https://doi.org/10.5194/egusphere-2026-210
Preprint. Discussion started: 26 January 2026

Precipitation reduction overrides edaphic controls on nitrous oxide emissions along a soil carbon, texture and pH gradient in a cereal field

Sigrid Trier Kjær[1], Peter Dörsch[1]

[1]Faculty of Environmental Sciences and Natural Resource Management, Norwegian University of Life Sciences, NMBU, Ås, 1433, Norway

Correspondence to: Sigrid Trier Kjær (sigrid.trier.kjar@nmbu.no)

Abstract. Nitrous oxide (N_2O) is a potent soil-borne greenhouse gas (GHG) which increases in the atmosphere due to the widespread use of synthetic nitrogen (N) fertilisers. Soil N_2O emissions are intrinsically controlled by soil moisture and edaphic properties such as soil organic carbon (SOC) content, texture and pH. With a future climate projected to increase frequency and severity of droughts in northern Europe, understanding how these factors interact to affect N_2O emissions is critical for predicting climate feedbacks. In this study, we investigated N_2O emissions along a hillslope gradient in an agricultural field in southeast Norway, characterised by increasing SOC and clay content and decreasing pH from top to bottom. Eight rainout shelters were installed along the hillslope, nominally reducing precipitation by 49%. N_2O emissions were measured weekly using static chambers over two years during the snow-free period. In the first year, N_2O emission measurements started two months after fertilisation and covered a four-month period, which included episodes of heavy rainfall; during this time, we found no effect of precipitation reduction or edaphic factors on emission rates. In the second year, reduced precipitation significantly decreased N_2O emissions (~25%). Under ambient precipitation, cumulative N_2O emissions were positively correlated with SOC content and showed weak negative and positive trends with soil pH and clay content, respectively. No significant correlations were found in plots with reduced precipitation. Altogether, our findings illustrate that soil physicochemical controls on N_2O emissions depend on the interaction between soil properties and climate. This has consequences for parameterising process-based N_2O models driven by soil properties and weather and calls for more in-depth studies on interdependencies of edaphic and climatic drivers of N_2O emissions.

1 Introduction

Nitrous oxide (N_2O) dominates the greenhouse gas (GHG) footprint of crop production and emissions have increased significantly since the industrialisation of agriculture. N_2O is a potent and long-lived GHG with a global warming potential 273 times that of carbon dioxide (CO_2) over a 100-year timescale (IPCC, 2023). Moreover, after successfully banning chlorofluorocarbons, N_2O has become the most important ozone-depleting substance in the stratosphere (Ravishankara et al.,

https://doi.org/10.5194/egusphere-2026-210
Preprint. Discussion started: 26 January 2026

2009). With the ongoing intensification of global food production based on synthetic fertiliser use, global N_2O emissions continue to grow, and the atmospheric mixing ratio of N_2O has risen by more than 20% from 1750 to 2018 (Tian et al., 2020).

While N_2O contributes to climate change, its production and consumption in soils are strongly driven by soil moisture and temperature. According to climate predictions for Norway and much of Europe, the future climate is expected to experience more frequent and severe summer droughts, along with increased and more extreme precipitation (Wong et al., 2011; Hanssen-Bauer et al., 2017; Zhu and Siebert, 2024). Soil moisture and oxygen (O_2) levels strongly control N_2O turnover in the soil (Firestone et al., 1979). For instance, nitrification requires O_2 to oxidise ammonium (NH_4^+) to nitrate (NO_3^-) and becomes more active at low soil moistures (WFPS < 60%) when O_2 diffuses readily into the soil. Conversely, high soil moistures (WFPS > 60%) support anoxic conditions, inducing denitrification which reduces NO_3^- to dinitrogen (N_2) with N_2O as an intermediate (Firestone, 1982; Davidson, 1993).

Fluctuations in soil moisture can trigger hot moments of increased N_2O emissions, especially when dry soils are rewetted (Butterbach-Bahl et al., 2013; Barrat et al., 2021). Although drying-rewetting cycles in soils experiencing drought or reduced precipitation can result in transiently large N_2O emissions, seasonal N_2O emissions are likely smaller than in wet soils receiving more precipitation (Borken and Matzner, 2009). In general, reduced precipitation tends to decrease N_2O emissions (Hartmann and Niklaus, 2012; Homyak et al., 2017; Li et al., 2020), the extent of which depends on the magnitude and duration of precipitation change. Meta-analyses of precipitation manipulation experiments revealed that reduced precipitation decreases N_2O emissions, although with variable effects, ranging from 7.1% (Yang et al., 2022) to 31% (Li et al., 2020) and 38.5% (Wu et al., 2022) compared to controls with ambient precipitation.

N_2O production and consumption are also influenced by soil physicochemical properties. Soil organic matter (SOM) is an important factor supporting heterotrophic microbial activity, N mineralisation and O_2 consumption in the soil matrix (Li et al., 2005; Jäger et al., 2011; Rummel et al., 2020). SOM also increases the soil's water holding capacity with consequences for soil redox conditions (Nemes et al., 2005). Once soils become anoxic, soil organic carbon (SOC) and its degradation products fuel anoxic respiration with N oxyanions (NO_3^-, NO_2^-) as terminal electron acceptors (i.e. denitrification). Furthermore, the decomposition of labile organic carbon (C) can create anoxic zones, promoting hotspots for denitrification (Schlüter et al., 2025).

Another important soil physicochemical property is soil texture which affects soil moisture and SOC content (Kaiser et al., 1996; Keiluweit et al., 2018; Li et al., 2024; Kjær et al., 2026a). Fine-textured soils have higher water retention capacity compared to coarse-textured soils and are more prone to developing anoxia due to slower diffusion of O_2 from the atmosphere (Pihlatie et al., 2004). At the same time, soil-borne N_2O lingers longer in the soil, increasing the opportunity for reduction to N_2. Additionally, SOC is stabilised by clay, resulting in generally greater C contents in clayey soils (Six et al., 2002a) due to physical protection of SOC from decomposition which, however, is susceptible to release by drying-rewetting and may fuel N_2O production (Harrison-Kirk et al., 2013). Clay content covaries with other soil chemical properties which potentially influence N_2O turnover; for example, clay particles have a higher cation exchange capacity compared to sand,

https://doi.org/10.5194/egusphere-2026-210
Preprint. Discussion started: 26 January 2026

allowing NH_4^+ to bind more effectively and making NH_4^+ less available for nitrification, thus reducing nitrification derived N_2O emissions and NO_3^- supply to denitrification (Yu et al., 2019). Therefore, clay may have adverse effects on N_2O production and consumption.

Another important edaphic parameter for net N_2O production in soils is pH. It is widely recognised that pH affects the final reduction step of denitrification, with lower pH (typically pH 5-8 in agricultural soils) increasing N_2O production relative to N_2 production (Stevens et al., 1998; Bakken et al., 2012). This is because acidic conditions impair the functioning of N_2O reductase (*NosZ*), which converts N_2O to N_2, leading to an increase of the N_2O product ratio of denitrification (Bakken et al., 2012).

Understanding the interactions between soil physicochemical properties and climatic conditions is critical for the parameterisation of process models predicting N_2O emissions and their responses to future environmental changes (Ge et al., 2024). While previous studies have explored the effects of reduced precipitation on N_2O emissions, there remain gaps in our understanding of how these effects interact with soil properties such as SOC, clay content and pH in cultivated soils.

