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Dynamics of greenhouse gas fluxes and soil physico-chemical properties in agricultural and forest soils

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ABSTRACT

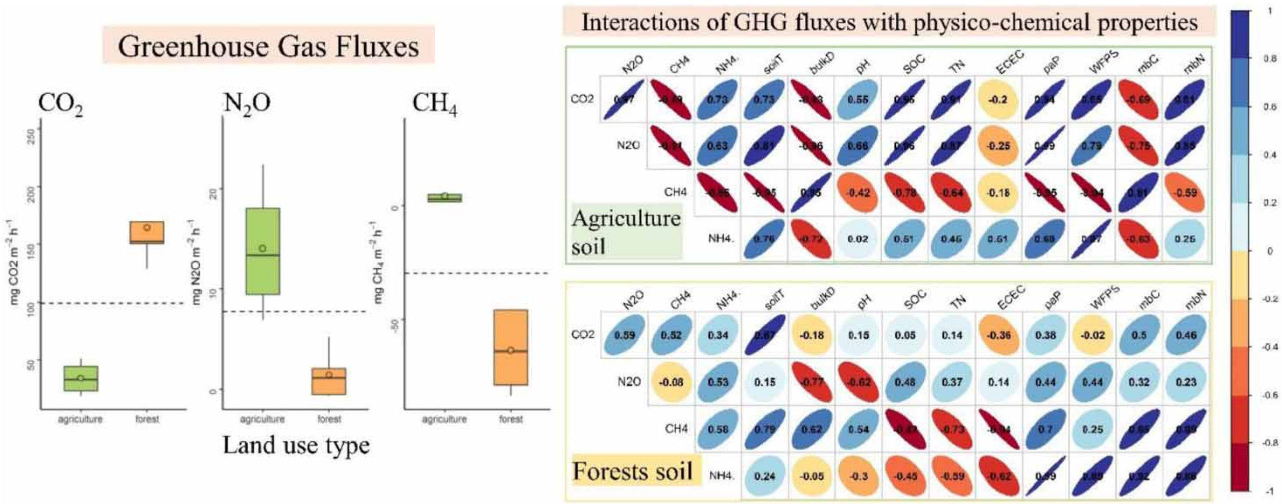
Examination of greenhouse gas (GHG) fluxes (CO_2, CH_4, and N_2O) in soils is crucial for developing effective strategies to mitigate climate change. In this study, we investigated the GHG fluxes in agricultural and forest soils to explore the changes in soil GHG fluxes, and assess the relationships of GHGs with other physico-chemical properties. Results show that forest soils have a higher CO_2 flux, while agricultural soils have a higher N_2O flux due to fertilizer application and heterotrophic nitrification. Forest soils act as a CH_4 sink, which are connected with increased porosity and decreased bulk density. In agricultural soils, CO_2 and N_2O were strongly linked with NH_4^+, soil temperature, pH, soil organic carbon, total nitrogen, plant-available phosphorous, and microbial biomass nitrogen (mbN) but were negatively connected with bulk density and microbial biomass carbon (mbC). In contrast to CO_2 and N_2O, CH_4 in agricultural soils exhibited inverse relationships with all physico-chemical properties. In forest soils, CO_2 and CH_4 were positively correlated with soil temperature and mbC, and mbN and N_2O were negatively correlated with bulk density and pH. This study highlights the critical need to comprehend the complex relationship between soil physico-chemical properties and GHG fluxes for effective climate change mitigation.

Key words: agriculture, forests, greenhouse gas fluxes, land use, physico-chemical properties, soil organic carbon

HIGHLIGHTS

- CO_2 flux of forest soils is greater than that of agricultural soils.
- Agricultural soils have a greater N_2O flux than forest soils.
- Forest soils have been identified as a CH_4 sink.
- CO_2 and N_2O fluxes were significantly correlated with NH_4^+, soil temperature, pH, and soil organic carbon.

GRAPHICAL ABSTRACT



1. INTRODUCTION

Greenhouse gas (GHG) emissions in the atmosphere have been predicted to result in significant climatic and ecological consequences (IPCC 2021). Climate change can also have an impact on variance in soil consumption and the release of major GHGs (Blankinship *et al.* 2010; van Groenigen *et al.* 2011; Feng *et al.* 2022). GHGs such as carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) have been accumulating at an alarming rate in the atmosphere in recent decades as a result of human activity (IPCC 2021). A greater knowledge of GHG cycles is necessary for more accurate estimation of future global change, as climate-induced instability of ecosystem functioning (above- and below-ground biomass production; Hossain & Li 2020; Hossain *et al.* 2022; Jabal *et al.* 2022) may alter GHG fluxes in soils. For example, in recent years, carbon uptake in grassland (Hossain *et al.* 2022) and forest (Smith *et al.* 2020) ecosystems in Germany has been found to be affected by extreme droughts and heatwaves, which ultimately impair GHG fluxes by reducing soil organic matter in these ecosystems.

Organic matter retains over 1,500 Pg of carbon on the earth's surface, roughly double the amount of carbon retained in the atmosphere (Jin *et al.* 2000). As a result, both the carbon pool in soils and soil carbon loss through emissions have a significant effect on atmospheric CO_2 concentrations. A considerable influence on the process of carbon balance between the pedosphere and atmosphere is exerted by land use changes and soil degradation (Zhang *et al.* 2020). The nutrient cycles (i.e., carbon and nitrogen), as well as the variations in GHG concentrations that result from them, are significant components of the biogeochemical cycle (Ekeleme *et al.* 2021). Environmental worries are currently concentrated on the steadily increasing levels of these GHGs in the atmosphere (Hashimoto *et al.* 2011; IPCC 2021; Carrillo *et al.* 2022), which are primarily the result of human activity. Thus, concerns of GHG sinks and sources have risen to the top of the global priority list in recent years (Chang *et al.* 2021; Harris *et al.* 2021; Hossain & Li 2021a; Barthel *et al.* 2022). A closer examination of the organic carbon and nitrogen stored in the global GHG reservoir, as well as the processes that control it, will enable researchers to gain a better understanding of the soil carbon and nitrogen cycle's role in global warming and climate change (Hossain *et al.* 2021, 2023a, 2023b). Despite the growing evidence of climate change impacts on grassland and forest ecosystems (Hossain *et al.* 2021, 2023a, 2023b; Carrillo *et al.* 2022), understanding of how GHG fluxes in these ecosystems change under increasing frequency and intensity of extreme climatic conditions remains poorly defined. Numerous studies have been conducted on the local and regional scales to determine the relationship between soil organic carbon, nitrogen, and site factors (Hashimoto *et al.* 2011; Tellez-Rio *et al.* 2017; Harris *et al.* 2021; Barthel *et al.* 2022; Līcīte *et al.* 2022). These studies established that connections varied according to the land use types, vegetation types, environment, and soil types (Busman *et al.* 2023).

CO_2 is a notably powerful GHG, and soil respiration is the major pathway through which terrestrial ecosystems (e.g., forests, shrublands, and grasslands) release CO_2 into the atmosphere (Schlesinger & Andrews 2000). Net soil CO_2 exchange is a consequence of root respiration and decomposition, soil organic matter, and microbial community composition in the soil.

A growing number of studies indicate that microbial communities can adapt their metabolic and substrate demands in response to available carbon and nitrogen (Manzoni *et al.* 2008; Eberwein *et al.* 2015). Methane accounts for more than 20% of the total radiative forcing of GHG emissions (IPCC 2013). Soil CH_4 emissions and consumption are influenced by a variety of factors, including moisture, oxygen, and nitrogen concentrations that are readily available in the soil (Aronson *et al.* 2013; Aronson *et al.* 2019). Environmental factors such as anaerobic conditions and the availability of precursor chemicals such as ammonia (NH_3), nitrite (NO_2), nitrate (NO_3), and nitric oxide (NO) encourage the production of N_2O and nitrogen gas (N_2). On the other hand, aerobic conditions, as well as the presence of ammonia (NH_3^-) for ammonia oxidation and nitrite (NO_2^-) for nitrite oxidation, enhance the production of N_2O and N_2 through the nitrification process.

Forest and agricultural soils are critical in terms of GHG sources and sinks (Hashimoto *et al.* 2011; Harris *et al.* 2021; Hossain & Li 2021b; Barthel *et al.* 2022). Forest soils act as a sink of CH_4 and a source of CO_2 and N_2O (van Groenigen *et al.* 2011). Despite agricultural soils sequestering large amount of carbon ($4 \times 10^8 - 9 \times 10^8$ Mg C yr^{-1}; Chen *et al.* 2016), this sector is thought to be a substantial source of GHGs worldwide, accounting for 10–12% of all anthropogenic GHGs (Tellez-Rio *et al.* 2017). Land use (e.g., agriculture and forest), and environmental conditions both influence GHG fluxes, rendering it susceptible to soil pH, organic nitrogen and carbon content, soil temperature and moisture, as well as the type and size of soil microbial communities (Braun *et al.* 2013; Aronson *et al.* 2019; Hossain & Li 2021a, 2021c). N_2O fluxes, for example, have been found to be significantly lower in grasslands in central Saskatchewan (Corre *et al.* 1999) than in cultivated soils in central Alberta (Nyborg *et al.* 1997) and in south central Saskatchewan (Izaurralde *et al.* 2004). This has been observed in both grassland and farmland locations throughout the world, including the United States (Dass *et al.* 2018), China (Genxu *et al.* 2002), and Central Europe (Hörtnagl *et al.* 2018). The amount of CO_2 released by vegetation and soil respiration varies depending on the type of vegetation, ranging from 1–15 kg C ha^{-1} d^{-1} in mixed grass prairie (LeCain *et al.* 2000) to 2–106 kg C ha^{-1} d^{-1} in tallgrass prairie (LeCain *et al.* 2000). Summer peak respiration rates in the mixed grass prairie range from 13 kg C ha^{-1} d^{-1} in Saskatchewan to 69 kg C ha^{-1} d^{-1} in North Dakota (Frank *et al.* 2002). Soils that have not been disturbed, on the other hand, consume CH_4 more frequently than they produce it (Wang & Bettany 1995). A mixed grass prairie in North Dakota absorbed 7.2 g of CH_4 ha^{-1} d^{-1} (Liebig *et al.* 2008).

For climate policy purposes, reliable quantification of the key GHGs is critical. As part of the Paris Climate Agreement, each signatory country is required to properly record GHG emissions from several fields, including farmland, forests, and other land uses. Additionally, countries must explore feasible GHG mitigation strategies, particularly establishing protected areas (Nila *et al.* 2019). Long-term GHG emissions measurements from different land uses, particularly agriculture and forest, are uncommon. *In-situ* data have the potential to adequately capture the small-scale variability of GHG emissions in agriculture and forest, which is necessary for stakeholders to receive targeted mitigation advice. The availability and consistency of such direct ecosystem GHG flow data are critical for evaluating the potential for GHG reductions associated with various management techniques (Luyssaert *et al.* 2014; Hörtnagl *et al.* 2018; Zhao *et al.* 2023). By understanding the relationships between soil properties, land use, and GHG fluxes, we can develop more effective strategies for mitigating GHG and promoting carbon sequestration in soil.

