Abstract  Owing to their high carbon and nitrogen contents, biogas residues may lead to higher carbon dioxide (CO₂) and nitrous oxide (N₂O) emissions from soils. Acidification of biogas slurry and application of nitrification inhibitors (NIs) could mitigate the emission of these gases. An incubation experiment was therefore carried out to investigate the effect of NIs, DMPP (3, 4-dimethylpyrazole phosphate), and PIADIN (active ingredients: 3.00–3.25% 1,2,4-triazole and 1.50–1.65% 3-methylpyrazole), on CO₂ and N₂O emissions from soils fertilized with biogas residues and acidified biogas residues. Biogas residues produced higher ammonium-nitrogen (NH₄⁺-N) and nitrate-nitrogen (NO₃⁻-N) concentrations in soils which resulted in higher emissions of CO₂-C and N₂O-N than that from acidified biogas residues. Both DMPP and PIADIN significantly decreased the emissions of CO₂-C (8.1–55.8%) and N₂O-N (87–98%) and maintained lower NH₄⁺-N and NO₃⁻-N concentrations when compared to control (without nitrification inhibitors). However, the DMPP had a higher reduction capability for CO₂-C emissions than PIA-DIN in acidified biogas residue applied soil. In conclusion, the acidification of biogas residues and application of NIs are effect in reducing gaseous emission from biogas residue fertilized soils and thus could improve the fertilizer effectiveness of the residues.

Keywords  Acidified biogas residues · Nitrification inhibitors · N₂O and CO₂ emissions · Nutrient cycling

1 Introduction

Carbon dioxide (CO₂) and nitrous oxide (N₂O) are the primary greenhouse gases (GHGs) present in the Earth’s atmosphere (IPCC, 2010). CO₂ could hang around for a long time, between 300 and 1000 years, once it is added to the atmosphere (Alan, 2019). The lifetime of atmospheric CO₂ concentration, with an increase of about 120 ppm over the past 250 years, has risen to a current global average of approximately 409 ppm, and future rapid increase is expected, with values likely to reach 550 ppm by mid-century and 1000 ppm by the end of this century (IPCC, 2014). Similarly, N₂O is a long-lived GHG, has a lifetime of 116 ± 9 years (Prather et al., 2015), and is a major stratospheric ozone-depleting substance (Thompson et al., 2019). Its concentration in the atmosphere has also risen steadily since the mid-twentieth century (IPCC, 2013), from approximately 290 ppb in 1940 to 330 ppb in 2017 (Park et al., 2012).
Agricultural practices, particularly the use of nitrogenous fertilizers, substantially contribute to enhancing N₂O emission and thus increasing the concentration of reactive nitrogen (N, N₂O) in the atmospheric environment (Bouwman et al., 2013). In soils, N₂O is produced as a by-product of nitrification and denitrification processes which are carried out by different types of microbes (Bremner, 1997). Nitrification is the biological oxidation of NH₄⁺ to nitrite (NO₂⁻) and further to nitrate (NO₃⁻). Denitrification is a microbially facilitated process in which NO₃⁻ is reduced, through a series of intermediate gaseous N oxide products, to molecular nitrogen (N₂) (Köster et al., 2013). To meet the ambitious climate change adaptations, CO₂ and N₂O emissions should be minimized (IPPC, 2018).

Various techniques are used to reduce the CO₂ and N₂O emissions while meeting the growing demand for food and other agricultural products (Thompson et al., 2019). One of the approaches is to produce biogas through fermentation of renewable sources, including organic manures, plant materials, food waste, etc. The use of biogas as an energy source could reduce dependency on fossil fuels and is expected to have no or even a positive effect on the atmospheric greenhouse gas balance (Herrmann, 2013). Biogas production plays an important role in the European bio-energy supply and has increased rapidly in Germany in recent years. According to the Agency of Renewable Energies, there are 837 biogas production factories in the northern part of Germany. It is evident that the recycling of the biogas residue (BR) from such a large number of biogas production factories installed in the state of Schleswig–Holstein became a problem. Mostly BR is applied to soil as organic fertilizers (Köster et al., 2014; Koszel & Lorencowicz, 2015) and is considered an essential component of cropping systems due to its high fertilization value (Möller & Stinner, 2009). The application of BR as a fertilizer improved soil fertility, plants quality, and their immunity to biotic and abiotic stress agents (Kouřimská et al., 2012).