The aim of this study was to investigate how precipitation regime interferes with the variability of SOC, clay content and pH common in cultivated upland soils in controlling N_2O emissions. We measured N_2O emission in paired plots under rainout shelters, that nominally reduced precipitation by 49%, and in control plots with ambient precipitation. The plots were cultivated identically and located along a natural gradient of SOC, clay and pH. We measured N_2O fluxes weekly throughout two growing seasons using manual chambers.

2 Materials and Methods

2.1 Field site and rainout shelters

The field experiment was conducted along a 62 m-long gentle (1.8%) hillslope on the experimental farm “Kjerringjordet” at the Norwegian University of Life Sciences in Ås, Norway (59°39'44.8"N, 10°45'48.3"E). The hillslope features a natural gradient with increasing SOC and clay and decreasing pH from top to bottom, and the soil type changes from loam to clay loam downslope. The slope has a north-east to south-west orientation with north-west-facing aspect. To the south, there is a treeline that shades the plots at the bottom of the slope for most of the day during the winter months.

Eight 2.5 × 2.5 m rainout shelters were installed along the hillslope (Fig. S1). The rainout shelters were custom-made as described in Kundel et al. (2018). Each shelter had nine 90° V-profiles, which were 2.5 m long and 96 mm wide. With nine V-profiles, 49% of the precipitation should be excluded from the plot. We installed a rain gutter on each shelter to ensure that the precipitation was diverted away from the plot. Uphill from each rainout shelter, a control plot with ambient precipitation was established resulting in a total of 16 plots. Soil volumetric water content (VWC) and temperature were measured at a depth of 5 cm in 13 of the 16 plots using time domain reflectometry (TDR, Decagon Em50). Based on the individual bulk density measured in each plot (n = 3) and a standard particle density of 2.65 g cm^{-3}, plot-wise water filled

https://doi.org/10.5194/egusphere-2026-210
Preprint. Discussion started: 26 January 2026

pore space (WFPS) was calculated. Precipitation data were obtained from a nearby weather station at NMBU, Ås (59°39'37.8"N, 10°46'54.5"E) (Wolff and Grimenes, 2024; Wolff, 2025).

The experiment was conducted from June 2023 until November 2024 (Table 1). The field was managed using conventional agricultural practices. It was harrowed in spring, sown to barley (*Hordeum vulgare*, cultivar Salome) with a seeding rate of 210 kg ha^{-1} and fertilised with 121 kg N ha^{-1} in 2023 and 132 kg N ha^{-1} in 2024 (YaraMila, NPK 22-3-10). Biomass samples for determining yield were collected by hand. In each plot, rows of barley were cut to a length of one metre under the rainout shelters and in the control plots. The distance between rows was 12.5 cm and yields were upscaled to the hectare. Rainout shelters were removed from the field just before the first snowfall and moved back again when snow was thawed (Table 1).

Table 1: Timeline of field activities during the experimental period for the first year (2023) and the second year (2024). Gas measurements continued beyond the removal of rainout shelters in the second year

	Sowing and fertilisation	Harvest	Rainout shelters	Gas measurements
Year 1: 2023	-	12.9	26.6 to 26.10	28.6 to 24.10
Year 2: 2024	7.5	28.8	21.3 to 14.11	2.3 to 26.11

2.2 Field N_2O measurements

N_2O emission rates were measured weekly from June to October 2023 and March to November 2024 by manual static chambers (Table 1). Frames with a water lock were installed in the middle of the rainout shelters and in a similar position in the control plots. N_2O emission rates were estimated from four samples drawn by syringe from the chambers for 30-60 minutes chamber deployment and transferred to pre-evacuated 10 ml glass vials with crimp sealed butyl rubber septa. Measurements were carried out between 11 a.m. and 2 p.m. Concentrations of N_2O, CO_2 and CH_4 in the vials were determined by a multi-column, multi-detector gas chromatograph (GC; Agilent 7890A) with two columns (PoraPLOT Q column and Molesieve column) and three detectors (TCD, FID and ECD) as described by Kjær et al. (2026b). Standard mixtures (AGA, Norway) were measured alongside the vials for calibration. CO_2 was used to quality-check N_2O measurements. Emission rates were fitted to a linear regression to estimate the emission rate from the change in concentration, and fluxes were cumulated using linear interpolation with the R package *DescTools* (version 0.99.54; Signorell and Ekstrom, 2024).

2.3 Ancillary variables

Two soil samples (0-20 cm) were taken monthly from each plot (during the snow-free period) and pooled, before extracting NO_3^- and NH_4^+ in a 1M KCl solution, followed by filtration and spectrophotometry (Doane and Horwáth, 2003; Krom,

https://doi.org/10.5194/egusphere-2026-210
Preprint. Discussion started: 26 January 2026

1980). Gravimetric soil moisture was determined in the same samples by drying. An extra set of soil samples for determining soil properties were collected on 24 October 2023. In each plot, five samples were taken from a depth of 0-20 cm and mixed. Each sample was analysed for total C and N, soil texture and pH (Table 2). Total C and N contents and stable isotope ratios ($\delta^{13}C$ and $\delta^{15}N$) were measured by an element analyser (FlashEA 1112 HT, Thermo Fisher Scientific) coupled to an isotope ratio mass spectrometer (Delta^{plus}XP, Thermo Finnigan). Soil texture was determined using the pipette method (Krogstad et al., 1991; Elonen, 1971) and pH was measured in water in autumn 2023 and spring 2024. pH was remeasured in 2024 because an adjacent field was limed in winter 2023/2024 and, as the snow thawed, some lime was transported close to our field. The pH increased by 0.18 ± 0.12 from 2023 to 2024 (Table 2). Volumetric samples for determining porosity and bulk density were taken in triplicate steel cylinders (100 cm^{-3}) at a depth of 10-15 cm in each plot at the end of the experiment (spring 2025). Field capacity was defined as the water content at −33 kPa. Volumetric water content (VWC) at field capacity was calculated from the mass difference between soil cores equilibrated at 33 kPa and oven-dried samples and normalised by the cylinder volume (n = 3).

https://doi.org/10.5194/egusphere-2026-210
Preprint. Discussion started: 26 January 2026

Table 2: Soil properties for control and rainout shelter plots along the hillslope (m from hilltop). Plot names are A to H and * indicates plots that were equipped with a TDR sensor. Given are soil organic carbon (SOC, mg g dry weight^{-1}), C:N ratio, clay (%), silt (%), sand (%), pH (measured in both 2023 and 2024), bulk density (g cm^{-3}) and volumetric water content at field capacity (VWC at FC, %). Samples were taken from a depth of 0-20 cm for all variables except bulk density and VWC at field capacity which were determined at a depth of 10-15 cm.