Soils are highly variable in their physico-chemical properties, which can influence GHG fluxes (Rütting *et al.* 2021). This variability can make it challenging to identify consistent relationships between soil properties and GHG emissions. GHG emissions from soils are influenced by a complex interplay of biotic and abiotic factors, including soil microbes, climate, and management practices. It can be challenging to disentangle the effects of these factors and identify the specific mechanisms that drive GHG emissions. Previous studies provide information on the relationships between GHG fluxes and soil properties at a single land use type (Drewer *et al.* 2021; Soosaar *et al.* 2022), however, the information on how these relationships may differ or be consistent across different land use types is poorly defined.

The study of GHG fluxes in soils is critical for understanding the relationships between soil physico-chemical properties and GHG emissions (Busman *et al.* 2023). The conceptual framework for this study is based on the premise that soil properties such as NH_4^+, soil temperature, pH, soil organic carbon, total nitrogen, plant-available phosphorous, microbial biomass nitrogen, and bulk density may influence GHG fluxes from soils. These soil properties can be influenced by factors such as land use, management practices, and climate. Understanding the relationships between GHG fluxes and physico-chemical properties is crucial for several reasons. Firstly, it can help identify the drivers of GHG emissions in different ecosystems, which is essential for developing effective strategies to mitigate climate change. For instance, knowing that nitrogen fertilization in agricultural soils is a major source of N_2O emissions can inform the development of targeted fertilizer management practices to reduce emissions (Soosaar *et al.* 2022). Secondly, understanding the linkages between GHG fluxes and physico-chemical properties can help predict the response of ecosystems to environmental changes, such as climate variability or land use change. For example, changes in soil temperature or moisture due to climate change can alter the activity of soil microorganisms, leading to changes in GHG emissions. Thirdly, knowledge of the relationships between GHG fluxes and physico-chemical properties can help identify potential feedback between ecosystems and the climate system. For instance, increased CO_2 emissions from soil respiration due to climate warming can lead to further warming, creating a positive feedback loop. Overall, understanding the complex and multifaceted relationships between GHG fluxes and physico-chemical properties is essential for developing effective strategies to mitigate climate change and anticipate the impacts of environmental changes on ecosystems.

The theoretical framework for this study draws on existing theories related to soil biogeochemistry and microbial ecology. Soil biogeochemical processes such as nitrification and denitrification can contribute to N_2O emissions (Rütting *et al.* 2021), while methanogenesis and methanotrophy can influence CH_4 emissions. Microbial communities in soil play a crucial role in mediating these biogeochemical processes. The study also incorporates theories related to the physical properties of soil, such as soil porosity and bulk density, which can influence gas diffusion and transport in soil. The research framework proposes that GHG fluxes from soils are influenced by a complex interplay of biotic and abiotic factors. Forest and agricultural soils may exhibit different patterns of GHG fluxes due to differences in soil properties and management practices.

The study aims to (i) identify the specific soil physico-chemical properties that are strongly linked to GHG fluxes in different types of soils and (ii) quantify the changes in soil GHG fluxes in forest and agricultural soils. The overarching goal is to develop a more comprehensive understanding of the factors that drive GHG emissions from soils and to identify strategies for reducing these emissions.

The study identified specific soil properties such as NH_4^+, soil temperature, pH, soil organic carbon, total nitrogen, plant-available phosphorous, microbial biomass nitrogen, and bulk density that are strongly linked with GHG fluxes in different types of soils. This information can be used to develop more targeted strategies for mitigating GHG emissions. The study identified forest soils as a CH_4 sink due to their negative CH_4 fluxes, which are connected with increased porosity and decreased bulk density. This finding highlights the potential role of forests in mitigating GHG emissions. The study provided insights into the complex interactions between soil properties and GHG emissions, highlighting the need for a more comprehensive approach to understanding the factors that drive GHG emissions from soils.

The scientific novelty of this research is the identification of specific physico-chemical properties that are closely linked with GHG fluxes in different types of soils. The study highlights the complex relationships between soil properties and GHG emissions and provides new insights into the mechanisms driving these relationships. The finding that forest soils have a higher CO_2 flux than agricultural soils and are a CH_4 sink due to negative CH_4 fluxes, while agricultural soils have a higher N_2O flux due to fertilizer application and heterotrophic nitrification, provides important information for understanding the differences in GHG emissions between these soil types. Additionally, the study identifies specific soil properties such as NH_4^+, soil temperature, pH, soil organic carbon, total nitrogen, plant-available phosphorous, and microbial biomass nitrogen that are strongly linked with CO_2 and N_2O fluxes in agricultural soils, and with CH_4 fluxes in forest soils. This information can be used to develop more targeted strategies for mitigating GHG emissions in different types of soils, which could have important implications for climate change mitigation efforts. Overall, the study provides a valuable contribution to the scientific understanding of the relationships between soil properties and GHG emissions, and the potential for soil management practices to reduce these emissions.

2. MATERIALS AND METHODS

2.1. Study area

The study was conducted on two land use types (agriculture and forest) in Goettingen, Germany. The study site is situated on the Nord (North) Campus, University of Goettingen in the Goettinger Wald (Lower Saxony, Germany) on a plateau around 400–420 m above sea level (Wolters 1999). The agriculture site (grassland ecosystems) was used for hay meadow production during the last 60 years, which received moderate fertilizers. In the forest site, the main tree species are beech (*Fagus sylvatica*) with a combination of ash (*Fraxinus excelsior*) and maple (*Acer campestre*). A considerable number of the regeneration, composed of broad-leaved species, was observed during the field study. The climate data were obtained from the Goettingen weather station, which is run by the German Weather Service. Goettingen has a mean annual air temperature of 7.9 °C and receives 720 mm of mean annual rainfall. According to the FAO and USDA classification systems, the soil is classified as

Chromo-Calcic Cambisol and Lithic Rendoll, respectively (Wolters 1999). The bedrock is shell-lime, and the humus form is mull.

2.2. Soil sample collection

We selected 20 sampling plots in agricultural and forest sites. On each site, we collected soil samples at every 15 m distance between plots (Figure 1). Soil samples were collected using soil probes and brought to the laboratory to analyze the GHG fluxes (CO_2, N_2O, and CH_4), effective cation exchange capacity (ECEC), total carbon, total nitrogen, pH, bulk density, base saturation, water-filled pore space (WFPS), plant-available P, microbial biomass C and N, mineral N in NO_3, microbial C:N ratio, and NH_4^+.

The parameters used in this study were chosen based on their known or potential relationships with GHG emissions and soil physico-chemical properties. The rationale behind the selection of each parameter is because of the following reasons. GHG fluxes are the focus of this study, as they are the ultimate indicator of soil emissions (Barthel *et al.* 2022). CO_2 is a product of respiration and decomposition, N_2O is produced during nitrification and denitrification, and CH_4 is produced during anaerobic decomposition. ECEC is a measure of the soil's ability to hold and release positively charged ions, which can influence nutrient availability and microbial activity. Total carbon and total nitrogen are important indicators of soil organic matter, which can influence nutrient cycling and microbial activity. Soil pH can influence microbial activity and nutrient availability, and it has been shown to influence GHG emissions. Bulk density is a measure of soil compaction, which can influence gas diffusion and transport in soil. Base saturation is a measure of the proportion of cation exchange sites that are occupied by basic cations such as calcium and magnesium, which can influence nutrient availability and microbial activity. WFPS is a measure of the amount of pore space in soil that is filled with water, which can influence gas diffusion and transport in soil. Plant-available phosphorus is an important nutrient for plant growth and can influence microbial activity. Microbial biomass C and N are important indicators of microbial activity, which can influence nutrient cycling and GHG emissions. Mineral N in NO_3 is an important indicator of nitrogen availability, which can influence microbial activity and GHG emissions. The microbial C:N ratio is a measure of the relative amounts of carbon and nitrogen in the microbial biomass, which can influence nutrient cycling and GHG emissions. Ammonium is an important source of nitrogen for plants and microbes, and can influence microbial activity and GHG emissions. Overall, the selected parameters represent a comprehensive set of indicators that can provide insight into the complex relationships between soil physico-chemical properties and GHG fluxes.

While the parameters used in this study provide a comprehensive set of indicators, there are several other parameters that could also be relevant for understanding the relationships between soil physico-chemical properties and GHG emissions. Some potential parameters that are not included in the study are: soil texture, land management practices, soil moisture, redox potential, soil enzymes, trace elements, and soil respiration. Although these parameters can influence nutrient

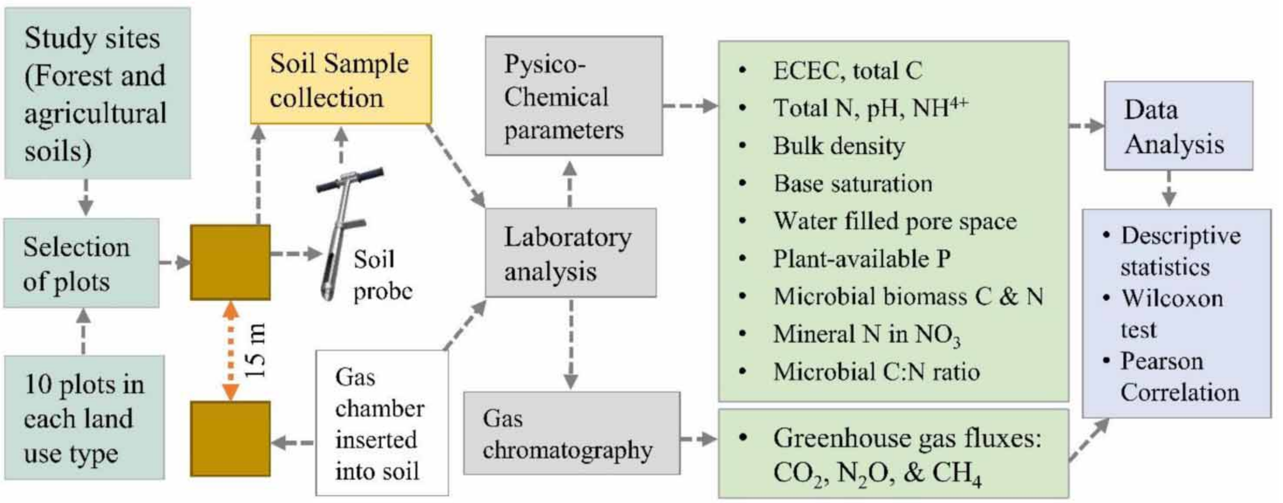


Figure 1 | Flowchart of the study design, sampling procedure, sample collection, and laboratory analysis of soil samples collected from forest and agricultural soils.

availability, nutrient cycling, CO_2 production from soil, gas diffusion, and microbial activity, they may not have been deemed critical for understanding the relationships between the physico-chemical properties and GHG fluxes investigated in this study. However, future studies could explore the relationships between GHG fluxes and these additional parameters to further enhance our understanding of soil processes and GHG fluxes.

