, washed with deionized water and dried again until weight constancy at 50°C. These samples were subsequently milled and analysed for C and N content. The same washing procedure was applied for the heavy, sinking fraction, which was considered to be the S+A fraction consisting of sand and stable aggregates.

2.2.3 | Main soil parameters

The C and N content was measured with an elemental analyser (LECO TruMac CN). Dissolved organic carbon (one of the investigated fractions described below) was measured with a Dimatoc 2000 (Dimatec GmbH). To distinguish between total organic carbon (TOC) and total inorganic carbon, samples with pH_{H_2O} >6.2 were heated in a muffle furnace at 440°C prior to elemental analyses.

The pH was determined in accordance with ISO 10390 (2005): An aliquot of 10 g soil was used to measure pH in H_2O at a soil:water ratio of 1:5. The sample was shaken in a horizontal shaker for 1 h and then measured with a potentiometric pH meter.

Soil texture was determined for the samples from the second depth increment (10–20 cm) according to DIN ISO 11277:2002-08 (2002), which is based on a combination of sieving and sedimentation of suspended particles according to Köhn (1929). The second depth increment was chosen in order to ensure comparability of the soil texture data with the results of the fractionation and to avoid a potential influence of the measurement by forest litter on top of the first depth increment.

Furthermore, soil phosphorus (P) was extracted from all the mixed samples from the first depth increment (0–10 cm) with the Olsen-P method (Olsen et al., 1954) and analysed via inductively coupled plasma optical emission spectroscopy. The first depth increment was chosen in order to quantify the potential impacts of fertilizer application at the agricultural plots.

TABLE 1 General site parameters of the sampled sites: land use, latitude, longitude, elevation, mean annual precipitation, mean annual temperature, years since land-use change (LUC), texture and pH (H_2O)

Site	Land use	Latitude (°N)	Longitude (°E)	Elevation (m.a.s.l.)	Mean annual precipitation (mm)	Mean annual temperature (°C)	Cumulative degree days >0°C	Years since LUC (in 2019)	Texture: Sand/silt/clay (g/kg)	pH_{H_2O}
Sites with permafrost										
DR	Grassland	64.01164	−138.766494	427	354	−3.2	1887	34	Forest: 442/467/91	5.45
									Grassland: 453/464/84	6.93
DW	Grassland	60.784162	−137.534659	637	343	−0.7	1804	9	Forest: 68/642/290	7.94
									Grassland: 49/573/379	8.17
KK	Cropland	64.036097	−139.492003	347	363	−3.5	1870	119	Forest: 125/776/99	7.68
									Cropland: 233/684/83	7.75
KL	Cropland	64.064224	−139.067374	378	361	−3.4	1879	21	Forest: 418/491/91	6.04
									Cropland: 338/555/107	6.24
MA	Grassland	60.364461	−137.045507	680	433	−0.3	1700	60	Forest: 171/623/206	6.24
									Grassland: 145/646/209	6.90
MG	Cropland and grassland	60.952879	−135.143858	628	308	+0.1	1996	32	Forest: 289/535/176	8.53
									Cropland: 198/621/181	8.89
									Grassland: 166/606/228	8.24
PC	Cropland and grassland	62.828496	−137.323740	461	337	−3.0	1807	Both 118	Forest: 44/782/174	7.70
									Cropland: 414/484/72	8.31
									Grassland: 62/793/145	8.59
RC	Cropland	60.811823	−137.4105572	673	340	−0.5	1841	3	Forest: 69/546/385	7.52
									Cropland: 251/515/234	7.96
RF	Two croplands, different ages	60.855739	−135.209860	663	314	+0.2	1985	Old plot: 49 Young plot: 4	Forest: 431/513/56	7.15
									Old Cropland: 212/701/87	7.99
									Young Cropland: 364/574/62	7.71
SI	Cropland	64.092483	−139.458949	317	362	−3.5	1874	117	Forest: 43/816/141	8.07
									Cropland: 520/398/82	8.30
TH	Cropland and grassland	64.036959	−139.154052	361	361	−3.4	1879	Both 100	Forest: 557/346/97	6.81
									Cropland: 65/723/212	5.79
									Grassland: 320/573/107	7.19

(Continues)

TABLE 1 (Continued)

Site	Land use	Latitude (°N)	Longitude (°E)	Elevation (m.a.s.l.)	Mean annual precipitation (mm)	Mean annual temperature (°C)	Cumulative degree days >0°C	Years since LUC (in 2019)	Texture: Sand/silt/clay (g/kg)	pH_{H_2O}
Sites without permafrost										
CD	Cropland and grassland	60.860362	−135.592933	646	315	−0.1	1909	43	Forest: 406/369/225	8.19
									Cropland: 163/642/195	7.19
									Grassland: 174/598/228	8.77
EF	Grassland	60.833628	−135.438689	701	315	0.0	1949	26	Forest: 86/618/296	8.43
									Grassland: 81/702/217	8.44
EG	Grassland	60.859964	−135.411148	638	315	0.0	1949	23	Forest: 465/486/49	8.01
									Grassland: 263/645/92	8.32
GV	Cropland and grassland	61.009636	−1335.212049	741	307	−0.2	1938	Both 26	Forest: 89/832/79	7.41
									Cropland: 97/807/96	7.70
									Grassland: 88/826/86	7.84
LR	Cropland	61.061670	−135.178039	635	307	−0.2	1938	10	Forest: 102/301/597	7.53
									Cropland: 109/313/578	7.90
NB	Grassland, only topsoil	63.701581	−135.844904	599	367	−2.1	1978	38	Forest: 143/747/110	5.54
									Grassland: 105/728/167	6.59
VE	Cropland and grassland	64.062466	−139.016306	387	361	−3.4	1879	Cropland: 119	Forest: 473/418/109	6.22
								Grassland: 100	Cropland: 57/705/238	5.94
									Grassland: 100/672/228	6.11

Note: Mean annual precipitation, mean annual temperature and cumulative degree days were obtained from Climate Data Canada (2021).