Metres from hilltop	Plot	Treatment	SOC[a] mg g^{-1}	C:N[a] ratio	Clay[a] %	Silt[a] %	Sand[a] %	pH[a] 2023	pH 2024	Bulk density g cm^{-3}	VWC at FC %
0	A *	Control	28.3	11.2	24	39	37	5.92	6.15	1.16 ± 0.03	32 ± 0.6
2	*	Rainout shelter	28.3	10.8	24	42	34	5.93	6.11	1.24 ± 0.03	33.4 ± 0.3
8	B *	Control	27.3	10.5	25	38	37	6.11	6.10	1.28 ± 0.02	33.4 ± 0.6
10		Rainout shelter	28.1	10.4	25	37	38	6.11	6.13	1.18 ± 0.05	32.1 ± 1.8
16	C *	Control	29.9	10.7	28	36	36	5.96	5.99	1.22 ± 0.03	31.5 ± 0.4
18	*	Rainout shelter	29.6	10.8	24	43	33	5.83	5.96	1.18 ± 0.04	30.6 ± 0.8
24	D	Control	28.2	10.7	24	39	36	5.89	6.10	1.17 ± 0.02	30.5 ± 0.4
26		Rainout shelter	29.1	11.0	28	36	36	5.86	6.14	1.24 ± 0.02	32.7 ± 0.6
32	E *	Control	29.5	10.9	24	41	34	5.90	6.05	1.22 ± 0.01	32.8 ± 0.5
34	*	Rainout shelter	28.8	11.1	28	38	34	5.92	6.11	1.23 ± 0.02	30.9 ± 0.1
44	F *	Control	30.5	10.9	26	38	36	5.86	5.98	1.24 ± 0.05	32.6 ± 0.8
46	*	Rainout shelter	31.7	11.2	29	35	36	5.83	5.91	1.26 ± 0.04	32.7 ± 0.6
52	G *	Control	36.1	11.2	30	36	33	5.44	5.88	1.19 ± 0.02	33.4 ± 0.6
54	*	Rainout shelter	37.2	11.3	32	35	33	5.64	5.86	1.13 ± 0.02	32.6 ± 1.1
60	H *	Control	38.1	11.3	32	36	32	5.55	5.92	1.15 ± 0.01	34.4 ± 0.3
62	*	Rainout shelter	37.7	11.4	34	34	32	5.62	5.90	1.12 ± 0.04	33.4 ± 1.3

[a] from Kjær et al. (2026a)

https://doi.org/10.5194/egusphere-2026-210
Preprint. Discussion started: 26 January 2026

2.4 Statistics

A linear mixed-effects model was used to compare N_2O emission rates and mineral N (NO_3^- and NH_4^+) contents between rainout shelter and control plots. We used the lmer and lmerTest packages with the plots along the gradient as pseudo-replicates to test the difference in precipitation treatment, accounting for repeated measurements on the same plots. Differences in cumulative N_2O emissions and dry matter yield (DMY) between treatments were tested using a non-parametric Wilcoxon rank-sum test and unpaired t-tests, respectively. These tests were chosen because cumulative N_2O emissions were not normally distributed, while yield data were approximately normal (Shapiro-Wilk tests). Levene's tests confirmed similar variances across treatments in both datasets. Relationships between cumulative N_2O emissions and soil properties (SOC, pH, clay content and bulk density) were analysed using linear regression, with regression coefficients, R^2 values and p-values reported. Statistical significance was set to $p < 0.05$. All statistical analyses were performed using R version 4.5.1 (R Core Team, 2025).

3 Results

3.1 Impact of rainout shelters on soil moisture distribution

Water-filled pore space (WFPS) values, calculated from plot-specific bulk densities and continuously logged volumetric water contents from periods with rainout shelters and gas measurements (excluding the winters), were higher in the control plots than under the rainout shelters. This confirms that the shelters effectively reduced soil moisture (Fig. 1). The frequency distribution of WFPS in the rainout shelter plots was bimodal, with two maxima at 35–50% and 65–75% WFPS (Fig. 1A). By contrast, the distribution of WFPS in the control plots was unimodal and left-skewed with a maximum at 65–85% WFPS. When plotting WFPS frequency distribution for soil moistures measured during flux measurements (Fig. 1B), the distribution closely resembled that of the entire experimental period, particularly for the rainout shelter plots, suggesting that flux measurements represented the rainout effect well.

https://doi.org/10.5194/egusphere-2026-210
Preprint. Discussion started: 26 January 2026

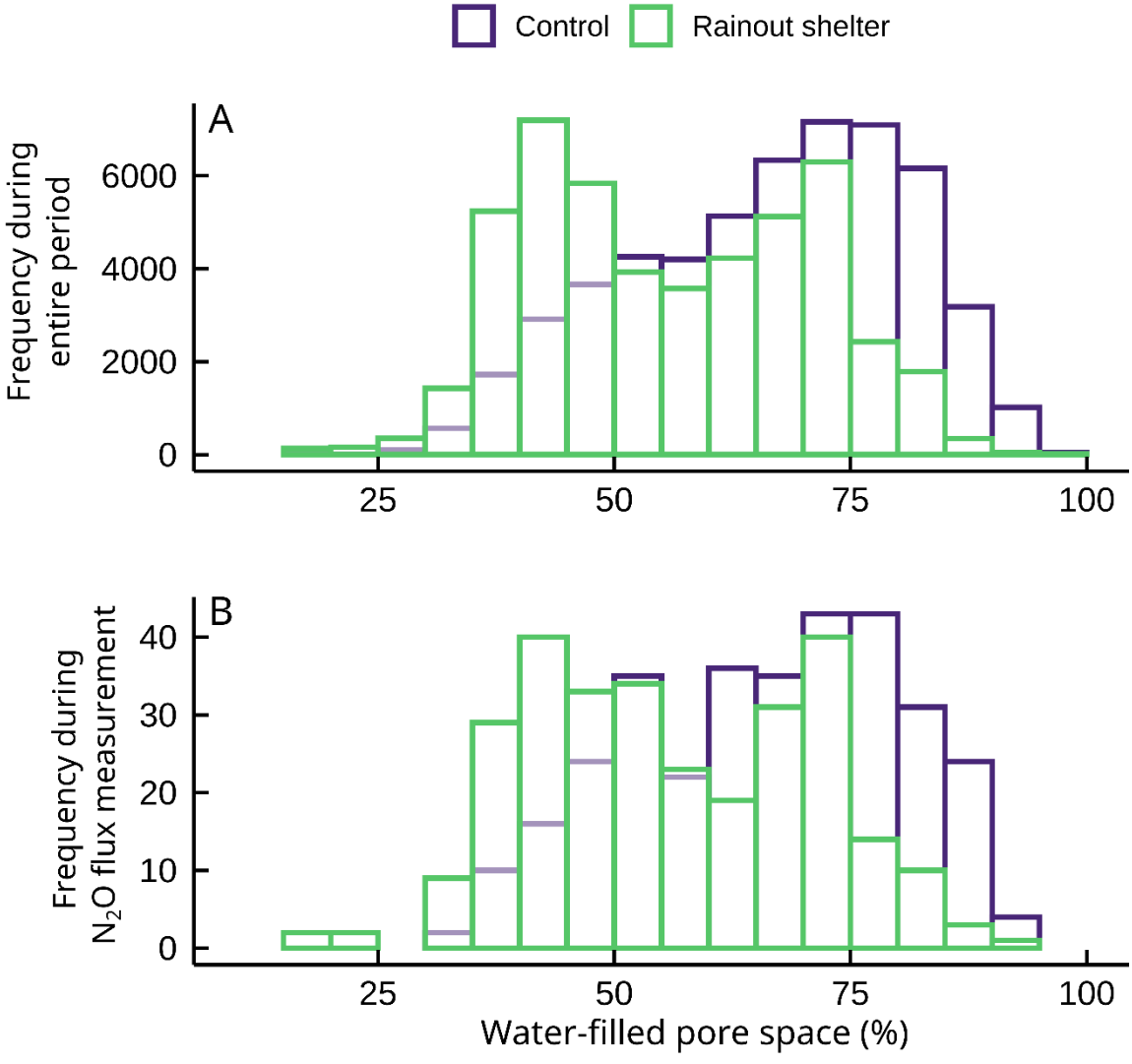

Figure 1: Frequency distribution of water-filled pore space values (WFPS, %) for control plots (purple) and rainout shelter plots (green). (A) WFPS frequency distribution based on hourly measurements during the entire experimental period, from the installation of rainout shelters until the end of gas measurements for both years. (B) WFPS frequency distribution based on average WFPS measurements taken between 11 a.m. and 2 p.m. in each plot during dates of flux sampling. Volumetric water content was measured in 13 out of the 16 plots at a depth of 5 cm, and WFPS was calculated using measured bulk density and a standard particle density

https://doi.org/10.5194/egusphere-2026-210
Preprint. Discussion started: 26 January 2026