2.3. Measurement of soil physico-chemical properties

The bulk density of the soil was measured using the core method at a depth ranging from 0 to 5 cm (Blake & Hartge 1986). The soil-to-water ratio was 1:2.50 for the purpose of determining the pH of the soil. Because soils contain Triassic limestone, we utilized acid fumigation to remove carbonates from the limestone before analyzing the soil organic carbon (Harris *et al.* 2001). The C and N contents of three separate samples, including air-dried, ground, and acid fumigated soils, were determined using a CN analyser (Elementar Vario EL; Elementar Analysis Systems GmbH, Hanau, Germany). An inductively coupled plasma-atomic emission spectrometer was used to calculate resin-exchangeable P (iCAP 6300 Duo VIEW ICP Spectrometer, Thermo Fischer Scientific GmbH, Dreieich, Germany). The chloroform fumigation-extraction method was used to estimate soil microbial biomass C and N in the top 5 cm of depth. Soil samples were divided into two parts: one part with 0.5 M K_2SO_4 under laboratory conditions, and another part fumigated with $CHCl_3$ period of 5 days. Extraction was done after that. Organic C was analyzed in the extracts by using UV-enhanced persulfate oxidation with the use of a Total Organic Carbon Analyzer (TOC-Vwp, Shimadzu Europa GmbH, Duisburg, Germany). The organic nitrogen content of the extracts was measured using UV-enhanced persulfate digestion followed by hydrazine sulfate reduction and colorimetry with a continuous flow injection (Method G-157-96; SEAL Analytical AA3, SEAL Analytical GmbH, Norderstedt, Germany). Fumigated and unfumigated soils were separated by kC = 0.45 and kN = 0.68 during a 5-day fumigation period to determine the soil microbial biomass C and N, as well as the difference in organic C and N extraction (Brookes *et al.* 1985). From unfumigated soil extracts, using continuous flow injection colorimetry, NH_4^+ and NO_3^- concentrations were determined (SEAL Analytical AA3, SEAL Analytical GmbH, Norderstadt, Germany); NH_4^+ was determined using a salicylate and dicloroisocyanuric acid reaction (Autoanalyzer Method G-102-93); and NO_3^- was analyzed using the cadmium reduction method with NH_4Cl buffer (Autoanalyzer Method G-254-02).

2.4. GHG fluxes measurements

GHG fluxes were measured using vented static chambers. All agricultural and forest sites were identified for the purpose of collecting gas samples utilizing chambers. Chambers were put into the soil and samples were collected every 10 min at 2, 12, 22, and 32 min. The sample of chamber gas was then transferred to 20 ml pre-evacuated glass vials (Exetainer; Labco Limited, Lampeter, UK). Within 48 h, the concentrations of the soil CO_2, N_2O, and CH_4 fluxes were determined using gas chromatography (GC 6000 Vega series 2, Carlo Erba Instruments, Milan, Italy). To determine gas concentrations, three to four standard gases were used to compare integrated samples (1,002, 2,999 and 4,998 for CH_4 ppb, 359, 1,004, and 1,601 for N_2O ppb, and 402, 1,501, and 3,008 for CO_2 ppb; Deuste Steiniger GmbH, Germany).

To calculate gas fluxes using this formula:

$$\Phi = \frac{V}{A}\left(\frac{P}{R \cdot T}\right) Mf \frac{\partial c}{\partial t}$$

where Φ is flux (g N m^{-2} h^{-1}), A is the chamber area (m^2), V is the chamber volume (L), R is the ideal gas constant (8.315 Pa m^3 mol^{-1} K^{-1}), P is the atmospheric pressure (Pa), T is temperature (K), $\delta c/\delta t$ is the rate of gas concentration change within the chamber (ppm h^{-1} = μL L^{-1} h^{-1}), M is the molar mass of N_2O–N, CH_4–C, or CO_2–C (g mol^{-1}), and f is the conversion factor (10^{-9} m^3 μL^{-1}).

Negative fluxes depict the consumption of the gas by the soil, and positive gas fluxes represent the emission of the gas from the soil. As a control, zero fluxes were included.

2.5. Data analysis

The data were analyzed using statistical methods to determine the distribution characteristics of the GHG fluxes and physico-chemical parameters in forest and agricultural soils. The distribution characteristics of the statistics included the skewness, mean, standard deviation, standard error, confidence interval, and coefficient of variation, which have been shown in Table 1. Nash–Sutcliffe efficiency, root mean square error, and r-squared are the common metrics to evaluate the

Table 1 | The distribution characteristics (skewness, mean, standard deviation, standard error, confidence interval, and coefficient of variation) of greenhouse gas fluxes and physico-chemical properties in forest and agricultural soils

	Skewness		Mean		Standard deviation		Standard error		Confidence interval		Coefficient of variation (%)	
Parameters	**Forest**	**Agriculture**	**Forest**	**Agriculture**	**Forest**	**Agriculture**	**Forest**	**Agriculture**	**Forest**	**Agriculture**	**Forest**	**Agriculture**
CO_2	−0.58	0.19	157.73	33.82	16.96	12.20	5.36	3.86	3.32	2.39	11	36
N_2O	0.62	−0.61	1.43	8.82	2.14	10.09	0.68	3.19	0.42	1.98	150	114
CH_4	−0.02	0.72	−63.29	4.35	16.19	6.95	5.12	2.20	3.17	1.36	−26	160
NH_4^+	1.63	1.09	85.19	45.93	56.99	11.01	18.02	3.48	11.17	2.16	67	24
Soil temp.	−1.00	−0.41	12.54	13.32	0.08	0.08	0.03	0.02	0.02	0.02	1	1
Bulk density	1.03	0.31	0.95	1.33	0.18	0.09	0.06	0.03	0.03	0.02	19	7
pH	0.74	−0.76	4.96	8.49	0.33	0.07	0.10	0.02	0.06	0.01	7	1
SOC	0.46	0.72	1,485	846	497	102	157	32.53	97.54	20.16	34	12
TN	0.48	1.15	92.20	82	16.80	5.16	5.31	1.63	3.29	1.01	18	6
ECEC	1.44	−0.24	3,719	28,450	1,242	1,215	393	384	243	238.	33	4
paP	1.55	−1.15	330.40	5,010	207	723	65.48	228	40.58	141.81	63	14
WFPS	0.25	0.34	17.40	63.60	1.43	5.56	0.45	1.76	0.28	1.09	8	9
mbC	0.95	0.37	25,456	32,226	5,041	3,901	1,594	1,233	988	764	20	12
mbN	1.08	0.81	2,273	3,515	611	140	193	44.51	119	27.59	27	4

performance of the model, of which r-squared is widely used to measure the goodness of fit of a regression model and to identify the most important predictors in the model (Moriasi *et al.* 2007; Burgan & Aksoy 2022). As our study aims to assess the relationships between soil physico-chemical properties and GHG fluxes, we considered r-squared values to identify the most important predictors in the model for forest (Supplementary Table S1) and agricultural (Supplementary Table S2) soils. Statistical analyses were performed in R (R Development Core Team 2020). A Wilcoxon test was performed to explore the variations of GHG fluxes in two land use types (Wilcoxon 1945). Here, GHG fluxes were the dependent variables, and land use types were the independent variables. To investigate the degree of association between soil physico-chemical parameters and GHG fluxes in each land use type, we conducted a Pearson's correlation analysis (Das *et al.* 2023) using the 'corrplot' package in R (R Development Core Team 2020). For each land use category, a correlation matrix was created to represent the relationships between GHG fluxes and soil physico-chemical properties. The correlation matrix identifies the variables in the datasets that are highly associated. Each of the correlation coefficients was color-coded based on its value (blue for positive correlation, red for negative correlation) and displayed in the upper triangle of the correlogram. Using the 'psych' package, the *p*-values of Pearson's correlation coefficients between the variables were calculated and displayed.

3. RESULTS

3.1. GHG fluxes

The CO_2 flux ranged between 18.76 and 51.16 μg C m^{-2} h^{-1} in agriculture and between 129.12 and 181.44 μg C m^{-2} h^{-1} in forest (Figure 2(a)). The average CO_2 flux across the land use types varied significantly (Figure 2(a), $p < 0.001$), where CO_2 flux was the highest in forest (157.72 μg C m^{-2} h^{-1}) and the lowest in agriculture (33.82 μg C m^{-2} h^{-1}). The N_2O flux ranged between 8.01 and 22.43 μg N m^{-2} h^{-1} in agriculture and between −0.9 and 5.21 μg N m^{-2} h^{-1} in forest, where negative numbers indicate uptake by the soil (Figure 2(b)). The average N_2O flux between the land use types varied significantly (Figure 2(b), $p < 0.001$). The N_2O flux in agriculture was higher than the N_2O flux in forest (Figure 2(b), all $p < 0.001$). The CH_4 flux significantly differed between the land use types (Figure 2(c), $p < 0.001$). The CH_4 flux was high in agriculture, which ranged between −4.43 and 16.36 μg CH_4 m^{-2} h^{-1} (Figure 2(c)). Forest is the sink of CH_4, where the CH_4 flux ranged between −83.87 and −44.85 μg CH_4 m^{-2} h^{-1} (Figure 2(c)).

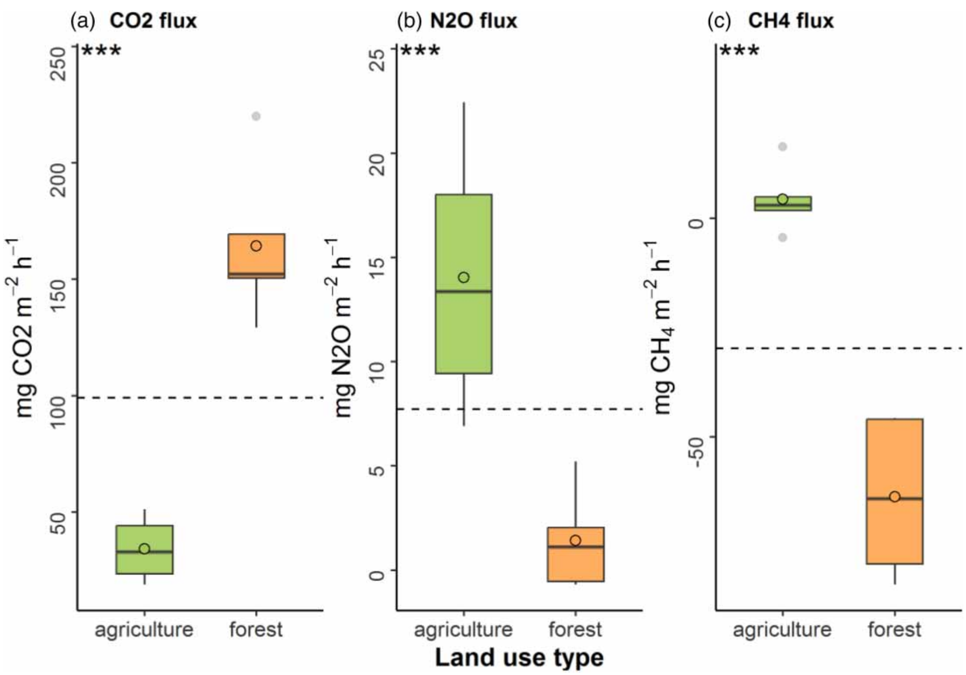


Figure 2 | Variations of greenhouse gas fluxes ((a) CO_2, (b) N_2O, and (c) CH_4) in two land use types (agriculture and forest). Solid lines in the boxplots represent medians, circles represent the mean, and dashed lines represent the mean values of both land use types. The placement of asterisks above the boxplot indicates the level of significance obtained using the Wilcoxon test. Three asterisks (***) represent the differences in GHG fluxes between agriculture and forest land uses that are statistically significant at $p < 0.001$.