2.2.4 | Calculation of SOC stocks

Cumulative SOC stocks of each soil core (0–80 cm) were calculated using Equation (1), with the TOC content (TOC_i [g/kg]), the dry mass of the fine soil ($Mass_i$ [g]), the volume of the soil core ($Volume_i$ [cm^3]) and the thickness ($Thickness_i$ [cm]) of every depth increment *i*.

$$\mathrm{SOC}\left[\mathrm{Mg\,ha^{-1}}\right] = \sum_{1}^{i}\left(\frac{\mathrm{TOC}_i}{1000} \times \left(\frac{\mathrm{Mass}_i}{\mathrm{Volume}_i} \times 100 \times \mathrm{Thickness}_i\right)\right) \quad (1)$$

Changes in bulk density after land-use change made it necessary to apply a mass correction, as discussed in various studies (Ellert & Bettany, 1995; Rovira et al., 2015; Wendt & Hauser, 2013). The individual cropland and grassland soil cores were mass corrected with the mean of the forest soil cores (mineral soil, without the litter layer), as described in Rovira et al. (2015). First, the cumulative mineral fine soil (MFS) of the soil cores was calculated as described in Equation (2) and then organic matter, as derived from the TOC and the van Bemmelens factor, was subtracted from the total fine soil (FS [$\mathrm{Mg\,ha^{-1}}$]).

$$\mathrm{MFS}\left[\mathrm{Mg\,ha^{-1}}\right] = \sum_{1}^{i} \mathrm{FS}_i \times \left(1 - \left(1.724 \times \mathrm{TOC}_i\right)\right) \quad (2)$$

The reference MFS for every site was then calculated as the mean MFS of the five forest soil cores. Afterwards, MFS of the cropland and grassland plots was adjusted to the reference MFS, and SOC stocks were calculated on the basis of a linear relationship between MFS and SOC stocks. Changes in SOC stocks were assessed by calculating the absolute difference (in $\mathrm{Mg\,ha^{-1}}$) and the relative difference (Equation 3) between agricultural land and forest, where SOC_{new} is the SOC stock of the cropland or grassland plot and SOC_{forest} is the SOC stock of the adjacent forest plot. Afterwards, litter C stocks were added to the SOC stocks of the mineral soil.

$$\Delta\,\mathrm{SOC}\,[\,\%\,] = \frac{\mathrm{SOC_{new}} - \mathrm{SOC_{forest}}}{\mathrm{SOC_{forest}}} \times 100 \quad (3)$$

In the exceptional cases of data gaps in bulk density, that is when not all five soil cores could be sampled entirely, the mean bulk density value of the remaining four soil cores where used for the missing soil sample. At the DW site, it was not possible to sample the deepest depth increment with the correct bulk density in all five soil cores. There, a pedo-transfer function based on the carbon content and bulk density of the overlying depth increment was used to estimate the bulk density of the deepest depth increment.

2.2.5 | Statistics

In order to identify patterns in the dataset and between the different variables, an analysis of the most important correlations in the dataset was performed. The dataset consisted of variables with different scale levels and non-linear relationships; therefore, the conditions (continuous scales in linear relationship) for a Pearson product correlation were not fulfilled and Spearman's rank correlation was used instead.

To calculate the influence of permafrost, time since land-use change and type of land-use change on SOC, the dataset was split into groups with type of change ('forest to cropland' or 'forest to grassland') and occurrence of permafrost ('yes' or 'no'). To test the hypothesis that SOC stocks of the plots with new land use are significantly different from SOC stocks of the corresponding forest plot, two linear mixed-effects models were fitted using SOC stock as the dependent variable, land use and permafrost (one model for the cropland/forest pairs and one model for the grassland/forest pairs) as fixed effects, and the specific sites as random effects. After checking the assumptions for linear mixed-effects models (homoscedasticity, normality of the residuals and linearity of the dataset), log transformation of the SOC stocks was necessary to meet all the criteria. After performing the linear mixed-effects models, analysis of variance and estimated marginal means with Tukey adjustment were used to obtain pairwise comparisons of all groups of the linear mixed-effects model (confidence level = 0.95).

The influence of time since land-use change on changes in SOC stocks was assessed using regression analysis. Based on the Akaike information criterion (AIC), a linear function or a second-degree polynomial was used. The regression was applied to cropland and grassland together, since the sample size of each group would have been too small for a meaningful regression. In the model, ΔSOC was the dependent variable, time since conversion from forest served as the explanatory variable and permafrost occurrence was used as the grouping variable.

To assess the influence of land-use change and permafrost occurrence on SOC fractions, the dataset was split into four groups (cropland and grassland sites with and without permafrost) and tested for significant differences between forest and the new land use. For relative shares of the fractions (all fractions normalized to 100%) as well as absolute values (g C per kg soil), a Wilcoxon rank sum test was used since the assumptions for parametric tests were not fulfilled and the subsets had only small sample sites.

All statistical analyses were conducted using R version 4.0.4 (R Core Team, 2021) with the packages dplyr (Wickham et al., 2020), emmeans (Lenth, 2021), ggplot2 (Wickham, 2016), ggpubr (Kassambra, 2020), ggthemes (Arnold, 2021), lme4 (Bates et al., 2015) and lmerTest (Kuznetsova et al., 2017). The level of significance for all statistical analyses was selected as $\alpha = .05$.