3.2 Differences between years

In the first year, N_2O emission measurements started in June right after installing the rainout shelters, about two months after sowing and fertilisation. Emission rates ranged from -10.4 to 149.6 µg N m^{-2} h^{-1} with a median of 5.6 and no significant differences between plots under rainout shelters and control plots (p = 0.8, linear mixed-effect model; Fig. 2A). In the second year, higher emission rates were measured in the control plots (p = 0.005, linear mixed-effect model). Large N_2O emission rates were measured after spring thaw when temperatures were still low, and in autumn right after harvest (Fig. 2A). N_2O emission rates ranged from -21.1 to 441.4 µg N m^{-2} h^{-1} with a median of 22.3, which was clearly higher than observed in the first year. After harvest, larger emission rates were measured under the rainout shelters than in control plots. We did not observe a clear emission response to fertilisation, but some larger emission rates were detected during summer following precipitation events, especially in the control plots.

Mineral N (NO_3^- and NH_4^+) contents at 0-20 cm depth were highest in spring and early summer (Fig. 2B and 2C), without showing a significant difference between the rainout shelter and control plots (p = 0.29 and 0.67 for NO_3^- and NH_4^+, respectively, Fig. S2). We tested differences in mineral N content across both years using a linear mixed-effects model.

Overall, rainout shelters reduced soil moisture (Fig. 2D). However, there were times when high precipitation (Fig. 2E) caused soil moisture in the rainout shelter plots to be similar to or even higher than in the control plots. This occurred in the first year, after two extreme precipitation events in August, when more than 40 mm of precipitation fell within 24 hours (Wolff and Grimenes, 2024); and again in the spring of the second year during snowmelt. The lowest WFPS values were measured at the beginning of the first year, following a drought period prior to the start of the experiment (Wolff and Grimenes, 2024). Similarly, low WFPS was recorded at the end of the second year when temperatures dropped below the freezing point, because time domain reflectometry (TDR) loggers do not detect frozen water (Fig. 2D and 2F).

https://doi.org/10.5194/egusphere-2026-210
Preprint. Discussion started: 26 January 2026

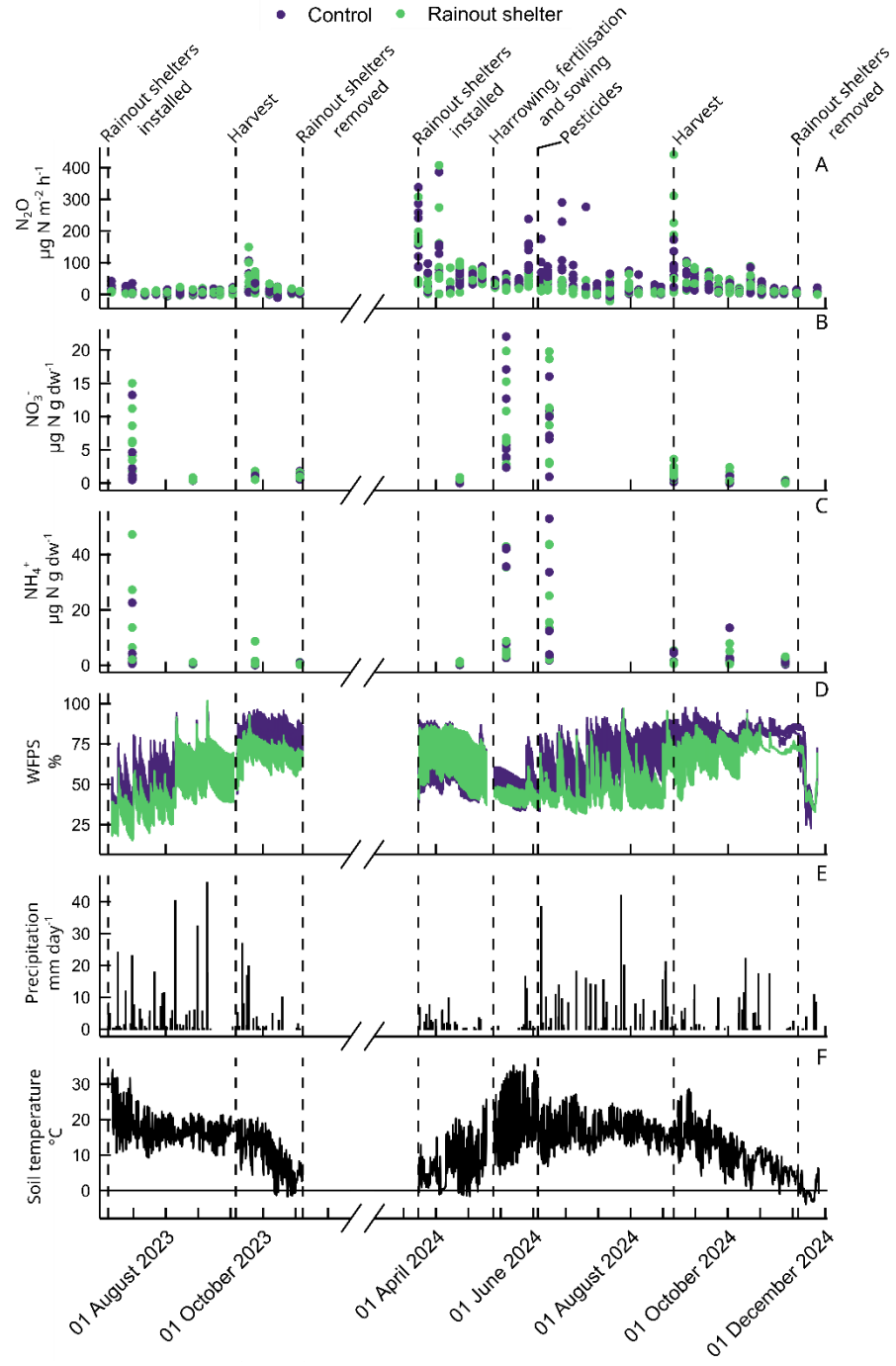

Figure 2: N_2O emission rates (single plot values) and ancillary variables in control plots (purple) and rainout shelter plots (green). (A) N_2O emission rates, (B) and (C) extractable soil NO_3^--N and $NH4^+$-N per g dry weight soil collected at a depth of 0-20 cm, (D) water-filled pore space (WFPS) calculated from volumetric water content, (E) precipitation (mm day^{-1}) obtained from a nearby weather station (Wolff and Grimenes, 2024; Wolff, 2025) and (F) mean soil temperature measured at a depth of 5 cm. Dashed vertical lines indicate key management events

https://doi.org/10.5194/egusphere-2026-210
Preprint. Discussion started: 26 January 2026

3.3 Variation in daily emission rates

N_2O emission rates were largest at WFPS values between 41% and 82% (Fig. 3A) and temperatures below 10°C (Fig. 3B). High flux rates (>100 µg N_2O-N m^{-2} h^{-1}) originated primarily from spring in the second year, when soil moisture was high and temperature was low (Fig. 2D and 2F). Large N_2O emission rates were also observed when temperatures were between 15°C and 25°C, particularly in the control plots. As shown in Fig. 1, rainout shelter plots had fewer N_2O measurements with WFPS values exceeding 75%, thus reducing high emission fluxes during wet conditions. This can also be seen from Fig. 3A, even though some rainout shelter plots still exhibited high N_2O emission rates under conditions of very high WFPS (>80%).