3.2. Soil physico-chemical properties

3.2.1. Variations of soil physico-chemical properties

Agricultural soils showed significantly higher ECEC, pH, bulk density, base saturation, WFPS, plant-available P, and microbial biomass C compared to forest soils (Table 2, all $p < 0.05$). However, the concentrations of total C, NH_4^+, and microbial C:N ratio were significantly higher in forest soils than in agricultural soils (Table 2, all $p < 0.05$). Although forest soils showed higher total N than agricultural soils, the difference in concentration of total N between these two land use types was not significant (Table 2, $p > 0.05$). Likewise, microbial biomass C was higher in agricultural soils than in forest soils, but it was not significantly different between these two land uses (Table 2, $p > 0.05$).

3.2.2. Correlation of soil physico-chemical properties in respective land use type

3.2.2.1. Agriculture. All three GHG fluxes in agricultural soils showed significant correlations with all physico-chemical variables, except ECEC (Figure 3, Table 3, all $p < 0.001$). For instance, the CO_2 flux in agricultural soils was significantly positively correlated with NH_4^+ ($r = 0.73$), soil temperature ($r = 0.73$), pH ($r = 0.55$), SOC ($r = 0.95$), total N ($r = 0.91$), plant-available P ($r = 0.94$), WFPS ($r = 0.85$), and microbial biomass N ($r = 0.81$), but significantly negatively correlated with bulk density ($r = -0.93$) and microbial biomass C ($r = -0.69$) (Figure 3, Table 3, all $p < 0.001$). Similarly, the N_2O flux in agricultural soils showed significant positive correlations with NH_4^+ ($r = 0.63$), soil temperature ($r = 0.81$), pH ($r = 0.66$), SOC ($r = 0.96$), total N ($r = 0.87$), plant-available P ($r = 0.99$), WFPS ($r = 0.79$), and microbial biomass N ($r = 0.85$), and significant negative correlations with bulk density ($r = -0.96$) and microbial biomass C ($r = -0.75$) (Figure 3, Table 3, all $p < 0.001$). However, compared with CO_2 and N_2O fluxes, the CH_4 flux in agricultural soils showed opposite interactions with all physico-chemical variables (Figure 3). For instance, the CH_4 flux in agricultural soils was significantly negatively correlated with NH_4^+ ($r = -0.86$), soil temperature ($r = -0.95$), pH ($r = -0.42$), SOC ($r = -0.78$), total N ($r = -0.64$), plant-available P ($r = -0.95$), WFPS ($r = -0.94$), and microbial biomass N ($r = -0.59$), and significantly positive correlated with bulk density ($r = 0.95$) and microbial biomass C ($r = 0.81$) (Figure 3, Table 3, all $p < 0.05$).

3.2.2.2. Forest. The CO_2 flux in forest soil showed significant positive correlations with soil temperature ($r = 0.87$), microbial biomass C ($r = 0.50$), and microbial biomass N ($r = 0.46$) (Figure 4, Table 4, all $p < 0.05$). Although bulk density, ECEC, and WFPS showed a negative correlation with the CO_2 flux in forest soils, the relationships were not significant (all $p > 0.05$, Figure 4, Table 4). The N_2O flux in forest soils was significantly negatively correlated with bulk density ($r = -0.77$, $p < 0.05$) and pH ($r = -0.62$, $p < 0.01$). The positive correlations of N_2O flux with soil temperature, SOC, total N, ECEC, plant-available P, WFPS, microbial biomass C, and microbial biomass N were observed, but the relationships were not

Table 2 | Soil physico-chemical properties in agricultural and forest soils

	Land use type	
Soil characteristics	**Agricultural**	**Forest**
ECEC (mmol$_c$ m^{-2})	28,450.64 ± 576.25a	3,719.18 ± 589.67a
Total C (kg C m^{-2})	846.79 ± 48.79a	1,485.22 ± 236.05a
Total N (kg N m^{-2})	82.02 ± 2.52b	92.15 ± 8.07b
pH	8.49 ± 0.03a	4.96 ± 0.15a
Bulk density (g cm^{-3})	1.33 ± 0.045a	0.95 ± 0.08a
Base saturation (%)	99.93 ± 0.01a	69.14 ± 5.255a
Water-filled pore space (%)	63.56 ± 2.70a	17.52 ± 0.72a
Plant-available P (mg P m^{-2})	5,010.55 ± 343.16a	330.55 ± 98.19a
Microbial biomass C (mg C m^{-2})	32,226.73 ± 185.745b	25,456.56 ± 2,391.19b
Microbial biomass N (mg N m^{-2})	3,515.99 ± 66.76a	2273.36 ± 289.97a
Mineral N in NO_3	239.0795 ± 30.72a	0.00000a
NH_4^+	45.9345 ± 5.22a	85.19 ± 4.21a
Microbial C:N ratio	9.18 ± 0.55a	11.38 ± 0.49a

'b' stands for no significant difference and 'a' stands for significant difference (at $p < 0.05$) of the respective parameters (t test).

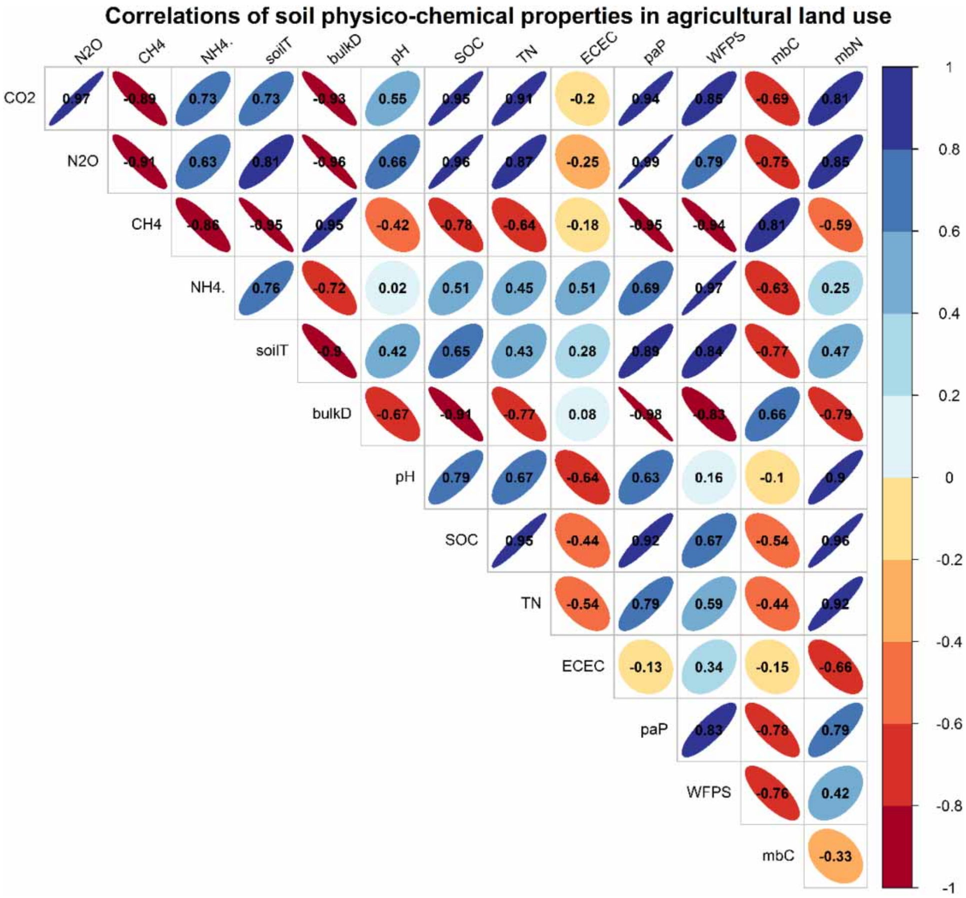


Figure 3 | Correlation matrix for *Pearson*'s correlation coefficients between soil GHG fluxes and physico-chemical properties in agricultural land use, including CO_2, N_2O, CH_4, NH_4^+, soil temperature (soilT), bulk density (bulkD), pH, total nitrogen (TN), soil organic carbon (SOC), effective cation exchange capacity (ECEC), plant-available phosphorous (paP), water-filled pore space (WFPS), microbial biomass nitrogen (mbN), and microbial biomass carbon (mbC). The values of correlation coefficients are represented in the matrix by the ellipses in the corresponding rows and columns. Positive correlations are represented by the color blue, whereas negative correlations are represented by the color red. Pearson's correlation coefficients are shown by the intensity of the color (darker color denotes higher correlation coefficients) and the size of the ellipse (smaller area reflects higher correlation coefficients). The legend on the right side of the correlogram shows *Pearson's* correlation coefficients with their corresponding colors. Table 3 contains the *p*-values for each of the tests.

significant (all $p > 0.05$, Figure 4, Table 4). The CH_4 flux in forest soil was significantly positively correlated with NH_4^+ ($r = 0.58$), soil temperature ($r = 0.79$), bulk density ($r = 0.62$), plant-available P ($r = 0.70$), microbial biomass C ($r = 0.85$), and microbial biomass P ($r = 0.89$), but significantly negatively correlated with SOC ($r = -0.82$), total N ($r = -0.73$), and ECEC ($r = -0.94$) (Figure 4, Table 4, all $p < 0.01$). Apart from CO_2, N_2O, and CH_4 fluxes, soil temperature showed significant negative correlations with SOC ($r = -0.34$), total N ($r = -0.17$), and ECEC ($r = -0.62$), and significant positive correlations with bulk density ($r = 0.32$), pH ($r = 0.57$), plant-available P ($r = 0.35$), microbial biomass C ($r = 0.55$), and microbial biomass N ($r = 0.57$) (Figure 4, Table 4, all $p < 0.05$).