3 | RESULTS

3.1 | SOC stocks

Both cropland and grassland had significantly smaller SOC stocks than the adjacent forest when permafrost was present at 0–80 cm depth. When permafrost-affected forest soils were converted to croplands, the average losses were 15.6 ± 21.3% or 23.7 ± 42.2 $\mathrm{Mg\,ha^{-1}}$. When converted to grasslands, the average losses amounted to 23.0 ± 13.0% or 40.9 ± 23.0 Mg ha^{-1}. Sites without permafrost in the forest plots showed no statistically significant change in SOC stocks upon conversion, with −3.1 ± 11.3% or 2.4 ± 14.0 Mg ha^{-1} after conversion to cropland and 15.7 ± 27.7% or 15.0 ± 20.6 Mg ha^{-1} after conversion to grassland (Figure 3). Table 2 summarizes the SOC stocks of the four observed land-use change classes: Sites with permafrost had significantly higher mean SOC stocks in both forest and agricultural land than sites without permafrost. Overall, soils lost 30.6 ± 35.8 Mg ha^{-1} (18.6 ± 18.3%) SOC at sites with permafrost and gained 10.0 ± 18.5 Mg C ha^{-1} (8.2 ± 23.7%) at sites without permafrost after land-use change, with the latter not being statistically significant. Reductions in SOC stocks were mostly, but not exclusively, connected to reductions in the carbon content in the uppermost 30 cm of the mineral soil (Figure S2).

3.2 | Soil organic matter fractions

In order to assess the influence of land-use change and permafrost on SOC fractions, the dataset was split into four groups: (1) change to cropland at sites without permafrost, (2) change to cropland at sites with permafrost, (3) change to grassland at sites without permafrost and (4) change to grassland at sites with permafrost. Each group contained the SOC fractions of the forest samples and the samples of the new land use. Comparisons were made within each group between forest and new land use. The fractionation showed that land-use change had the strongest effects on POM and S+C. The proportional share of these fractions changed significantly with land-use change: at sites with permafrost, POM was significantly lower in cropland and grassland soils than in forest soils, while S+C increased significantly (Figure 4). However, except for permafrost-affected croplands, only fractions of the topsoil changed significantly. In absolute terms (Figure 5; Table S2), the topsoil of the agricultural plots (cropland and grassland) had more SOC stored in the S+C fractions than the forest, irrespective of permafrost occurrence. Topsoil POM of the permafrost-affected sites showed a strong dependency on the occurrence of permafrost in the forest, with large losses in both cropland and grassland. Due to high variability and relatively small sample sizes, statistically significant differences were only observed in grassland soils with and without permafrost and in cropland soils without permafrost. Overall, the share of rSOC and DOC was small in both absolute content and relative share of total SOC. Furthermore, due to inorganic carbon in some subsoils, it was not possible to determine reliable rSOC values at all sites, hence rSOC is not reported in the subsoil fractions.

3.3 | Temporal dynamic and additional drivers of SOC change

The temporal dynamic of differences in SOC stocks between forest and agricultural land was best described (i.e. $p < .05$ and best AIC) by a second-order polynomial fit, but only in the case of relative changes at

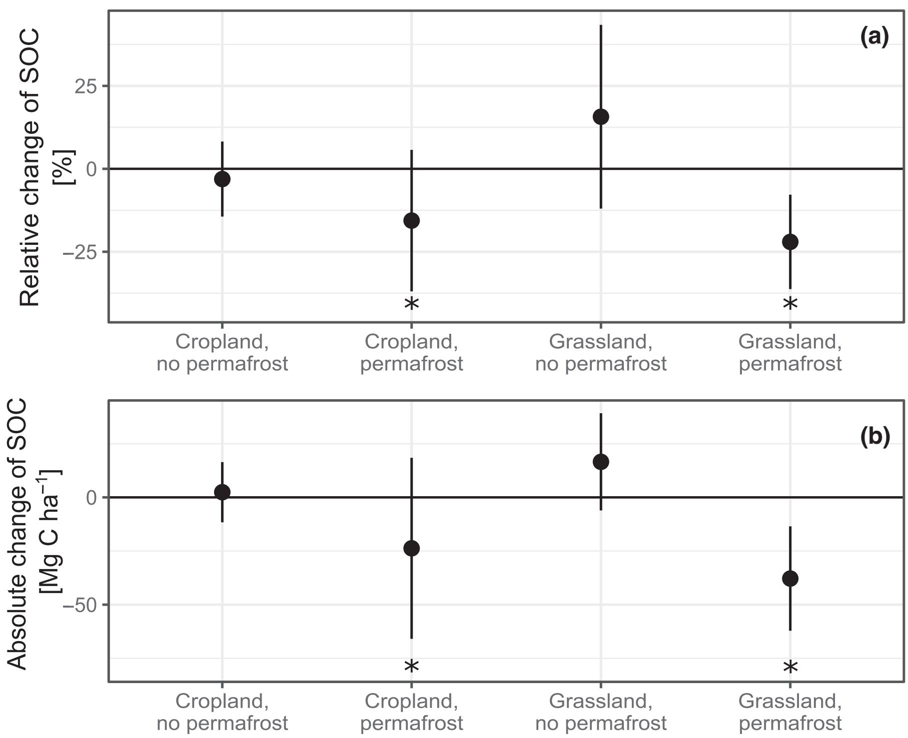


FIGURE 3 (a) Relative [%] and (b) absolute [Mg ha^{-1}] changes in soil organic carbon (SOC) stocks under cropland and grassland, compared with forest with and without permafrost. Points indicate the mean change and error bars indicate the standard deviation of the change in SOC stocks. * Indicates whether the SOC stocks of the given land-use class are significantly different ($p < .05$) from the corresponding forest plots (zero line).

TABLE 2 Summary of SOC stocks in forests (mineral and organic layers) and agricultural land under the different permafrost occurrences and land-use classes

Permafrost in forest	New land use	Forest mineral soil SOC stock (mg ha^{-1})	Forest litter C stock (mg ha^{-1})	Total SOC stock in forest (mg ha^{-1})	Total SOC stock in new land use (mg ha^{-1})	Farms sampled
No	Cropland	118.4 ± 79.6	13.3 ± 4.1	131.7 ± 80.9	134.1 ± 94.8	4
	Grassland	85.4 ± 70.8	13.6 ± 4.8	98.5 ± 70.2	113.5 ± 81.7	6
Yes	Cropland	144.1 ± 78.1	21.1 ± 8.0	163.8 ± 79.4	140.1 ± 78.6	9
	Grassland	171.9 ± 54.2	25.5 ± 8.6	194.2 ± 59.9	153.3 ± 67.9	6

Note: Values are reported as mean ± standard deviation.
Abbreviation: SOC, soil organic carbon.