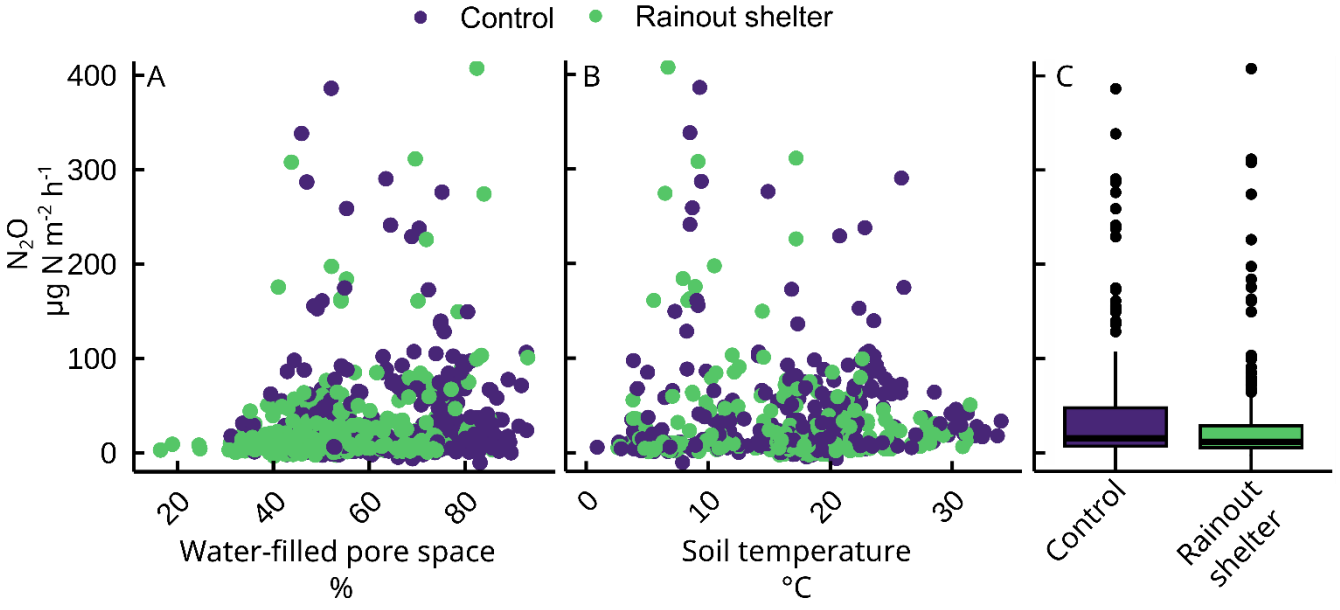

Figure 3: Relationship between N_2O emission rates (µg N m^{-2} h^{-1}) and (A) water-filled pore space (WFPS, %) and (B) soil temperature (C°). Both WFPS and soil temperature were measured at a depth of 5 cm. Data points represent individual measurements with colours indicating either control plots (purple) or rainout shelter plots (green). (C) Boxplot of N_2O emission rates shown in (A) and (B) with median, interquartile range and whiskers indicating variability outside the upper and lower quartiles

3.4 Cumulative N_2O emissions and yield

Cumulative N_2O emissions cannot be compared between the years because of different measurement periods (Table 1). In the first year, cumulative N_2O emissions were smaller in five out of the eight rainout shelter plots compared with their adjacent control plots (Fig. 4A). However, the differences between plots were small, and a non-parametric Wilcoxon rank-sum test showed no significant difference in cumulative N_2O emission between rainout shelter and control plots (Fig. 4C, p = 0.645). In the second year, the differences in cumulative N_2O emissions between treatments were more pronounced (Fig. 4B). Six out of eight rainout shelter plots had smaller emissions than the adjacent control plots. When testing the difference across all plots, cumulative N_2O emissions in the rainout shelter plots were 1.94 ± 0.62 kg N ha^{-1} (±SD) and significantly smaller than the 2.58 ± 0.43 kg N ha-1 (±SD) in the control plots (Fig. 4D, p = 0.038, Wilcoxon rank-sum test).

https://doi.org/10.5194/egusphere-2026-210
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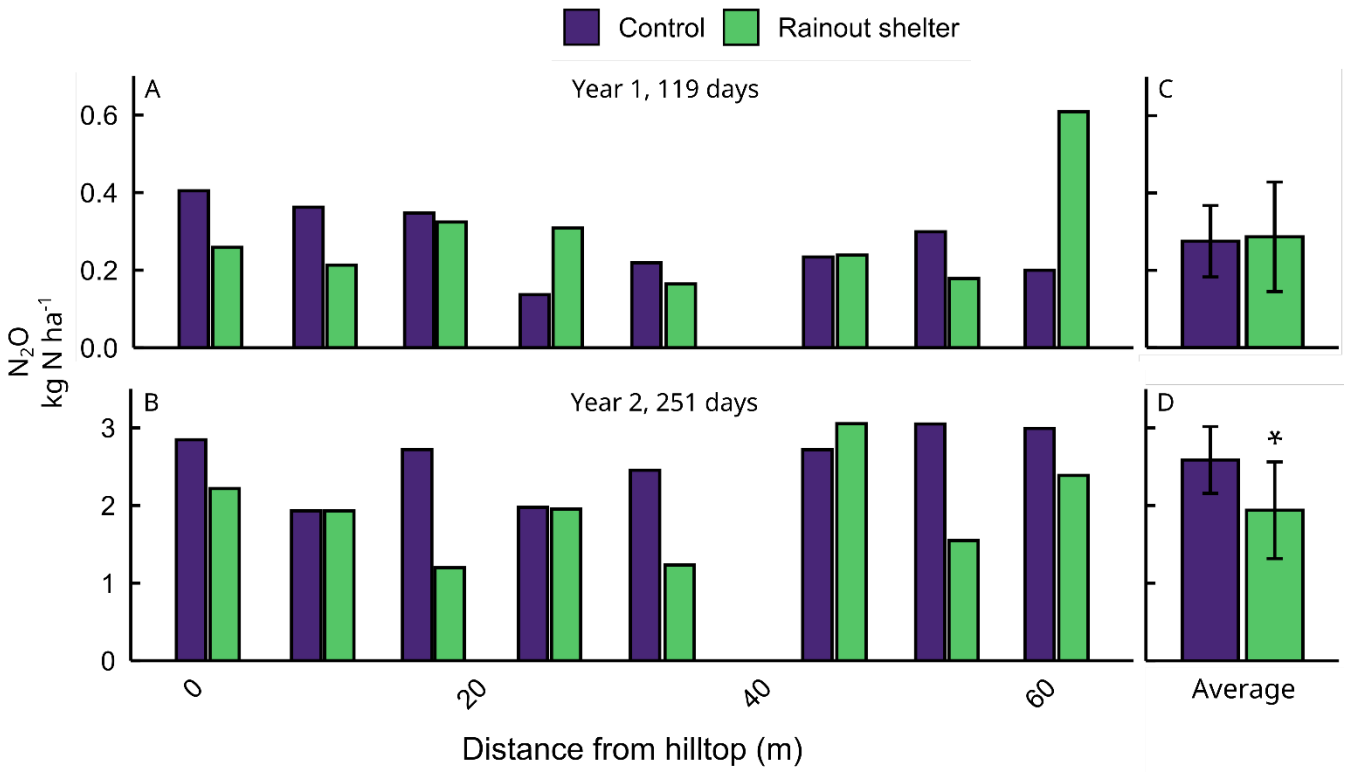