4. DISCUSSION

4.1. GHG fluxes in agricultural and forest soils

The laboratory measurements of GHG fluxes were made from the collected samples of agricultural and forest lands. The results of this study show that there are significant differences in GHG fluxes between two land use types, with forest soils acting as a sink for CH_4 and having higher CO_2 fluxes compared to agricultural soils. The average CO_2 flux in forest soils was more than 4 times higher than in agricultural soils, indicating that forest soils have a higher rate of CO_2 production. For CO_2 fluxes, the pattern seen is that forests have higher concentrations than agricultural lands, which is consistent

Table 3 | The *p*-values of *Pearson's* coefficients between GHG flux and soil physico-chemical variables in agricultural land use

	N_2O	CH_4	NH_4^+	soilT	bulkD	pH	SOC	TN	ECEC	paP	WFPS	mbC	mbN
CO_2	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	*ns*	<0.001	<0.001	<0.001	<0.001
	N_2O	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	*ns*	<0.001	<0.001	<0.001	<0.001
		CH_4	<0.001	<0.001	<0.001	<0.05	<0.001	<0.001	*ns*	<0.001	<0.001	<0.001	<0.01
			NH_4^+	<0.001	<0.001	<0.05	<0.01	<0.01	*ns*	<0.001	<0.001	<0.001	<0.05
				soilT	<0.001	<0.05	<0.001	<0.001	*ns*	<0.001	<0.001	<0.001	<0.01
					bulkD	<0.01	<0.001	<0.001	*ns*	<0.001	<0.001	<0.001	<0.001
						pH	<0.001	<0.001	*ns*	<0.001	<0.05	<0.01	<0.001
							SOC	<0.001	*ns*	<0.001	<0.001	<0.001	<0.001
								TN	*ns*	<0.001	<0.01	<0.01	<0.001
									ECEC	*ns*	*ns*	*ns*	*ns*
										paP	<0.001	<0.001	<0.001
											WFPS	<0.001	<0.01
												mbC	<0.01

The symbol '*ns*' represents that *Pearson*'s correlation between the variables is insignificant. Abbreviations are given in Figure 3.

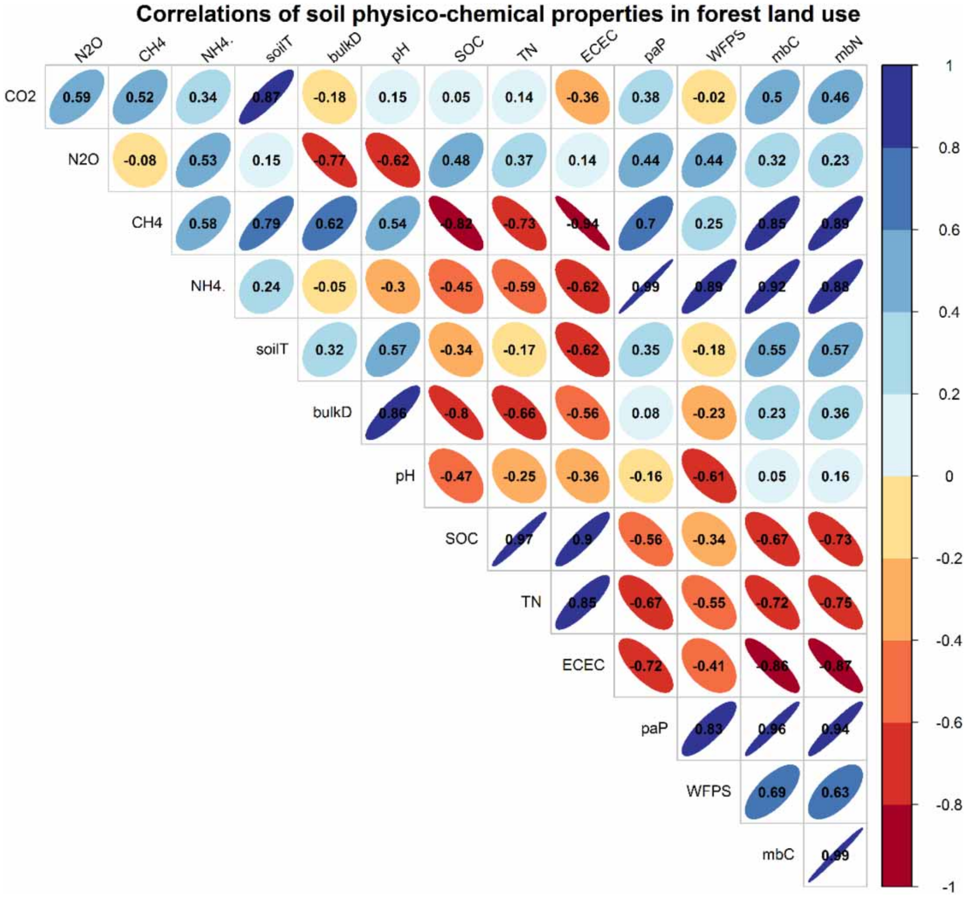


Figure 4 | The same as Figure 3, but for the correlation matrix of *Pearson*'s correlation coefficients between soil GHG fluxes and physico-chemical properties in forest land use. The *p*-values are given in Table 4.

Table 4 | The *p*-values of *Pearson's* coefficients between GHG flux and soil physico-chemical variables in forest land use

	N_2O	CH_4	NH_4^+	soilT	bulkD	pH	SOC	TN	ECEC	paP	WFPS	mbC	mbN
CO_2	*ns*	*ns*	*ns*	<0.05	*ns*	*ns*	*ns*	*ns*	*ns*	*ns*	*ns*	<0.05	<0.05
	N_2O	*ns*	*ns*	*ns*	<0.05	<0.01	*ns*	*ns*	*ns*	*ns*	*ns*	*ns*	*ns*
		CH_4	<0.01	<0.001	<0.01	*ns*	<0.001	<0.001	<0.001	<0.01	*ns*	<0.001	<0.001
			NH_4^+	*ns*	*ns*	*ns*	<0.05	<0.01	<0.01	<0.001	<0.001	<0.001	<0.001
				soilT	<0.01	<0.05	<0.001	<0.05	<0.01	<0.05	*ns*	<0.05	<0.01
					bulkD	<0.01	<0.001	<0.01	<0.01	*ns*	*ns*	<0.05	<0.05
						pH	*ns*	*ns*	*ns*	*ns*	*ns*	*ns*	*ns*
							SOC	<0.001	<0.001	<0.01	*ns*	<0.01	<0.001
								TN	<0.001	<0.001	*ns*	<0.001	<0.001
									ECEC	<0.01	*ns*	<0.001	<0.001
										paP	<0.001	<0.001	<0.001
											WFPS	<0.01	<0.01
												mbC	<0.001

The symbol '*ns*' represents that *Pearson*'s correlation between the variables is insignificant. Abbreviations are given in Figure 3.

with previous findings (Dass *et al.* 2018; Aronson *et al.* 2019). This is likely due to the higher organic matter content and microbial biomass in forest soils, which can lead to higher rates of decomposition and respiration. The presence of litter, microbiological activity, and root activities all contributed to enhanced soil CO_2 fluxes in the forest (Van Straaten *et al.* 2011). Furthermore, fungi, living roots, and microorganisms all contribute to the increase in soil CO_2 fluxes in a similar manner. As there is no vegetative cover on an agricultural site, there is less microbial activity, which results in lower CO_2 fluxes. Because of the use of heavy machinery, the soil becomes more compact, reducing the amount of oxygen available in agricultural fields. The presence of oxygen has an effect on the process of respiration. The pace of decomposition is affected by a variety of parameters such as soil temperature, pH, soil texture, and water content (Aronson *et al.* 2013; Braun *et al.* 2013; Hossain & Beierkuhnlein 2018; Aronson *et al.* 2019), all of which have an impact on CO_2 fluxes (Dass *et al.* 2018).

Examination of fluxes of N_2O showed that N_2O concentrations were greater in the agricultural soils compared to the forest soils. This could be due to agricultural practices such as tillage, which can increase soil N_2O emissions. The input of fertilizer into agriculture may be one of the contributing factors, and this pattern is consistent with earlier research that has found fertilizer application to be correlated with N_2O fluxes (Jennifer *et al.* 2012). The presence of water in the pore space has an effect on N_2O fluxes as well. In our experiment, we found it within the range of 61.1–75.6%. It is well known that soil microorganisms are responsible for both the generation and consumption of N_2O (Davidson 1991; Aronson *et al.* 2019). The process of heterotrophic nitrification is carried out by bacteria and fungi, with a byproduct created by N_2O as a consequence (Dass *et al.* 2018). The nitrification process is crucial in the release of N_2O into the atmosphere, and it is also connected with the decrease of NO_2 in the environment (Poth & Focht 1985). The negative N_2O fluxes in forest soils indicate that these soils are taking up N_2O from the atmosphere, potentially due to denitrification occurring in anaerobic soil conditions.

The CH_4 fluxes also differed significantly between the land use types, with agricultural soils acting as a source of CH_4 emissions and forest soils acting as a sink. This could be due to the presence of anaerobic microsites in agricultural soils, which can support methanogenic microbial activity and lead to CH_4 emissions. It has been demonstrated in previous investigations that microorganisms create and consume CH_4 in the forest environment (Hashimoto *et al.* 2011; Harris *et al.* 2021). The release of CH_4 into the environment results in oxygen deficiency. Agricultural sites produce more methane than non-agricultural sites, despite the fact that microorganisms consume more of it. Land conversion has resulted in a reduction in methane uptake. In contrast, the high CH_4 uptake in forest soils could be due to the presence of methanotrophic microbial communities that can consume CH_4 from the atmosphere. High levels of CH_4 absorption in the forest are associated with high porosity and low bulk density in the forest. Methane is a significant trace gas because of its effects on global warming (Veldkamp *et al.* 2013), and as a result, the effects of methane on land use changes must be taken into consideration.

These findings highlight the importance of considering land use type in GHG mitigation strategies. Forest conservation and reforestation efforts could play a significant role in mitigating GHG emissions by acting as sinks for CO_2 and CH_4, while reducing agricultural practices that contribute to N_2O emissions could also help mitigate GHG emissions. Further research could investigate the underlying factors that drive GHG fluxes in different land use types, such as soil physico-chemical properties and microbial communities, to better inform land use management and mitigation strategies.