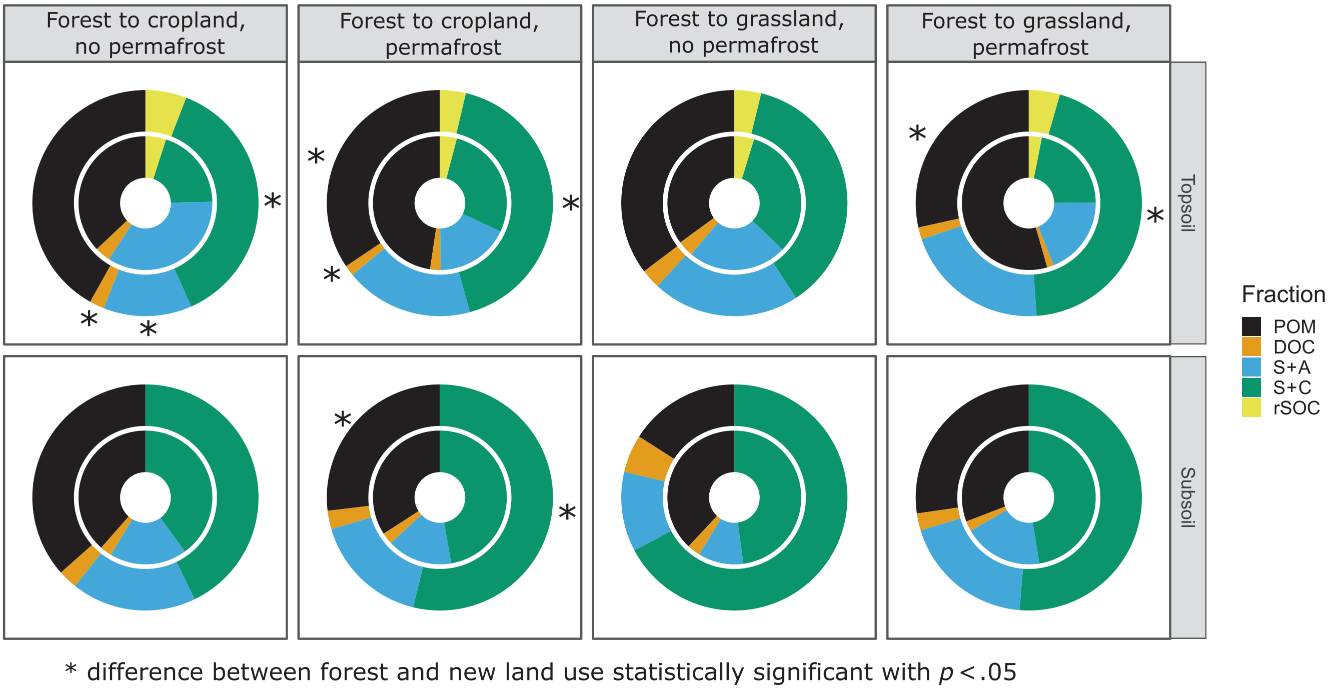


FIGURE 4 Relative share of particulate organic matter (POM), dissolved organic carbon (DOC), sand and stable aggregates (S + A), silt and clay (S + C) and recalcitrant soil organic carbon (rSOC) in forests (inner circle) and new land use (outer circle) at 10–20 cm (topsoil, upper row) and 40–60 cm (subsoil, lower row), differentiated by permafrost occurrence and land use (columns). Significant differences between forest and new land use are indicated by *, with $p < .05$.

sites with permafrost (Figure 6). For sites without permafrost, as well as for absolute changes, no significance was found in any regression, hence no regression line is displayed. The fit showed that there was a large loss of SOC shortly after land-use change and a tendency for subsequent SOC replenishment. At sites without permafrost, agricultural land tended to have larger SOC stocks than the forest after 100 years. Soil organic carbon losses at young sites without permafrost were small compared with sites with permafrost. In general, temporal trends were less clear than expected, which might be due to a relatively small sample size along the time axis and a lack of sites aged between 50 and 100 years.

Spearman's correlation coefficient (Figure 7) revealed distinct patterns in the dataset and delivered a broad overview of the most important site variables. Soil organic carbon stocks of the forest soils were significantly correlated with geographical variables (longitude, latitude mean annual precipitation, mean annual temperature, frost days and cumulative degree days). There was no correlation between forest SOC stocks and the general soil parameters (soil texture, pH and Olsen-P), except for C:N ratio. The content of the C fractions (Fraction forest) correlated significantly negatively with the C fractions of the agricultural land (ΔFraction). The depth of permafrost was significantly correlated with the thickness of the A-horizon in the forest as well as with the C-stock of the litter layer. Overall, the correlation matrix suggested that climatic drivers were more important for SOC than soil properties.