Figure 4: Cumulative N_2O emissions (kg N ha^{-1}) along the transect calculated by linear interpolation for the first year (Year 1, 119 days: A and C) and the second year (Year 2, 251 days: B and D). A and B show cumulative N_2O emissions as a function of distance from the hilltop (m) for control plots (purple) and rainout shelter plots (green). C and D present averages ± standard deviation for control plots (purple) and rainout shelter plots (green) across the hillslope. The asterisk denotes significance ($p < 0.05$) between treatments

The rainout treatment had no significant effect on dry matter yields (DMY) in either year ($p = 0.071$ and 0.435 for the first and the second year, respectively; Fig. 5) but showed a tendency towards lower yields under rainout shelters.

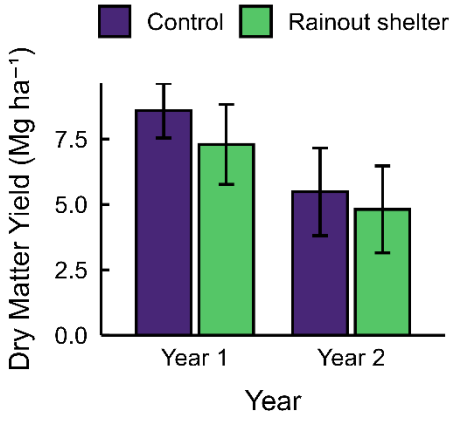

Figure 5: Mean dry matter yield (DMY) of barley for the first (Year 1) and the second (Year 2) experimental year (Mg ha^{-1}). Bars represent the average DMY ± standard deviation (Mg ha^{-1}) for control plots (purple) and rainout shelter plots (green)

https://doi.org/10.5194/egusphere-2026-210
Preprint. Discussion started: 26 January 2026

3.5 Soil properties and cumulative N_2O emissions

Soil properties varied along the hillslope (Table 2) with SOC and clay content increasing and pH decreasing toward the bottom of the slope. We tested the relationships between cumulative N_2O emissions and soil properties separately for rainout shelter and control plots for both experimental years. In the first year, no significant correlations with edaphic parameters emerged for either treatment (control: p = 0.53 (SOC), p = 0.46 (pH), p = 0.78 (clay) and p = 0.48 (bulk density); rainout shelters: p = 0.22 (SOC), p = 0.19 (pH), p = 0.31 (clay) and p = 0.24 (bulk density); data not shown). In the second year, cumulative N_2O emissions in the control plots were positively correlated with SOC content (Fig. 6A, p = 0.04), weakly negatively with soil pH (Fig. 6B; p = 0.08) and weakly positively with clay content (Fig. 6C, p=0.06). This pattern disappeared when analysing cumulative emissions in rainout shelter plots. Here, neither SOC (p = 0.66), soil pH (p = 0.62) nor clay (p = 0.49) was near to significance. Although correlations with pH and clay content in the control plots were not significant, they showed near-significant trends. SOC, pH, and clay content were all significantly correlated with each other ($p < 0.05$), making it difficult to disentangle their individual effects on cumulative N_2O emissions. Bulk density showed no correlation with cumulative N_2O emissions for either treatment (Fig. 6D).

https://doi.org/10.5194/egusphere-2026-210
Preprint. Discussion started: 26 January 2026