The differential findings of GHG fluxes in agricultural and forest soils necessitate the examination of variations in soil physico-chemical properties for several reasons. Firstly, the physico-chemical properties of soil, such as nutrient availability, soil moisture, and organic matter content, can influence the activity of soil microorganisms, which in turn affect GHG fluxes. For instance, nitrogen fertilization in agricultural soils can stimulate the activity of denitrifying bacteria, leading to higher N_2O emissions. Secondly, land use practices, such as tillage, crop rotation, and grazing, can alter soil physico-chemical properties and GHG fluxes. For example, forest soils tend to have higher porosity and lower bulk density than agricultural soils, which can enhance CH_4 uptake and reduce CO_2 emissions. Finally, climate and weather conditions, such as temperature and precipitation, can also influence both soil physico-chemical properties and GHG fluxes. For instance, higher temperatures can increase the activity of soil microorganisms, leading to higher GHG emissions. Overall, these complex and interconnected factors highlight the need for a comprehensive understanding of the variations of soil physico-chemical properties in different land use types to develop effective strategies for mitigating climate change.

4.2. Variations of soil physico-chemical properties between two land use types

The results of this study show that there are significant differences in soil properties between agricultural and forest soils (Table 2). Agricultural soils had significantly higher values for ECEC, pH, bulk density, base saturation, WFPS, plant-available P, and microbial biomass C compared to forest soils. These differences can be attributed to the differences in land use and management practices between these two types of soils. Agricultural soils are typically subjected to more intensive management practices, such as tillage and fertilization, which can lead to higher nutrient availability and microbial biomass. Organic matter is exposed in agricultural sites as a result of decomposition, and this has an impact on pH as well (Havlin *et al.* 2014). In the case of pH measurements, there are statistically significant differences between two land use types. The alkaline content of agricultural soil is higher, whereas the acidic content of forest soil is lower. Litter and plant residues are the elements that contribute to the high pH levels found in the forest. pH is regulated by the decomposition of organic matter in plants, microbial activity, and the release of ions, such as cations and anions (Xu *et al.* 2005). Furthermore, the ammonification process results in the production of hydroxyl ions, which causes the pH of the soil to rise (Xu *et al.* 2005). Consequently, it can be concluded that plant organic matter is the most important element influencing pH, which explains why our study discovered a difference between the two sites.

However, the concentration of total C, NH_4^+, and microbial C:N ratio were significantly higher in forest soils than in agricultural soils. This suggests that forest soils have a higher level of organic matter accumulation and lower rates of mineralization compared to agricultural soils. The higher microbial C:N ratio in forest soils may indicate a slower rate of microbial decomposition, which could be due to the lower nutrient availability and lower levels of disturbance in forest soils. Although forest soils showed higher total N than agricultural soils, the difference was not significant. This could be due to the fact that N is a relatively mobile nutrient in soil and can be easily lost through leaching or volatilization. The higher microbial biomass C in agricultural soils compared to forest soils, although not significantly different, could indicate a higher level of microbial activity and nutrient cycling in agricultural soils.

It demonstrates that the bulk density of agricultural soils is higher than that of forest soils. Bulk density plays a critical role in the conductivity of soil carbon stocks (Blecourt *et al.* 2013). Organic matter has been depleted as a result of land use changes, and land that has been devoid of vegetation has become more vulnerable (Hossain 2022). As a result, soil erosion occurs, and the soil structure becomes weaker over time. As a result of the conversion process, the soil's porosity and bulk density rose, as reported by Kuykendall (2008). Agricultural soils become compacted as a result of the use of heavy machinery, resulting in a high bulk density on the agricultural site.

The total carbon in the soil was substantially higher in forest soils than it was in agricultural soils. A similar trend may be seen in the comparison between forests and plantations, with forests having higher carbon stocks than crop plantations (Blecourt *et al.* 2013). When comparing carbon stocks between agriculture and forest, the pattern is identical (Manning *et al.* 2015; Hörtnagl *et al.* 2018; Raturi *et al.* 2022). The presence of high carbon concentrations in the forest may be due to the presence of a low pH, as acidity increases, carbon concentrations in the site increase as well (Hörtnagl *et al.* 2018). When comparing two land use types that were relatively close to each other, the difference in nitrogen stocks was reported to be slightly higher for forest sites compared to agricultural sites.

It also demonstrates that soil microbial biomass carbon and nitrogen concentrations were higher in agricultural sites than in forest sites, respectively. However, because of the high root respiration and microbial activity in forests, it is expected that microbial biomass carbon will be higher in forests (Kemmitt *et al.* 2005). The ratio of microbial carbon to nitrogen was lower in the agricultural location. There is a relationship between the C:N ratio and the change in pH. The C:N ratio is growing, followed by acidity (Kemmitt *et al.* 2005). Previous studies found that the amount of ions rose as a result of the increase in pH (Hörtnagl *et al.* 2018; Harris *et al.* 2021).

Overall, these findings suggest that land use and management practices can have a significant impact on soil properties and nutrient cycling. Agricultural practices can lead to higher nutrient availability and microbial biomass, but also higher soil disturbance and potential for nutrient loss. In contrast, forest soils have a higher level of organic matter accumulation and lower nutrient availability, but also lower rates of disturbance and potential for nutrient loss. Understanding these differences in soil properties between land use types can help inform land management practices and nutrient management strategies to optimize soil health and productivity.

4.3. Relationships between GHG fluxes and physico-chemical properties

In this study, we found that all three GHG fluxes in agricultural soils showed significant correlations with all physio-chemical variables while they were significantly negatively correlated with bulk density, and microbial biomass C. These findings suggest that the physico-chemical properties of the soil play an important role in GHG fluxes from agricultural soils. This result is consistent with Aronson *et al.* (2019), which claimed that in addition to CH_4, CO_2, and N_2O fluxes, there are strong negative relationships between soil temperature and SOC and total N, and positive relationships of soil temperature with bulk density, microbial biomass C, pH, plant-available P, and microbial biomass N. Interestingly, there was a positive correlation between soil moisture and respiration. The negative correlation between GHG fluxes and bulk density suggests that compacted soils may reduce gas diffusion and increase the anaerobic conditions that favor GHG production. Similarly, the negative correlation between GHG fluxes and microbial biomass C suggests that microbial activity may play a role in GHG emissions from agricultural soils. CH_4 intake grew with increasing soil moisture content during dry conditions (Aronson *et al.* 2019). The maximum CO_2 generation was reported in depressions due primarily to their more favorable soil water conditions (Braun *et al.* 2013). Specifically, the resistance of grasslands to climate warming, droughts, and fire, together with the preference for carbon capture below ground, helps to protect terrestrial carbon sequestration. In contrast, California forests looked incapable of adapting to uncontrolled global climate change, converting from large carbon sinks to carbon sources by at least the middle of the 21st century (Dass *et al.* 2018). The results demonstrated that CO_2, N_2O, and CH_4 fluxes differed significantly between different sites. The soil GHG fluxes were substantially influenced by daily precipitation and soil temperature. Along a gradient of altitude, rainfall and soil temperature are positively correlated with soil GHG fluxes (Fatumah *et al.* 2019).

Our result showed that the CO_2 flux in forest soil showed strong positive correlations with soil temperature, microbial biomass N, and microbial biomass C, while ECEC, bulk density, and WFPS showed a negative correlation with the CO_2 flux in forest soils. This suggests that higher soil temperatures and greater microbial biomass may lead to increased CO_2 emissions from forest soils. This result is consistent with recent study findings (Carrillo *et al.* 2022), which reported that drought stress enhances CO_2 emission through increasing root decomposition in an experimental site in Australia. The N_2O flux in forest soils was negatively correlated with bulk density and pH. The positive correlations of N_2O flux with soil temperature, SOC, total N, ECEC, plant-available P, WFPS, microbial biomass N, and microbial biomass C were found. These findings are consistent with the findings of Sgouridis & Ullah (2017), which reported that the overall N_2O fluxes in the intensive grasslands were almost 40 times higher than those in the peatlands and were positively correlated with most of the soil physico-chemical properties. The amount that denitrification contributed to net N_2O emissions varied depending on the kind of land use and ranged anywhere from 9 to 60%. They also contended that the moisture content of the soil (e.g., aridity in grasslands: Hossain & Li 2023) was the most important factor in determining how N_2O sources were distributed. In contrast, Yan *et al.* (2018) found that a rise in precipitation resulted in a substantial increase in N_2O emissions (+154.0%) and CO_2 fluxes (+112.2%) while simultaneously resulting in a significant drop in CH_4 absorption (41.4%). With the exception of CO_2, there was no evidence of any geographical variation in the fluxes of GHGs. Positive connections were found between soil moisture and all GHG fluxes, with the exception of N_2O (Raturi *et al.* 2022). A significant increase in CO_2 and CH_4 fluxes was caused by an increase in surface temperature, as well as an increase in the moisture content of soil and manure. CH_4 fluxes were shown to be linked to all of the parameters, although the association with TN was by far the highest. It is interesting to note that the amount of N_2O fluxes had a weak but substantial inverse correlation with the amount of moisture present, total organic carbon, total nitrogen, microbial biomass C, and microbial biomass N (Liu *et al.* 2017). This finding has important implications for understanding the drivers of N_2O emissions from soils and developing strategies to mitigate these emissions. For example, management practices that increase soil moisture or add organic matter to the soil may help reduce N_2O emissions. Similarly, practices that promote microbial activity, such as adding nitrogen fertilizer or using cover crops, may also help reduce N_2O emissions.

Overall, these results highlight the importance of considering multiple GHG fluxes and soil physico-chemical properties when developing strategies to mitigate GHG emissions from land use practices. They also emphasize the need for continued research to better understand the complex interactions between soil and ecosystem processes and to inform more effective land management and policy decisions that promote both productivity and environmental sustainability. Practitioners should consider soil properties and management practices, such as reducing soil compaction and fertilizer application, to mitigate GHG emissions from land use practices.

5. CONCLUSION

Agricultural and forest soils are both sources and sinks of GHGs, and they have a considerable impact on the global budget of GHGs. Using observation data from the agricultural and forest soils, we estimated GHG fluxes, physico-chemical properties, and their relationships in the respective land use types in Gottingen, Germany. Higher CO_2 flux in forest soils compared to agricultural soils highlights that the presence of litter, living roots, and more microbial activities have enhanced soil CO_2 flux in forest floors. Conversely, higher N_2O flux in agricultural soils than that in forest soils suggests that the application of fertilizer may be the root cause of higher N_2O flux in agricultural soils, along with the heterotrophic nitrification process. Forest soils have been found to sink CH_4 as CH_4 fluxes in forest soils are negative, which is associated with higher porosity and lower bulk density in forest soils. The high porosity and low bulk density indicate high O_2 in the soils, thus leading to high activity of methanooxidizing bacteria in the forest soils. Agricultural soils showed a slight CH_4 source, as the CH_4 flux was positive in this land use. Irrespective of land use types, all three GHGs showed a differential correlation with soil physico-chemical properties. For instance, CO_2 and N_2O fluxes in agricultural soils were significantly positively correlated with NH_4^+, soil temperature, pH, SOC, total N, plant-available P, WFPS, and microbial biomass N, while negatively correlated with bulk density and microbial biomass C. Unlike CO_2 and N_2O fluxes, CH_4 flux in agricultural soils showed opposite interactions with all physico-chemical variables. For forest soils, CO_2 and CH_4 fluxes showed positive relationships with soil temperature, microbial biomass C, and microbial biomass N, and N_2O flux was negatively correlated with bulk density and pH. This study implies that differences in soil physico-chemical properties and levels of microbial activity are the primary causes of the observed higher CO_2 and CH_4 but lower N_2O in forest soils compared to agricultural soils. Our study highlights the importance of considering land use type and soil properties in developing effective management practices to reduce GHG emissions. For example, reducing soil compaction and fertilizer application in agricultural soils and promoting high porosity and lower bulk density in forest soils can help reduce GHG emissions. This study provides valuable insights into the complex interactions between soil physico-chemical properties and GHG fluxes in forest and agricultural soils and can guide policy decisions to promote sustainable land management practices.