In addition to improving soil quality, the BR application to soil, due to their high carbon and ammonium content, may increase the CO₂ and N₂O emissions and NO₃⁻ leaching from the soils (Hennig & Gawor, 2012). N₂O emissions and NO₃⁻ leaching from BR may be minimized by their acidification before soil application and by using nitrification inhibitors (NIs). While application of NIs can further reduce NO₃⁻ leaching from BR applied soils by suppressing autotrophic nitrification and subsequent denitrification, which will ultimately also reduce N₂O emission. In fertilizer (mineral, slurry, and organic manure) applied to soil, the use of NIs has shown reduced N₂O emissions (VanderZaag et al., 2011) and thereby increased N use efficiency of the applied fertilizers (Subbarao et al., 2006).

We hypothesized that (a) application of ABR to the soil will decrease soil pH and show lower CO₂ and N₂O emissions than the application of BR and (b) NI application to ABR-amended soil will further lower the gaseous emission from the soil. Keeping this in view, an incubation pot experiment was conducted to investigate CO₂ and N₂O emissions as well as NH₄⁺ and NO₃⁻ dynamics in the soils amended with BR and ABR and applied with DMPP and PIADIN nitrification inhibitors.

2 Material and Methods

2.1 Collection and Preparation of Soil and Biogas Residues

The biogas residues (BR) were collected from a large commercial biogas company in Germany in May 2019. Its digesters were used to feed with the following materials: 18-t corn silage, 55-t dry chicken feces, 4-t whole crop silage (rye), and 7 m³ swine manure. For the incubation experiment, BR was sampled directly from the biogas residue storage pool. Acidified biogas residues (ABR) were prepared by lowering the pH of BR from 7.9 to 5.5 with the addition of H₂SO₄.

The soil was collected from an upper 20-cm layer of a natural grassland (which is adjacent to an agricultural farm and its soil has the same characters as that of agricultural soil) in Grevenkrag, Schleswig–Holstein (54° 11’ 09.1” N and 10° 00’ 36.6” E). The soil was sandy in texture, with 5% silt and 95% sand. The water-holding capacity (WHC) of the soil was 24.34% and bulk density was 1.4 g cm⁻³. The soil was air-dried and sieved through a 2-mm sieve to remove visible plant residues, roots, and stones. The total soil C and N contents in the soil were determined using a CN analyzer (Flash EA™ 1112, Thermo Fischer Scientific, Springe
Waltham, Massachusetts, USA). The fresh BR was got analyzed from Raiffeisen Laborservice (Ormont, Germany) for the salient characteristics. The salient characteristics of the soil and fresh BR are given in Table 1.

2.2 Treatments and Experimental Design

Three treatments of biogas residues (unamended, BR and ABR) were tested against three treatments of nitrification inhibitors (control, DMPP (3, 4-dimethylpyrazole phosphate), and PIADIN (active ingredients: 3.00–3.25% 1,2,4-triazole and 1.50–1.65% 3-methylpyrazole)), yielding 9 treatment combinations in total. The soil was filled in cylindrical pots (15-cm diameter and 33-cm length) up to 20-cm height while adjusting the bulk density to 1.4 g cm\(^{-3}\). Both BR and ABR were applied to the respective pots at the rate of 27.8 g kg\(^{-1}\) soil (equivalent to 0.1 g NH\(_4^+\)-N kg\(^{-1}\) soil), whereas unamended treatment did not receive any amendment. The DMPP and PIADIN were applied at the rate of 5 mg kg\(^{-1}\) soil (5% of the applied NH\(_4^+\)-N), whereas no NI was added to the control pots. The pots were incubated for 57 days in a climatic chamber at a constant temperature (15 °C), soil moisture (80% WHC), and air humidity (50%). Deionized water was added daily to maintain the desired soil moisture level. The experiment followed a two-factorial completely randomized design with four replicates.