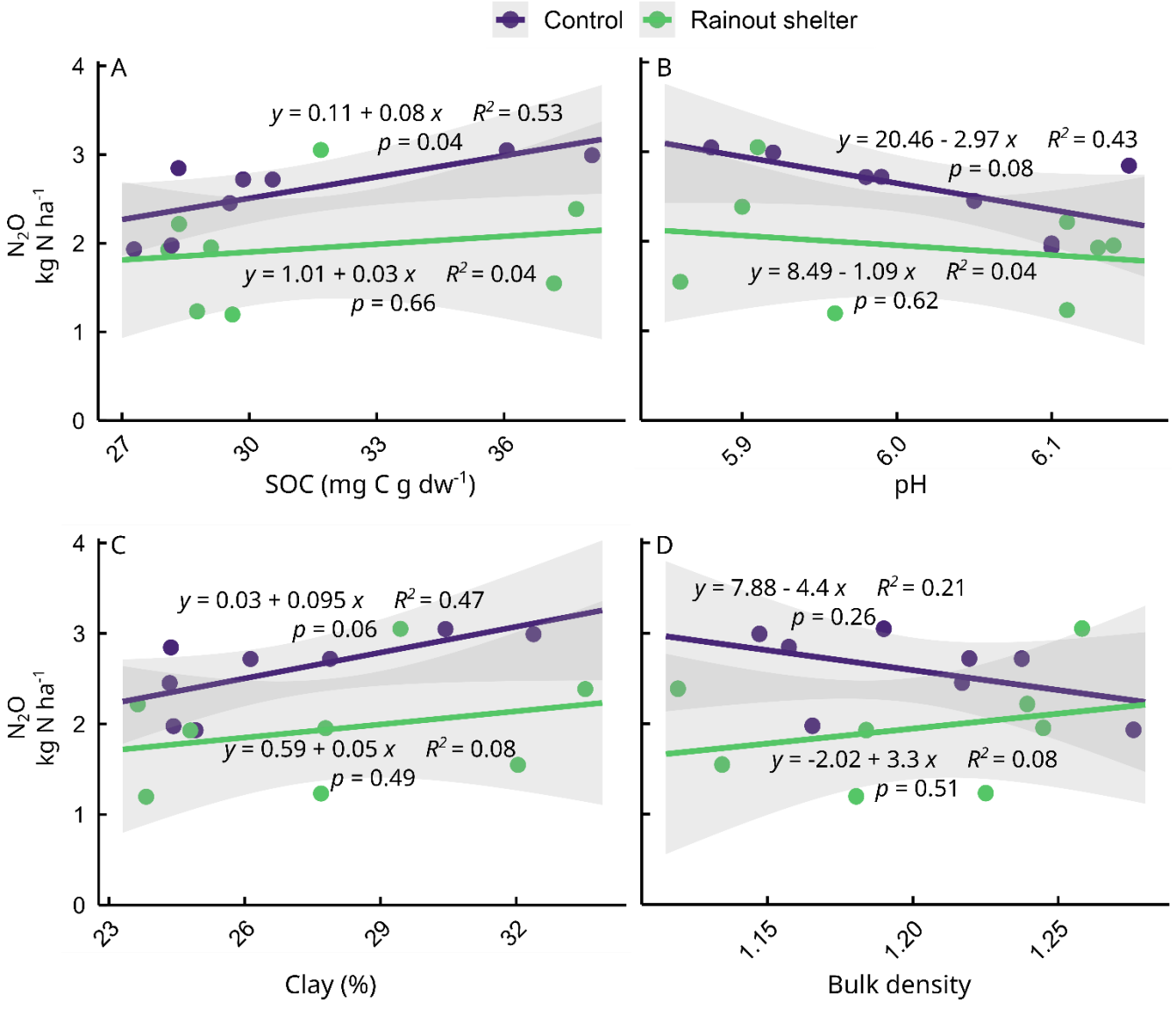

Figure 6: Cumulative N_2O emissions as a function of (A) soil organic carbon (SOC) content, (B) pH, (C) clay content and (D) bulk density (g cm^{-3}) during the second experimental year. Data are shown for control plots (purple) and rainout shelter plots (green). Solid lines are linear regression lines for control or rainout shelter plots with the shaded areas indicating 95% confidence intervals. The regression equations, coefficients of determination (R^2) and *p*-values are displayed next to each regression line

https://doi.org/10.5194/egusphere-2026-210
Preprint. Discussion started: 26 January 2026

4 Discussion

4.1 Effect of reduced precipitation

Climate models predict that Norway, like much of northern Europe, will experience a warmer and wetter climate with more frequent and severe summer droughts (Wong et al., 2011; Hanssen-Bauer et al., 2017; Zhu and Siebert, 2024). The rainout shelters in our study effectively simulated these conditions by consistently reducing precipitation and soil moisture compared to the control plots, especially during the summer months (Fig. 2D and 1). However, during periods of heavy precipitation, soil moisture under the rainout shelters raised to levels comparable or above those in control plots. This resulted in a bimodal WFPS distribution in the rainout shelter plots (Fig. 1), which resembles the future climate with both more frequent low soil moistures due to drought and more frequent wet soils due to increased precipitation.

The effect of the rainout shelters on N_2O emissions differed between the two years. In the first year, average cumulative N_2O emissions did not vary between control and rainout shelter plots (Fig. 4C). The rainout shelters were installed end of June, about two months after fertilisation, which likely limited mineral N availability. Still, high concentrations of mineral N were observed during this period, although this did not translate into higher N_2O emissions (Fig. 2B, C). Later in the first year, extreme precipitation events (> 40 mm day^{-1}; Fig. 2E) cancelled out differences in soil moisture between the control and rainout shelter plots (Fig. 2D). For example, the sixth rainout shelter from the hilltop (plot F; Table 2) had a similar median WFPS as its corresponding control plot, while the eighth rainout shelter (plot H; Table 2) had a higher median WFPS than its control plot during flux measurement periods (Fig. S3A). This resulted in no clear difference in N_2O emissions between control and rainout shelter in plot F and higher N_2O emissions in the rainout shelter plot H (Fig. 4A). Snow came early (October) in the first year, and we only measured N_2O for four months. We believe the combination of limited N availability during the growing season, heavy precipitation events in autumn and the short measurement period explains the absence of a consistent rainout effect on N_2O emissions in the first year.

In the second year, rainout shelters were installed earlier and were deployed for almost nine months, which likely resulted in a more sustained effect of precipitation reduction on soil processes. Average cumulative N_2O emissions were significantly larger in control plots (Fig. 4D), indicating that reduced precipitation decreased soil moisture, and thereby decreased denitrification and N_2O emissions. Consistent with the findings of our study, meta-analyses have shown that rainout shelters generally reduce N_2O emissions compared to ambient precipitation in field studies (Homyak et al., 2017; Li et al., 2020). The primary mechanism for this reduction is a decrease in the anoxic volume in the soil, which limits denitrification (Schlüter et al., 2025; Firestone et al., 1979). Larger N_2O emissions are commonly observed at WFPS values >60%, as anoxic microsites become more prevalent, supporting denitrification (Davidson, 1993; Robertson and Groffman, 2007; Schlüter et al., 2025). Conversely, smaller N_2O emissions can be expected when soil moisture is low (WFPS <60%), as low soil moisture reduces the anoxic volume in the soil and limits diffusion of mineral N and organic C to microbes (Linn and Doran, 1984; Roman-Perez and Hernandez-Ramirez, 2021). This aligns well with our observation of more flux measurements at WFPS <60% and hence smaller N_2O emissions in the rainout shelter plots, whereas control plots had more

https://doi.org/10.5194/egusphere-2026-210
Preprint. Discussion started: 26 January 2026

frequent WFPS values >60% resulting in larger N_2O emissions (Fig. 1B and 4D). Base N_2O emissions, i.e. emissions measured outside periods of high-flux events, appeared to be generally higher in the control plots, displaying a larger interquartile range than emissions in the rainout shelter plots (Fig. 3C).

Plot-specific variability in WFPS and N_2O emissions was observed also in the second year. For instance, in plot F, the rainout shelter plot had a lower median WFPS than the control plot, but the rainout shelter WFPS exhibited greater variability, with some measured WFPS higher than in the control plot (Fig. S3B). This variability likely contributed to higher N_2O emissions in the rainout shelter plot in periods with high WFPS. Despite such plot-level differences, the overall trend of reduced N_2O emissions under rainout shelters was evident.

The reduced N_2O emissions under rainout shelters, however, did not result in a climate benefit when considering the yield. Barley dry matter yield (DMY) exhibited high variability, and although we saw no significant difference between the control and rainout shelter plots in either year, there was a trend towards lower yields in the rainout shelter plots (Fig. 5). In the second year, when the N_2O measurement period was longer and more representative of the growing season, the average yield-scaled N_2O emissions were not significantly different between the two treatments amounting to 0.53 ± 0.23 and 0.48 ± 0.28 kg N_2O-N Mg DMY ha^{-1} for the control and rainout shelter plots, respectively. Although the second year better represents growing season and post-harvest emissions, cumulative N_2O emissions were likely underestimated, as only few freeze-thaw cycles were captured in spring and most of the off-season was not included. Freeze-thaw cycles are known to contribute a large part of the annual emission in cool-temperate climates (Christensen and Tiedje, 1990; Flessa et al., 1995; Wagner-Riddle and Thurtell, 1998; Wagner-Riddle et al., 2024). At our experimental farm, a cover crop study on a neighbouring field found that 62-80% of the annual N_2O emissions occurred during the off-season in plots without cover crops (Kjær et al., 2026b). Investigating how precipitation manipulation during the growing season affects the off-season emissions was beyond the scope of the present study.

4.2 Impact of edaphic drivers

In addition to the dominant effect of precipitation, our results highlight the importance of soil physicochemical properties in driving N_2O emissions when soil moisture is not limiting. In the second year, reduced precipitation not only lowered cumulative N_2O emissions but also diminished the influence of environmental drivers such as SOC, clay content and pH along the hillslope. This suggests that under drier conditions, limited soil moisture can suppress the effect of soil properties in regulating N_2O fluxes.