4 | DISCUSSION

4.1 | SOC stocks and permafrost

It was hypothesized that land-use change leads to larger SOC losses from permafrost-affected soils than from non-permafrost soils. This was strongly supported by the data obtained in this

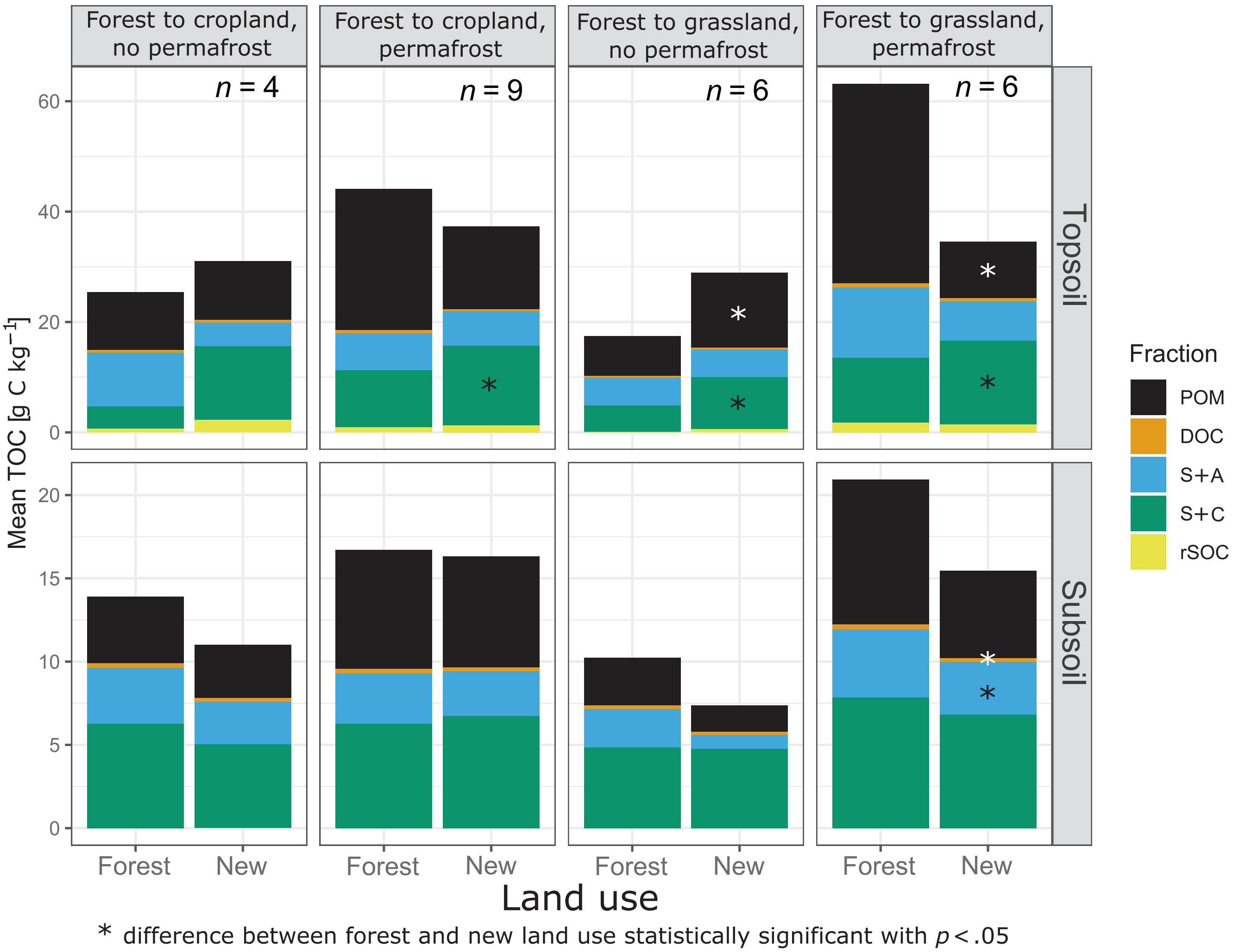


FIGURE 5 Comparison of mean total organic carbon (TOC) content of particulate organic matter (POM), dissolved organic carbon (DOC), sand and stable aggregates (S + A), silt and clay (S + C) and recalcitrant soil organic carbon (rSOC) at 10–20 cm (topsoil, upper row) and 40–60 cm (subsoil, lower row), differentiated by permafrost occurrence and land use (columns). Significant differences between forest and new land use are indicated by *, with $p < .05$.

study, as an average SOC loss of 30.6 ± 35.8 Mg ha^{-1} (18.6 ± 18.3%) was identified at sites with permafrost. For sites without permafrost, no statistically significant changes in SOC were detected. From the difference between sites with and without permafrost, it was concluded that abiotic site conditions were of major importance for the magnitude and direction of SOC dynamics after deforestation.

These results are in line with Grünzweig et al. (2015), who also found dramatic SOC losses (69%) after conversion of forests to agriculture on permafrost soils in Alaska, while forests without permafrost showed lower SOC losses upon conversion. Greater losses from sites with permafrost can be attributed to the rapid thaw of permafrost that previously protected SOM from decomposition (Grünzweig et al., 2015). When removing the insulating forest vegetation and litter layer, the microclimate of a site, and thus, also the soil temperature and moisture regime, is greatly changed (Shur & Jorgenson, 2007). In fact, during the sampling campaign, no signs of frost were detected in the 0–80 cm soil profiles in any of the agricultural fields sampled, while the permafrost-affected forests had an average active layer depth of 50 cm. The fact, that the conversion from permafrost-affected forests to grassland resulted in higher SOC losses than the conversion to cropland is most likely explicable by the differences in hydrology of the sites. Forest sites with the highest carbon stocks, thickest A-horizons and the shallowest permafrost were also the wettest and thus also those that were least suitable for cropping. Permafrost-affected forests that became grasslands had an average SOC stock of 194 ± 60 Mg C ha^{-1}, while permafrost-affected forest soils that were converted to croplands had an average 164 ± 79 Mg C ha^{-1} Soil hydrology plays a major role in permafrost regions, since permafrost acts as a barrier for infiltrating water (Klinge et al., 2021). In fact, individual farmers reported that after clear-cutting, soils are usually left to drain for some years before they can be cultivated. This has also been reported for soils in the Fairbanks Area of Alaska in the mid of the last century (Péwé, 1954). Furthermore, much of the SOC losses were observed to occur in the topsoil,