2.3 Collection and Measurement of N\(_2\)O and CO\(_2\)

The N\(_2\)O and CO\(_2\) samples were collected daily during 1st week, once after 2 days during the 2nd week, and once after 3 days during rest of the incubation period. Before collecting gas samples, the pots were tightly closed with air-tight lids having one rubber membrane, which served as a contact between the sample collecting syringe and the incubated environment. The samples were collected every 0, 20, 40, and 60 min. A 10-ml syringe with a hypodermic needle was placed in the pre-evacuated 2-ml headspace of Chromacol glass vials to collect the gas samples. The glass vials had a chloro-butyl rubber septum. Each gas sampling was carried out between 09:00 and 11:00 am. Except for the times when samples were being taken, the pots were left open. The concentrations of CO\(_2\) and N\(_2\)O in the gas samples were measured by gas chromatography (Agilent 7890A GC, Agilent, CA, United States). An electron capture detector (ECD), adjusted to a temperature of 300 °C with N\(_2\) as a carrier gas, was used to measure N\(_2\)O concentration. A thermal conductivity detector (TCD), adjusted to a temperature of 250 °C with He as a carrier gas, was used to measure CO\(_2\) concentration (Guo et al., 2021a). For each gas measurement, the gas chromatograph was calibrated with respective certified gas standards. The rate of CO\(_2\) and N\(_2\)O emissions from each pot (ppm/min) during lid closure was calculated using headspace volume and a linear relation between the CO\(_2\) and N\(_2\)O concentration and time (Venterea et al., 2020). The emission rates of CO\(_2\)-C (μg h\(^{-1}\) kg\(^{-1}\)) and of N\(_2\)O-N (ng h\(^{-1}\) kg\(^{-1}\)) were calculated with the following equations:

\[
EN_{2O} = \frac{R \times 60 \times V_{gas} \times AR}{W_{soil} \times V_m} \times 2 \times 1000 \quad (1)
\]

\[
ECO_{2} = \frac{R \times 60 \times V_{gas} \times AR}{W_{soil} \times V_m} \quad (2)
\]

where ECO\(_2\) and EN\(_2O\) are emission rates of CO\(_2\)-C (μg h\(^{-1}\) kg\(^{-1}\)) and N\(_2\)O-N (ng h\(^{-1}\) kg\(^{-1}\)), respectively; R is the rate of CO\(_2\) and N\(_2\)O emissions from each pot (ppm/min); V\(_{gas}\) is the gas volume in pot (L); W\(_{soil}\) is the weight of dry soil in pot (kg); AR is the relative atomic mass of C and N, i.e., 12 and 14, respectively;

Table 1 Characteristics of experimental soil and fresh biogas residues

| Characteristic          | Soil | Biogas residues |
|------------------------|------|-----------------|
| Water content (%)      | –    | 90.3            |
| Total carbon (C, g kg\(^{-1}\)) | 11.6 | 31.4            |
| Total nitrogen (N, g kg\(^{-1}\)) | 0.8  | 5.6             |
| Organic N (g kg\(^{-1}\)) | 0.7  | 2.0             |
| NH\(_4^+\)-N (mg kg\(^{-1}\)) | 2.16 | 3600            |
| NO\(_3^-\)-N (mg kg\(^{-1}\)) | 4.8  | 0               |
| pH\(^a\)              | 6.5  | 7.9             |

\(^a\)In soil:CaCl\(_2\) (1:4) extract
and $V_m$ is the molar volume of gas which is 23.7 L/mol at 15 °C.

Total $N_2O$ and $CO_2$ emissions during the incubation period were calculated by adding the total daily $N_2O$ and $CO_2$ emissions. For this purpose, the emission rates of $CO_2$-C (μg h$^{-1}$ kg$^{-1}$) and $N_2O$-N (ng h$^{-1}$ kg$^{-1}$) were multiplied by the hours, i.e., 24 h for the 1st week, 48 h for the 2nd week, and 72 h for the last 6 weeks. The submission of all these gave the total emissions of $CO_2$ (mg kg$^{-1}$) and $N_2O$ (μg kg$^{-1}$).