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AUTHOR CONTRIBUTIONS

Conceptualization: A.R., M.L.H., M.H.K.; Methodology: A.R., M.L.H.; Software: M.L.H., J.L.; Validation: A.R., M.L.H.; Visualization: M.L.H., M.H.K., M.M.A.S.; Formal analysis and investigation: M.L.H., J.L.; Writing – original draft preparation: A.R., M.L.H.; Writing – review and editing: M.L.H., M.H.K., M.M.H.S., M.M.A.S., J.L.; Resources: A.R., M.M.H.S.; Supervision: M.L.H., J.L., Funding acquisition: A.R., J.L.

DATA AVAILABILITY STATEMENT

All relevant data are included in the paper or its Supplementary Information.

CONFLICT OF INTEREST

The authors declare there is no conflict.

REFERENCES

Aronson, E. L., Goulden, M. L. & Allison, S. D. 2019 Greenhouse gas fluxes under drought and nitrogen addition in a Southern California grassland. *Soil Biology and Biochemistry* **131**, 19–27.

Barthel, M., Bauters, M., Baumgartner, S., Drake, T. W., Bey, N. M., Bush, G., Boeckx, P., Botefa, C. I., Dériaz, N., Ekamba, G. L., Gallarotti, N., Mbayu, F. M., Mugula, J. K., Makelele, I. A., Mbongo, C. E., Mohn, J., Mandea, J. Z., Mpambi, D. M., Ntaboba, L. C., Rukeza, M. B., Spencer, R. G. M., Summerauer, L., Vanlauwe, B., Oost, K. V., Wolf, B. & Six, J. 2022 Low N_2O and variable CH_4 fluxes from tropical forest soils of the Congo Basin. *Nature Communications* **13**, 330.

Blake, G. R. & Hartge, K. H., 1986 Bulk density. In: *Methods of Soil Analysis, Part 1. Physical and Mineralogical Methods* (Klute, A., ed.). Agronomy Monograph, Soil Science Society of America, Madison, WI, USA, p. 12.

Blankinship, J. C., Brown, J. R., Dijkstra, P. & Hungate, B. A. 2010 Effects of interactive global changes on methane uptake in an annual grassland. *Journal of Geophysical Research* **115**, G02008.

Blécourt de, M., Brumme, R., Xu, J., Corre, M. D. & Veldkamp, E. 2013 Soil carbon stocks decrease following conversion of secondary forests to rubber (*Hevea brasiliensis*) plantations. *PLoS ONE* **8** (7), e69357.

Braun, M., Bai, Y., McConkey, B., Farrell, R., Romo, J. T. & Pennock, D. 2013 Greenhouse gas flux in a temperate grassland as affected by landform and disturbance. *Landscape Ecology* **28** (4), 709–723.

Brookes, P. C., Landman, A., Pruden, G. & Jenkinson, D. S. 1985 Chloroform fumigation and the release of soil nitrogen: a rapid direct extraction method to measure microbial biomass nitrogen in soil. *Soil Biology and Biochemistry* **17**, 837–842.

Burgan, H. I. & Aksoy, H. 2022 Daily flow duration curve model for ungauged intermittent subbasins of gauged rivers. *Journal of Hydrology* **604**, 127249.

Busman, N. A., Melling, L., Goh, K. J., Imran, Y., Sangok, F. E. & Watanabe, A. 2023 Soil CO_2 and CH_4 fluxes from different forest types in tropical peat swamp forest. *Science of the Total Environment* **858**, 159973.

Carrillo, Y., Tissue, D. T., Bruna, S., Maier, C. & Dijkstra, F. A. 2022 Drought impacts on tree root traits are linked to their decomposability and net carbon release. *Frontiers in Forests and Global Change* **5**, 836062.

Chang, J., Ciais, P., Gasser, T., Smith, P., Herrero, M., Havlík, P., Obersteiner, M., Guenet, B., Goll, D. S., Li, W. & Naipal, V. 2021 Climate warming from managed grasslands cancels the cooling effect of carbon sinks in sparsely grazed and natural grasslands. *Nature Communications* **12**, 118.

Chen, Z., Chen, F., Zhang, H. & Liu, S. 2016 Effects of nitrogen application rates on net annual global warming potential and greenhouse gas intensity in double-rice cropping systems of the Southern China. *Environmental Science and Pollution Research* **23**, 24781–24795.

Corre, M. D., Pennock, D. J., Van Kessel, C. & Elliott, D. K. 1999 Estimation of annual nitrous oxide emissions from a transitional grassland-forest region in Saskatchewan, Canada. *Biogeochemistry* **44**, 29–49.

Das, A. C., Shahriar, S. A., Chowdhury, M. A., Hossain, M. L., Mahmud, S., Tusar, M. K., Ahmed, R. & Salam, M. A. 2023 Assessment of remote sensing-based indices for drought monitoring in the North-western region of Bangladesh. *Heliyon* **9** (2), E13016.

Dass, P., Houlton, B. Z., Wang, Y. & Warlind, D. 2018 Grasslands may be more reliable carbon sinks than forests in California. *Environmental Research Letters* **13** (7), 074027.

Davidson, E. A. 1991 *Microbial Production and Consumption of Greenhouse Gases: Methane, Nitrogen Oxides and Halomethanes*. American Society of Microbiology, Washington, DC, pp. 219–235.

Drewer, J., Kuling, H. J., Cowan, N. J., Majalap, N., Sentian, J. & Skiba, U. 2021 Comparing soil nitrous oxide and methane fluxes from oil palm plantations and adjacent riparian forests in Malaysian Borneo. *Frontiers in Forests and Global Change* **4**, 738303.

Eberwein, J. R., Oikawa, P. Y., Allsman, L. A. & Jenerette, G. D. 2015 Carbon availability regulates soil respiration response to nitrogen and temperature. *Soil Biology and Biochemistry* **88**, 158–164.

Ekeleme, A. C., Ekwueme, B. N. & Agunwamba, J. C. 2021 Modeling contaminant transport of nitrate in soil column. *Emerging Science Journal* **5** (4), 471–485.

Fatumah, N., Munishi, L. K. & Ndakidemi, P. A. 2019 Variations in greenhouse gas fluxes in response to short-term changes in weather variables at three elevation ranges, Wakiso District, Uganda. *Atmosphere* **10**, 1–17.

Feng, S., Gu, X., Luo, S., Liu, R., Gulakhmadov, A., Slater, L. J., Li, J., Zhang, X. & Kong, D. 2022 Greenhouse gas emissions drive global dryland expansion but not spatial patterns of change in aridification. *Journal of Climate* **35** (20), 2901–2917.

Frank, A. B., Liebig, M. A. & Hanson, J. D. 2002 Soil carbon dioxide fluxes in northern semiarid grasslands. *Soil Biology and Biochemistry* **34**, 1235–1241.

Genxu, W., Ju, Q., Guodong, C. & Yuanmin, L. 2002 Soil organic carbon pool of grassland soils on the Qinghai-Tibetan Plateau and its global implication. *Science of the Total Environment* **291** (1–3), 207–217.

Harris, D., Horwáth, W. R. & Van Kessel, C. 2001 Acid fumigation of soils to remove carbonates prior to total organic carbon or carbon-13 isotopic analysis. *Soil Science Society of America Journal* **65** (6), 1853–1856.

Harris, N. L., Gibbs, D. A., Baccini, A., Birdsey, R. A., De Bruin, S., Farina, M., Fatoyinbo, L., Hansen, M. C., Herold, M., Houghton, R. A., Potapov, P. V., Suarez, D. R., Roman-Cuesta, R. M., Saatchi, S. S., Slay, C. M., Turubanova, S. A. & Tyukavina, A. 2021 Global maps of twenty-first century forest carbon fluxes. *Nature Climate Change* **11**, 234–240.

Hashimoto, S., Morishita, T., Sakata, T. & Ishizuka, S. 2011 Increasing trends of soil greenhouse gas fluxes in Japanese forests from 1980 to 2009. *Scientific Reports* **1**, 116.

Havlin, J. L., Beaton, J. D., Tisdale, S. L. & Nelson, W. L. 2014 *Soil Fertility and Fertilizers: An Introduction to Nutrient Management*, 6th edn. Prentice Hall, Upper Saddle River, NJ.

Hörtnagl, L., Barthel, M., Buchmann, N., Eugster, W., Butterbach-Bahl, K., Díaz-Pinés, E., Zeeman, M., Klumpp, K., Kiese, R., Bahn, M. & Hammerle, A. 2018 Greenhouse gas fluxes over managed grasslands in Central Europe. *Global Change Biology* **24** (5), 1843–1872.

Hossain, M. L. 2022 *Grassland Ecosystems Functioning and Stability in Response to Climatic Variability and Climate Extremes*. PhD Dissertation. Hong Kong Baptist University. Available from: https://scholars.hkbu.edu.hk/ws/portalfiles/portal/59830046/G22THFL-032373T.pdf (accessed 27 February 2023).

Hossain, M. L. & Beierkuhnlein, C. 2018 Enhanced aboveground biomass by increased precipitation in a central European grassland. *Ecological Processes* **7**, 37.

Hossain, M. L. & Li, J. 2020 Effects of long-term climatic variability and harvest frequency on grassland productivity across five ecoregions. *Global Ecology and Conservation* **23**, e01154.

Hossain, M. L. & Li, J. 2021a Disentangling the effects of climatic variability and climate extremes on the belowground biomass of C_3- and C_4-dominated grasslands across five ecoregions. *Science of the Total Environment* **760**, 143894.

Hossain, M. L. & Li, J. 2021b Biomass partitioning of C_3- and C_4-dominated grasslands in response to climatic variability and climate extremes. *Environmental Research Letters* **16**, 074016.

Hossain, M. L. & Li, J. 2021c NDVI-based vegetation dynamics and its resistance and resilience to different intensities of climatic events. *Global Ecology and Conservation* **30**, e01768.

Hossain, M. L. & Li, J. 2023 Asymmetric response of above- and below ground biomass of C_3- and C_4-dominated grasslands to aridity. *Journal of Water and Climate Change* **14** (7), 2465–2478.