While other studies have examined N_2O emissions in relation to soil and climate variables across broad management and climate gradients (Jia et al., 2025; Hargreaves et al., 2021), our study offers insights into the effect of small-scale variability on the field level. Along the 62 metres of hillslope, we observed notable variation in SOC (27.3–38.1 mg C g^{-1}), clay (24–34%) and pH (5.86–6.15), with SOC increasing downslope and correlating positively with clay content and negatively with pH.

https://doi.org/10.5194/egusphere-2026-210
Preprint. Discussion started: 26 January 2026

SOC content is often considered a potential driver for N_2O emissions (Li et al., 2005; Stehfest and Bouwman, 2006) among others by creating anoxic microsites due to increased O_2 consumption (Jäger et al., 2011). Particulate organic matter (POM) especially can create hotspots of increased microbial activity and O_2 consumption resulting in enhanced denitrification in otherwise oxic soil (Lucas et al., 2024; Schlüter et al., 2025; Surey et al., 2021). In our study, we found no relationship between POM and N_2O emissions for either ambient or reduced precipitation conditions ($p = 0.86$ and $p = 0.31$, respectively; data not shown), suggesting that the positive correlation between SOC and N_2O was not driven by C availability.

A companion study using soil from the same plots identified clay content as the dominant factor controlling SOC stabilisation (Kjær et al., 2026a). Soils with higher clay content along the gradient had lower microbial respiration potentials and stabilised more ^{13}C-labelled litter into mineral associated organic matter (MAOM). Therefore, the availability of SOC was not higher in the high-SOC soils, refuting the idea that higher N_2O emissions in plots with higher SOC content were fuelled by SOC.

High SOC content is commonly associated with certain soil physicochemical properties, such as improved aggregation, gas diffusivity and increased substrate availability, all of which are known to affect N_2O emissions in several, sometimes opposing, ways (Li et al., 2021; Six et al., 2002b; Kelley et al., 2024). For instance, poor soil aggregation may restrict O_2 diffusion, potentially promoting denitrification and N_2O emissions under wet conditions. Conversely, better aggregation and protection of SOC in microaggregates limits the availability of labile C-substrates for microbes, thereby reducing N_2O fluxes (Sato et al., 2019). The multiple, partly adverse effects of SOC on N_2O make it difficult to pinpoint direct effects.

In our study, SOC and clay content were closely correlated, raising the possibility that the observed positive relationship between SOC and N_2O was due to collinearity with clay content rather than a direct effect of SOC on N_2O turnover. High clay content is known to enhance soil water-holding capacity, creating anoxic microsites conducive to denitrification (Li et al., 2024; Keiluweit et al., 2018). Clay content may also promote N_2O emissions through interactions between soil texture and root exudates, which can create localised anoxic zones (Lacroix et al., 2025). However, we did not see an increase in volumetric water content at field capacity with increasing SOC or clay (Table 2), which may be due to the fine texture of our soils (Minasny and Mcbratney, 2018).

Both SOC and clay content were negatively correlated with pH, and a near-significant relationship was observed between pH and N_2O emissions. A field study from southeastern Norway found that pH, rather than SOC, was the dominant factor controlling N_2O emissions during the off-season (Russenes et al., 2016). This suggests that pH modulates *in situ* N_2O emissions, although its effect is intertwined with SOC and clay content. The decline in pH downslope may be due to the accumulation or production of organic acids (Adeleke et al., 2017), which can be influenced by both SOC and clay content. This interconnectedness indicates that even if pH appears to be a dominant factor controlling N_2O emissions, its effects are modulated by SOC and clay content, making it impossible to fully separate their contributions. In summary, while SOC, clay and pH each showed some degree of influence on cumulative N_2O emissions, their effects were secondary to the impact of precipitation reduction which appeared to be the dominant factor. Despite the confounding effect of edaphic factors, the

https://doi.org/10.5194/egusphere-2026-210
Preprint. Discussion started: 26 January 2026

observed trends in our small-scale study highlight the importance of considering local soil variability when assessing N_2O emissions in agricultural systems.

The most striking finding of our study was that the influence of physicochemical variables on cumulative N_2O emissions changed markedly with precipitation regime in the second year. While it is well established that soil N_2O emissions depend on soil moisture, our rainout shelter experiment demonstrated that manipulating precipitation not only reduced cumulative N_2O emissions (by ~25%), but also effectively cancelled out the relationships between N_2O emissions and SOC content, clay content and pH that were evident under ambient precipitation. The shift in soil moisture distribution towards more frequent WFPS values between 35 and 50% (Fig. 1) likely constrained the range of conditions under which these soil properties exert an effect.

This phenomenon could be explained by a shift in prevailing soil moisture altering the dominant N_2O production process, each with distinct sensitivities to SOC, clay and pH. For instance, nitrification and its N_2O yield are known to be positively related to soil pH (Nadeem et al., 2020) potentially cancelling out the well-known negative relationship between the N_2O product ratio of denitrification and pH (Bergaust et al., 2010). Additionally, the edaphic gradient may have given raise to distinct microbial communities that respond differently to changes in soil moisture. In our companion study (Kjær et al., 2026a) we found that the abundance of *nosZ* was negatively correlated with SOC along the studied gradient ($p < 0.01$) indicating a higher potential for complete denitrification at the top of the hillslope. Rain exclusion may have weakened this control by shifting the soil towards nitrifying conditions.

Most process-based models are limited in capturing dynamic interactions between soil physicochemical and soil moisture controls on N_2O emissions. Our findings highlight that the influence of SOC, clay content and pH on N_2O emissions can be strongly conditional and are easily cancelled out by changes in precipitation regime. This context-dependency presents a challenge for model parameterisation, as relationships derived under one set of moisture conditions may not be transferable to others.

5 Conclusion

This study demonstrates that reduced precipitation limits cumulative N_2O emissions by decreasing soil moisture, likely resulting in less denitrification. We found that reduced precipitation overrode well-known physicochemical controls of N_2O emissions observed under ambient conditions. While soil physicochemical controls on N_2O emissions at small scales are difficult to disentangle, these properties likely interact to shape N-dynamics and control N_2O emissions when soil moisture is not limiting. The observed trends suggest that small-scale soil variability plays an important role in regulating N_2O emissions, but only when precipitation is sufficient to allow denitrification to be the dominant N_2O source. Overall, our findings emphasise the dominant role of precipitation in controlling N_2O emissions in cool-humid crop production, which is particularly relevant as future climate scenarios predict changes in rainfall patterns and the occurrence of droughts.

https://doi.org/10.5194/egusphere-2026-210
Preprint. Discussion started: 26 January 2026

Data availability

Data and code are available at https://doi.org/10.5281/zenodo.18173363.

Author contributions

Sigrid Trier Kjær: Conceptualisation, Data curation, Formal analysis, Investigation, Visualisation, Writing – original draft, Writing – review & editing. **Peter Dörsch**: Conceptualisation, Funding acquisition, Supervision, Writing – review & editing.

Declaration of generative AI and AI-assisted technologies

During the preparation of this work, ChatGPT (GPT-4; OpenAI) was used to improve readability and language in the manuscript and for data handling by writing code for R. All output from the AI tool were quality checked and revised by authors before going into the manuscript.

Competing interests

The authors declare that they have no conflict of interest.

Acknowledgements

We would like to thank Trygve Fredriksen, Øyvind Peder Vartdal and Elias Rogstad for their valuable assistance in field and laboratory work. Sigrid would also like to thank Lena Bakker and Eliane Schmid for support and writing huddles.

Financial support

This work was supported by TRUESOIL, a project funded by the 1st external call from the European Joint Programme EJP SOIL (Horizon 2020: 862695), with funding from the Research Council of Norway (Project number 336692).

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