2.4 Soil Sampling and Measurement of $NH_4^+$, $NO_3^-$, and pH

For the measurement of $NH_4^+$-N, $NO_3^-$-N, and pH, soil samples (each 50 g approx.) were collected up to 20-cm depth using a specialized soil agar on days 1, 15, 29, 43, and 57 of incubation. Each soil sample was divided into two subsamples; the first was oven-dried at 105 °C for 8 h to calculate the water content, while the other was used for the determination of $NH_4^+$-N, $NO_3^-$-N and pH. For soil mineral N analysis, 10-g fresh soil was extracted with 40 mL of 0.0125 M CaCl$_2$ solution (1:4) for 1 h on a reciprocating shaker. The suspensions so obtained were centrifuged for 10 min, filtered through Whatman filter paper No. 42, and stored at 4 °C. The concentrations of $NH_4^+$ and $NO_3^-$ were measured by a continuous flow autoanalyzer (San++ Automated Wet Chemistry Analyzer—Continuous Flow Analyzer (CFA), Skalar, The Netherlands). For pH measurement, 10-g air-dried and sieved soil was mixed with 25 mL of 0.0125 M CaCl$_2$ solution (1:2.5) and shaken for 1 h. After centrifugation of the suspensions, the pH of the upper clear liquid was measured using a pH meter.

2.5 Statistical Analysis

Data were verified for normal distribution, treatment means for total $N_2O$-N, and $CO_2$-C emissions over the incubation period were compared using a two-way analysis of variance. pH over the different treatment at the same day was compared using a two-way analysis of variance. The significance of differences between individual treatment means was determined using Tukey’s honestly significant difference (HSD) test at $P \leq 0.05$. Tukey’s HSD test was performed by R statistical software (Oakland, CA, USA). R was used to create the artwork.

3 Results

3.1 Rate of $CO_2$-C and $N_2O$-N Emissions

In unamended soil, the rates of $CO_2$-C and $N_2O$-N emissions were substantially lower than the biogas residues amended soils and there was no consistent pattern of increase or decrease with the passage of time during the incubation period (Figs. 1 and 2). However, rates of both $CO_2$-C and $N_2O$-N emissions were substantially higher in BR- and ABR-amended soils than unamended soil, with BR application showing more increase than ABR application. In BR- and ABR-amended soils, rates of emissions reached maximum by the end of the first or mid of the 2nd week, and thereafter decreased gradually, reaching the minimum value at the end of the incubation period. Application of DMPP and PIADIN to BR- and
ABR-amended soils decreased the emissions rates of both gases.

3.2 Total CO$_2$-C and N$_2$O-N Emissions

Irrespective of whether NIs were applied or not, total CO$_2$-C emission was very low from unamended soil than that from BR- and ABR-amended soils. The total CO$_2$-C emission was the highest from BR-amended soil (169 mg kg$^{-1}$ soil), followed by ABR-amended soil (81 mg kg$^{-1}$ soil) and the lowest from unamended soil (37 mg kg$^{-1}$ soil) (Table 2). Compared to unamended soil, the application of BR and ABR increased total CO$_2$-C emission by 3.60- and 1.21-fold, respectively. Total N$_2$O-N emission was also significantly higher from BR- and ABR-amended soils than that from unamended soil (Table 2). The mean total N$_2$O-N emission was the highest from BR-amended soil (3678 μg kg$^{-1}$), followed by ABR-amended soil (1464 μg kg$^{-1}$), and the lowest from unamended soil (5 μg kg$^{-1}$). Compared to unamended soil, the application of BR and ABR increased the total N$_2$O-N emission by 73- and 29-folds, respectively.

ABR-amended soil showed half CO$_2$-C emission of what did the BR-amended soil (Table 2). The NIs (DMPP and PIADIN) significantly lowered total CO$_2$-C emission compared to control, but their relative effectiveness depended upon the residue types (Table 2). In BR-amended soil, DMPP and PIADIN were equally effective in reducing CO$_2$-C emission, with a reduction of 55% over control. However, in ABR-amended soil, DMPP was more effective in reducing CO$_2$-C emission, showing a 38% decrease over control, than PIADIN which showed only an 8% decrease. Overall, the CO$_2$-C emission-reducing capacity of both the NIs was higher in BR-amended soil than that in ABR-amended soil.

ABR-amended soil showed 60% lower total N$_2$O-N emission than BR-amended soil (Table 3). The application of NIs further much lowered the total N$_2$O-N emission compared to control, with DMPP having higher reduction efficiency than PIADIN under both BR- and ABR-amended soils. The mean reduction in total N$_2$O-N emission in BR- and ABR-amended soil by DMPP and PIDIN was 96 and 90%, respectively.