Hossain, M. L., Kabir, M. H., Nila, M. U. S. & Rubaiyat, A. 2021 Response of grassland net primary productivity to dry and wet climatic events in four grassland types in Inner Mongolia. *Plant-Environment Interactions* **2**, 250–262.

Hossain, M. L., Li, J., Hoffmann, S. & Beierkuhnlein, C. 2022 Biodiversity showed positive effects on resistance but mixed effects on resilience to climatic extremes in a long-term grassland experiment. *Science of the Total Environment* **827**, 154322.

Hossain, M. L., Li, J., Hoffmann, S. & Beierkuhnlein, C. 2023a Divergence of ecosystem functioning and stability under climatic extremes in a 24-year long-term grassland experiment. In *EGU General Assembly 2023*, 24–28 Apr 2023, Vienna, Austria. EGU23-17067.

Hossain, M. L., Li, J., Lai, Y. & Beierkuhnlein, C. 2023b Long-term evidence of differential resistance and resilience of grassland ecosystems to extreme climate events. *Environmental Monitoring and Assessment* **195**, 734.

IPCC 2013 Climate change 2013: the physical science basis. In: *Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K. B., Tignor, M. & Miller, H. L., eds.). Cambridge University Press, Cambridge and New York, NY, pp. 996.

IPCC 2021 Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change (Masson-Delmotte, V., Zhai, P., Pirani, A., Connors, S. L., Péan, C., Berger, S.,Caud, N., Chen, Y., Goldfarb, L., Gomis, M. I., Huang, M., Leitzell, K., Lonnoy, E., Matthews, J. B. R., Maycock, T. K., Waterfield, T., Yelekçi, O., Yu, R. & Zhou, B., eds.). Cambridge University Press, Cambridge, UK.

Izaurralde, R. C., Lemke, R. L., Goddard, T. W., McConkey, B. & Zhang, Z. 2004 Nitrous oxide emissions from agricultural toposequences in Alberta and Saskatchewan. *Soil Science Society of America Journal* **68** (4), 1285–1294.

Jabal, Z. K., Khayyun, T. S. & Alwan, I. A. 2022 Impact of climate change on crops productivity using MODIS-NDVI time series. *Civil Engineering Journal* **8** (6), 1136–1156.

Jennifer, L. M., Marcelo, A. & Emilys, B. 2012 Greenhouse gas fluxes in southeastern U.S. coastal plain wetlands under contrasting land uses. *Ecological Applications* **22** (1), 264–280.

Jin, F., Yang, H. & Zhao, Q. 2000 Progress in the research of organic carbon storage. *Soil* **32** (1), 11–17.

Kemmitt, S., Wright, D. & Jones, D. 2005 Soil acidification used as management strategy to reduce nitrate losses from agricultural land. *Soil Biology and Biochemistry* **37**, 867–875.

Kuykendall, H. 2008 *Soil quality physical indicators: Selecting dynamic soil properties to assess soil function*. United States Department of Agriculture, NRCS, Soil Quality National Technology Development Team, Washington, DC.

LeCain, D. R., Morgan, J. A., Schuman, G. E., Reeder, J. D. & Hart, R. H. 2000 Carbon exchange rates in grazed and ungrazed pastures of Wyoming. *Journal of Range Management* **53**, 199–206.

Līcīte, I., Popluga, D., Rivža, P., Lazdiņš, A. & Meļņiks, R. 2022 Nutrient-rich organic soil management patterns in light of climate change policy. *Civil Engineering Journal* **8** (10), 2290–2304.

Liebig, M. A., Kronberg, S. L. & Gross, J. R. 2008 Effects of normal and altered cattle urine on short-term greenhouse gas flux from mixed-grass prairie in the Northern Great Plains. *Agriculture, Ecosystems & Environment* **125**, 57–64.

Liu, Y., Yan, C., Matthew, C., Wood, B. & Hou, F. 2017 Key sources and seasonal dynamics of greenhouse gas fluxes from yak grazing systems on the Qinghai-Tibetan Plateau. *Scientific Reports* **7**, 1–11.

Luyssaert, S., Jammet, M., Stoy, P. C., Estel, S., Pongratz, J., Ceschia, E., Churkina, G., Don, A., Erb, K. H., Ferlicoq, M., Gielen, B., Grunwald, T., Houghton, R. A., Klumpp, K., Knohl, A., Kolb, T., Kuemmerle, T., Laurila, T., Lohila, A., Loustau, D., McGrath, M. J., Meyfroidt, P., Moors, E. J., Naudts, K., Novick, K., Otto, J., Pilegaard, K., Pio, C. A., Rambal, S., Rebmann, C., Ryder, J., Suyker, A. E., Varlagin, A., Wattenbach, M. & Dolman, A. J. 2014 Land management and land-cover change have impacts of similar magnitude on surface temperature. *Nature Climate Change* **4**, 389–393.

Manning, P., de Vries, F. T., Tallowin, J. R., Smith, R., Mortimer, S. R., Pilgrim, E. S., Harrison, K. A., Wright, D. G., Quirk, H., Benson, J., Shipley, B., Cornelissen, J. H. C., Kattge, J., Bonisch, G., Wirth, C. & Bardgett, R. D. 2015 Simple measures of climate, soil properties and plant traits predict national-scale grassland soil carbon stocks. *Journal of Applied Ecology* **52** (5), 1188–1196.

Manzoni, S., Jackson, R. B., Trofymow, J. A. & Porporato, A. 2008 The global stoichiometry of litter nitrogen mineralization. *Science* **321**, 684–686.

Moriasi, D. N., Arnold, J. G., Van Liew, M. W., Bingner, R. L., Harmel, R. D. & Veith, T. L. 2007 Model evaluation guidelines for systematic quantification of accuracy in watershed simulations. *Transactions of the ASABE* **50** (3), 885–900.

Nila, M. U. S., Beierkuhnlein, C., Jaeschke, A., Hoffmann, S. & Hossain, M. L. 2019 Predicting the effectiveness of protected areas of Natura 2000 under climate change. *Ecological Processes* **8**, 13.

Nyborg, M., Laidlaw, J. W., Solberg, E. D. & Malhi, S. S. 1997 Denitrification and nitrous oxide emissions from soil during spring thaw in a Malmo loam, Alberta. *Canadian Journal of Soil Science* **77**, 153–160.

Raturi, A., Singh, H., Kumar, P., Chanda, A. & Shukla, N. 2022 Characterizing the post-monsoon CO_2, CH_4, N_2O, and H_2O vapor fluxes from a tropical wetland in the Himalayan foothill. *Environmental Monitoring and Assessment* **194**, 50.

R Core Team. 2020 *R: A Language and Environment for Statistical Computing*. R Foundation for Statistical Computing, Vienna. Available from: www.R-project.org/.

Rütting, T., Björsne, A.-K., Weslien, P., Kasimir, Å. & Klemedtsson, L. 2021 Low nitrous oxide emissions in a boreal spruce forest soil, despite long-term fertilization. *Frontiers in Forests and Global Change* **4**, 710574.

Schlesinger, W. H. & Andrews, J. A. 2000 Soil respiration and the global carbon cycle. *Biogeochemistry* **48**, 7–20.

Sgouridis, F. & Ullah, S. 2017 Soil greenhouse gas fluxes, environmental controls, and the partitioning of N_2O sources in UK natural and seminatural land use types. *Journal of Geophysical Research: Biogeosciences* **122**, 2617–2633.

Smith, N. E., Kooijmans, L. M. J., Koren, G., Schaik, E. V., Woude, A. M. V. D., Wanders, N., Ramonet, M., Xueref-Remy, I., Siebicke, L., Manca, G., Brümmer, C., Baker, I. T., Haynes, K. D., Luijkx, I. T. & Peters, W. 2020 Spring enhancement and summer reduction in carbon uptake during the 2018 drought in northwestern Europe. *Philosophical Transactions of the Royal Society B* **375**, 20190509.

Soosaar, K., Schindler, T., Machacova, K., Pärn, J., Fachín-Malaverri, L. M., Rengifo-Marin, J. E., Alegría-Muñoz, W., Jibaja-Aspajo, J. L., Negron-Juarez, R., Zárate-Gómez, R., Garay-Dinis, D. J., Arista-Oversluijs, A. G., Tello-Espinoza, R., Pacheco-Gómez, T. & Mander, Ü. 2022 High methane emission from palm stems and nitrous oxide emission from the soil in a Peruvian Amazon peat swamp forest. *Frontiers in Forests and Global Change* **5**, 849186.

Tellez-Rio, A., Vallejo, A., García-Marco, S., Martin-Lammerding, D., Tenorio, J. L., Rees, R. M. & Guardia, G. 2017 Conservation agriculture practices reduce the global warming potential of rainfed low N input semi-arid agriculture. *European Journal of Agronomy* **84**, 95–104.

van Groenigen, K. J., Osenberg, C. W. & Hungate, B. A. 2011 Increased soil emissions of potent greenhouse gases under increased atmospheric CO_2. *Nature* **475**, 214–218.

Van Straaten, O., Veldkamp, E. & Corre, M. D. 2011 Simulated drought reduces soil CO_2 efflux and production in a tropical forest in Sulawesi, Indonesia. *Ecosphere* **2** (10), 119.

Veldkamp, E., Koehler, B. & Corre, M. D. 2013 Indications of nitrogen-limited methane uptake in tropical forest soils. *Biogeosciences* **10**, 5367–5379.

Wang, F. L. & Bettany, J. R. 1995 Methane emissions from a usually well-drained prairie soil after snowmelt and precipitation. *Canadian Journal of Soil Science* **75**, 239–241.

Wilcoxon, F. 1945 Individual comparisons by ranking methods. *Biometrics* **1**, 80–83.

Wolters, S. 1999 Spät- und postglaziale Vegetationsentwicklung im Bereich der Fercher Berge südwestlich von Potsdam. *Gleditschia* **27** (1-2), 25–44.

Xu, J. M., Tang, C. & Chen, Z. L. 2005 Chemical composition controls residue decomposition in soils differing in initial pH. *Soil Biology and Biochemistry* **38** (3), 544–552.

Yan, G., Mu, C., Xing, Y. & Wang, Q. 2018 Responses and mechanisms of soil greenhouse gas fluxes to changes in precipitation intensity and duration: a meta-analysis for a global perspective. *Canadian Journal of Soil Science* **98**, 591–603.

Zhang, Y., Marx, E., Williams, S., Gurung, R., Ogle, S., Horton, R., Bader, D. & Paustian, K. 2020 Adaptation in U.S. *Corn Belt increases resistance to soil carbon loss with climate change. Scientific Reports* **10** (1), 13799.

Zhao, J., Chi, J. & Jocher, G. 2023 Editorial: greenhouse gas fluxes in forest ecosystems. *Frontiers in Forests and Global Change* **6**, 1200668.

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