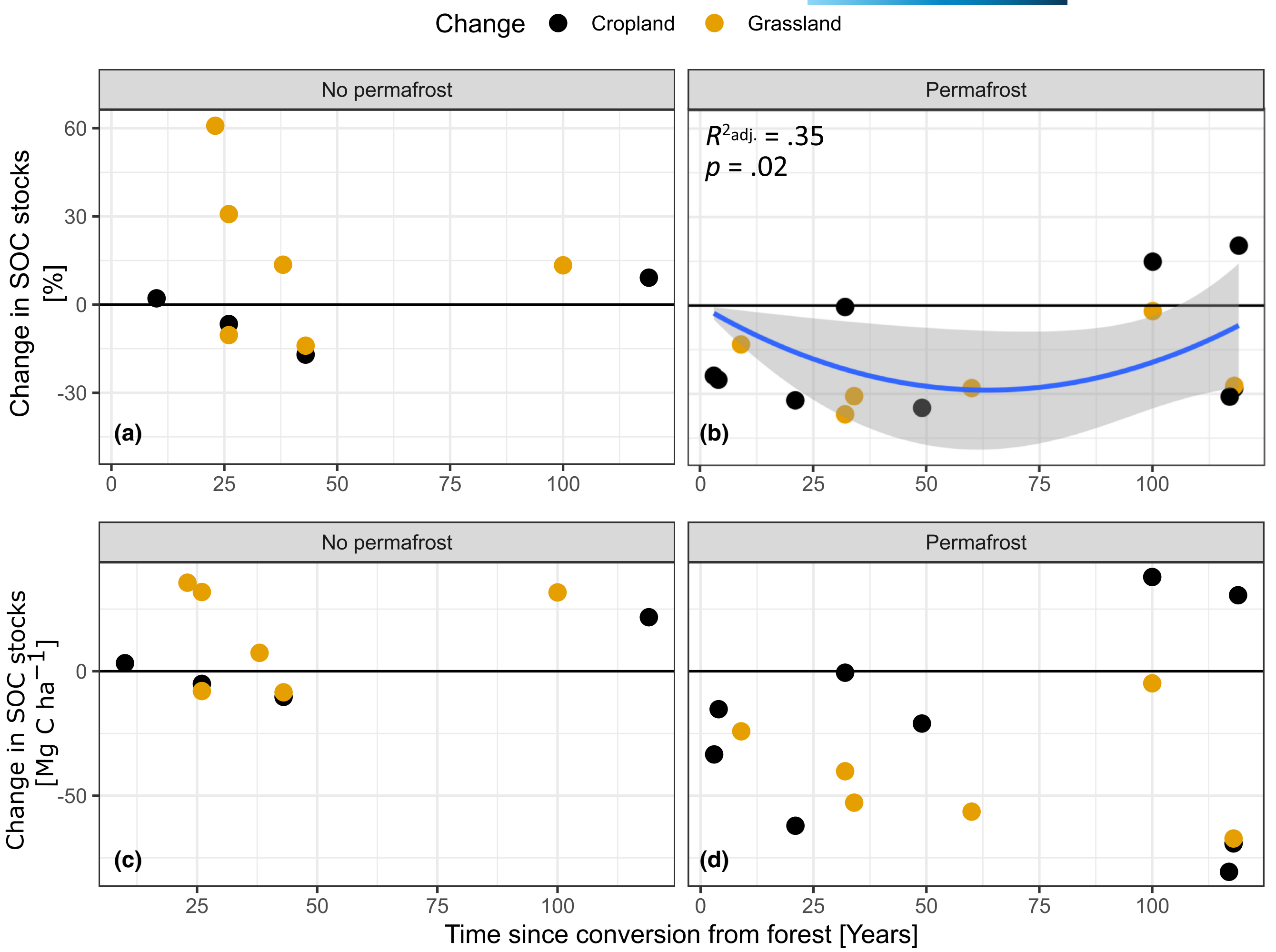
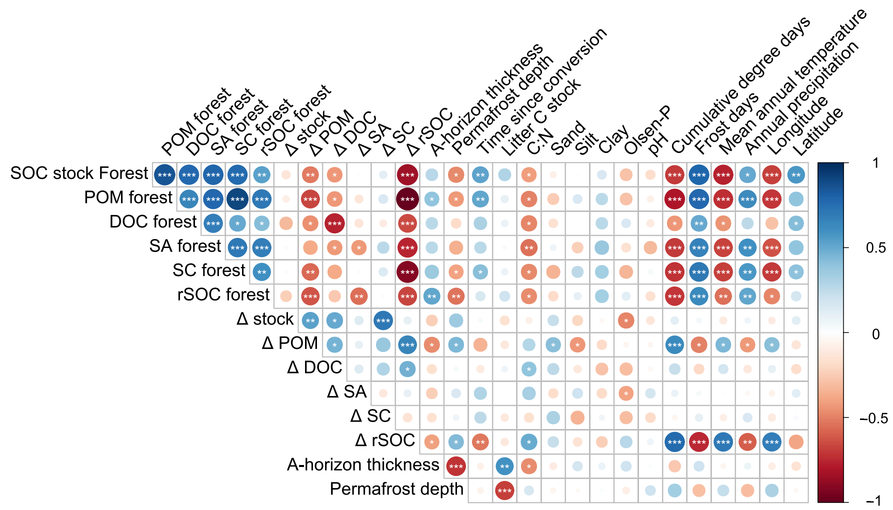


FIGURE 6 Time-dependent change in total soil organic carbon (SOC) stocks, with relative (a, b) and absolute changes (c, d) at sites without (a, c) and with (b, d) permafrost. The solid blue line indicates the only significant model fit and the grey area indicates the standard deviation of the fit.

FIGURE 7 Correlograms of the Spearman's correlation coefficient. Colours represent the correlation coefficient, statistical significance is indicated by asterisks, with $^*p<.05$, $^{**}p<.01$ and $^{***}p<.001$.



which is not permanently frozen (Figure S2). This might additionally indicate that it is indeed largely a drainage effect that causes SOC losses after conversion to agricultural land and the associated deepening of the active layer. At the same time, irrigation can also cause permafrost thaw as infiltrating water in summer may lead to an amplified heat transport from the ground surface to the permafrost (Lopez et al., 2010). Conversely, SOM in permafrost-free soils is more vulnerable to decomposition than SOM in permafrost soils, and permafrost-free soils are therefore generally lower in SOC (Zimov et al., 2006). When these soils are irrigated and fertilized, plant growth is encouraged, leading to increased biomass production and C inputs to the soil and thus potentially also higher SOC stocks after conversion (Grünzweig et al., 2004). It should however be noted that carbon stocks in agricultural soils are also fed by external carbon inputs (Table S1), which should have partly compensated or even overcompensated losses after deforestation. Most farmers applied compost, animal manure and similar organic fertilizers to their fields, which were partly derived from external sources such as imported animal fodder. Unfortunately, even after a detailed farmers questionnaire, the data were not good enough to make reliable estimates on the contribution of external organic fertilizers on the overall SOC stock difference between forest and agricultural soils. Maillard and Angers (2014) reported in a global meta-analysis that around 12 ± 4% of manure-C is retained as SOC, when applied regularly. As a very rough estimate, we calculated the potential manure effect of 100 chickens as a realistic, yet high, number for Yukon farms: Fresh chicken manure contains around 20% C (Singh et al., 2018) and a chicken produces around 25 kg manure per year (Tanczuk et al., 2019). This would result in 2.5 Mg manure farm^{-1} year^{-1} and 0.5 Mg manure-C farm^{-1} year^{-1}. If 12% of this C is retained in the soil, 0.06 Mg C would be added to the farm's soil. This is a negligible amount of carbon when compared to the land-use change effects of up to −80.6 Mg C ha^{-1} (site 'SI'). We thus assume that fertilization practices did not add a significant bias to the comparison of permafrost and non-permafrost-affected soils, even if organic fertilization would have fully relied on external sources. Moreover, SOC gains after deforestation should also not be interpreted as net carbon sinks, since losses from forest biomass are not accounted for in this study.

At sites without permafrost, no statistically significant difference in SOC change was found between the conversion to cropland and grassland, although grasslands tended to gain more SOC at sites without permafrost. The pattern of grasslands gaining SOC, which was found at sites without permafrost, was also observed by Deng et al. (2016) in a global meta-analysis. Conversion of forest to grassland increased SOC stocks by 11.53 Mg ha^{-1}, while conversion of forest to cropland decreased SOC stocks. In a meta-analysis focusing on Canadian soils without permafrost, VandenBygaart et al. (2003) also found large losses (24%) in SOC when native land was converted to agricultural land. Moreover, VandenBygaart et al. (2010) compared various long-term cropping experiments across Canada and found a significant increase in SOC when annual cropland was converted into perennial grassland, which is in line with the trend observed in the present study of increasing SOC under grassland at sites without permafrost. VandenBygaart et al. (2003) not only identified management and land-use change as important drivers of SOC dynamics, but also emphasized the interactive effects of climate variables and management. In dry conditions in western Canada, SOC storage could be increased by switching from conventional agriculture to conservation agriculture, but that was not found to be the case in the more humid eastern part of Canada. This indicates that climatic conditions need to be considered in order to predict the magnitude and direction of SOC change after alterations to land use or management. In the present study, management of the agricultural land at sites with and without permafrost was similar, so changes in microclimate may have played a major role in the magnitude of the changes, supported strongly by the correlation between SOC stocks and climate variables.

4.2 | Long-term trends in SOC changes

Differences between agricultural land and associated forest were compared on the basis of time since conversion. Clear evidence was found that sites with permafrost lost large amounts of SOC in the first years after clearing. At sites with permafrost, old farms (>50 years) had smaller losses than young farms or even more SOC in cropland than in the forest. This result has to be interpreted with care as, despite being statistically significant, this trend of SOC accumulation was driven by two cropland sites (KK and TH). These sites had a very high small-scale variability in the SOC content (Figure S1), which might not be explained by land-use change but by their location on the banks of the Yukon and Klondike rivers. The differences in soil texture (Table 1) between forest and cropland, especially at TH, support this interpretation. As visualized by the large confidence interval (Figure 6b), it is thus highly uncertain whether initial SOC losses from former permafrost soils can be offset by appropriate agricultural management. However, the long-term replenishment of SOC, that is smaller losses at old farms at sites with permafrost, fits well with the observations of Grünzweig et al. (2004) who explained this pattern of short-term loss and long-term gain by quick decomposition of rather labile forest-derived organic matter after clearing, followed by slower breakdown of more stable organic matter and simultaneously subsequent higher input of fresh crop-derived organic matter. At some of the sites without permafrost, a short-term increase in SOC was found, which may be explained by the incorporation of forest litter into the mineral soil when the forest was cleared (Table S1, sites EF, EG, LR and NB). This effect has also been reported in other studies (Dean et al., 2017; Grünzweig et al., 2004; Karhu et al., 2011).

It was also hypothesized that forest soils with initially low SOC stocks accumulate SOC over decades when turned into agricultural land, leading to the same level or even higher SOC stocks in agricultural land compared with forest. However, at sites without permafrost, no statistically significant evidence for a time dependency of SOC changes was found, hence this hypothesis must be rejected. At six sites, equal or increased SOC stocks were found in the agricultural land compared with forest, but these sites covered a wide range of SOC stocks (Figure S1) and were not limited to those sites low in forest SOC. Moreover, it remains unclear how SOC at young farms will develop in the next few decades.

At sites with permafrost, it was possible to sample farms of various ages, well distributed over time back to the days of the gold rush. However, farms that were 43–100 years old without permafrost could not be sampled, which may explain the absence of statistical evidence for this hypothesis. Furthermore, observations of around 100 years may be considered long term for a study about the effect of agriculture on SOC, but could still be short term in relation to organic matter turnover in subarctic regions. Even though the IPCC assumes 20 years of linear carbon sequestration for ecosystems to reach a new equilibrium after land-use change (IPCC, 2003), other authors have estimated different times to reach equilibrium. Depending on the type of land-use change, Poeplau et al. (2011) used exponential, polynomial and linear functions to calculate the time to reach a new equilibrium after land-use change. According to Poeplau et al. (2011), it takes between 23 years (land-use change from forest to cropland) and over 200 years (land-use change from grassland to forest) to reach a new steady state in temperate soils. Moreover, Karhu et al. (2011) also concluded that longer time spans than the IPCC default value are necessary to observe land-use change effects on SOC in the boreal region.

4.3 | Changes in SOC fractions

It was hypothesized that the remaining SOC under agricultural land consists of relatively more carbon of the fine mineral fraction than under native forest, as POM is partly removed by clearing and quickly decomposed after the start of the new land use (Chen et al., 2019; Grünzweig et al., 2015; Poeplau & Don, 2013). Significant changes in SOC fractions were found, mostly at sites with permafrost. At these sites, POM-C was significantly reduced, while the S+C fraction had more C in agricultural land than in the forest. Therefore, this hypothesis was well supported for sites with permafrost. The S+A fraction, as an intermediately labile fraction (Zimmermann et al., 2007), was reduced in agricultural land independently of permafrost, again supporting the hypothesis of this study. Despite all statistical evidence, results from SOC fractionation should be considered as an indicative characterization of the soil organic matter composition and not as mass balance.

The POM fraction was observed to increase after land-use change to grassland at sites without permafrost and to decrease at sites with permafrost, underlining the sensitivity of the POM fraction to permafrost thaw. The observed decrease in POM in the topsoil after permafrost thaw was also in line with the results of Mueller et al. (2015). In permafrost soils in Alaska, Mueller et al. (2015) reported that most SOC was stored within the active layer and that the largest part (around 73%) of SOC within the uppermost metre consisted of POM, which is mostly composed of easily degradable carbohydrates. Since the active layer depth at the sites with permafrost in the present study decreased rapidly after land-use change, the labile POM fraction was exposed to decomposition, leading to large losses of SOC. Losses of POM-C after deforestation have also been observed in other studies (Balesdent et al., 1998; Del Galdo et al., 2003; Karhu et al., 2011) and are in line with the results of the present study. The observed accumulation of mineral-associated carbon, here represented by the S+C fraction, pointed to greater stabilization of SOC during agricultural land use. This could potentially suggest a shift from plant-derived compounds to more microbial-derived ones (Angst et al., 2021), which are acknowledged to be a dominant fraction of mineral-associated organic matter (Buckeridge et al., 2020; Ludwig et al., 2015). Indeed, Schroeder et al. (in prep.) found higher microbial carbon-use efficiencies (i.e. more biomass production per total carbon uptake) in croplands and grasslands than in forest soils (the same soils as used in the present study), which might indicate an increased importance of this in vivo pathway of SOM stabilization (Liang et al., 2017; Sokol et al., 2019).

4.4 | Implications for future subarctic agriculture

The shift of agricultural regions will inevitably lead to the conversion of subarctic forest to agricultural land. Climate change and a growing demand for locally produced food in the north are encouraging the establishment of food production systems even in largely untouched areas, putting pressure on vulnerable subarctic forests. As shown in this study, subarctic agriculture can have strong negative impacts on SOC, potentially accelerating the positive feedback loop of warming and loss of SOC (Heimann & Reichstein, 2008). However, the present study also highlighted that the impacts of agriculture on SOC can be minimized or even offset when the potential new agricultural land is limited to areas without permafrost and if short-term losses of SOC can be minimized. However, the focus of this study was on SOC and did not quantify losses of other ecosystem C pools due to land-use change. In a synthesis, Kurz et al. (2013) calculated that boreal forests of Canada store around 40 Mg C ha^{-1} (21% of ecosystem C) in aboveground biomass, 10 Mg C ha^{-1} (6% of ecosystem C) in belowground biomass, that is living roots, and around 20 Mg C ha^{-1} (11% of ecosystem C) as deadwood, which is not included in the litter layer. Considering that these amounts of C (38% of ecosystem C) are certainly removed by deforestation, the small gains in SOC at sites without permafrost are negligible. This study revealed that, in terms of SOC storage, the abundance of permafrost within the upper first metre of forest soil should be carefully considered when establishing agriculture in pristine subarctic ecosystems. Furthermore, the focus of this study was on permafrost at a depth that is relevant for agriculture, but no account was taken of deeper permafrost. Therefore, sites that were classified as having no permafrost may have permanently frozen layers that could thaw due to land-use change. Short-term losses can potentially be avoided by adapted management and deforestation techniques that conserve as much of the litter layer as possible. Sustainable land management has been studied extensively in pedo-climatic regions with a distinct agricultural sector, but is understudied in the cold and dry subarctic where agriculture currently plays a minor role. In temperate climates, no till is acknowledged to have no significant effect on whole-profile SOC stocks (Chenu et al., 2019), but there is evidence that cropland soils in cold and dry regions may benefit from no till (VandenBygaart et al., 2010). In this study, no direct effect of tillage depth on SOC was identified, but since most farmers till only irregularly, the database may be insufficient for any evidence of tillage or other management effects. Promising concepts for more sustainable clearing, such as chipping and spreading or mulching and subsoiling of deforestation residues (roots, stumps, deadwood and litter layer) instead of burning or the introduction of silvopasture (Lim et al., 2018) or agroforestry (Tsuji et al., 2019) instead of clear-cutting, may help to reduce the negative impacts of agriculture on subarctic SOC.

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CONFLICT OF INTEREST

All authors have no conflict of interest.

DATA AVAILABILITY STATEMENT

All data and the R-script used for this study are freely available at 10.5281/zenodo.6460208.

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