3.3 Soil NH$_4^+$-N and NO$_3^-$-N Concentrations

The application of both BR and ABR substantially increased NH$_4^+$-N and NO$_3^-$-N concentrations.
compared to control (Fig. 3). With the passage of time, ABR application showed a slow gradual decline in \(\text{NH}_4^+\) concentration and a rise in \(\text{NO}_3^-\) concentration as compared to BR-amended soil. Similarly, the residue-amended soils applied with NIs showed a gradual decline in \(\text{NH}_4^+\)-N concentration and rise in \(\text{NO}_3^-\)-N concentration, whereas these changes were quite sharp in control soils. Moreover, DMPP more strongly retarded the fall in \(\text{NH}_4^+\)-N concentration and rise in \(\text{NO}_3^-\)-N concentration than PIADIN in both BR- and ABR-amended soils. The peak \(\text{NO}_3^-\)-N concentrations were lower in NI-treated soils as compared to control and DMPP produced the lowest \(\text{NO}_3^-\)-N concentration.

Table 3: Total \(\text{N}_2\text{O}-\text{N}\) emission (mg kg\(^{-1}\) soil) from unamended, biogas residue (BR)–, and acidified biogas residue (ABR)–amended soils applied with control, DMPP, and PIADIN nitrification inhibitors

| NI     | Unamended soil | BR-amended soil | ABR-amended soil |
|--------|----------------|-----------------|-----------------|
| Control| 5.04 ± 0.35e    | 3678 ± 64a      | 1464 ± 72b      |
| DMPP   | 8.48 ± 0.53 (+ 68.3)e | 84 ± 4 (- 98)d  | 87 ± 6 (- 94)d  |
| PIADIN | 5.31 ± 0.24 (+ 5.4)e | 233 ± 28 (- 94)c | 187 ± 15 (- 87)c |

Values (mean ± SE, \(n=4\)) followed by different letters are significantly different at \(P=0.05\). The values in parenthesis represent % decrease (–) or increase (+) over the respective controls.

![Fig. 3](image)

Fig. 3: Time-dependent change in \(\text{NH}_4^+\)-N and \(\text{NO}_3^-\)-N concentrations in unamended, biogas residue (ABR)– and acidified biogas residue (ABR)–amended soils applied with control, DMPP, and PIADIN nitrification inhibitors under controlled environment. The vertical bars indicate the standard errors of the means \((n=4)\)
3.4 Soil pH

The application of BR did not affect the soil pH throughout the course of the experiment (Fig. 4). However, ABR application lowered the soil pH as compared to unamended soil and the mean decrease during the incubation period was 0.34 units. The application of DMPP and PIADIN had no influence on the soil pH.

4 Discussion

4.1 Effect of Biogas Residues

The application of biogas residues to agricultural soils as an organic fertilizer has become a common practice and is further growing with the increase in the number of biogas plants in some European countries (Köster et al., 2015; Wolf et al., 2014). The application of BR no doubt improves soil organic matter and N availability to crops, but on other sites, it can also provoke CO₂ and N₂O emissions from soils (Köster et al., 2011, 2015; Senbayram et al., 2014). Thus, high application rates of BR to cultivated lands could be linked to environmental problems such as high levels of CO₂ and N₂O in the atmosphere and NO₃⁻ leaching to groundwaters (Qu et al., 2014). This study revealed that BR application to soil significantly increased CO₂ and N₂O emissions (Figs. 1 and 2; Tables 2 and 3). A sharp decline in NH₄⁺-N concentration and a rise in NO₃⁻-N concentration was recorded in BR-amended soil (Fig. 3). Concurrent presence of NH₄⁺, NO₃⁻, and high level of labile C in the BR could have hastened the nitrification and denitrification processes in the soil (Jaeger et al., 2013) and thus increased CO₂ emissions (Köster et al., 2015) and N₂O emissions (Senbayram et al., 2009) compared with unamended soil. The availability of a high level of NO₃⁻ as a result of nitrification together with labile C under anaerobic soil conditions serves as a driving force for N₂O emissions (Senbayram et al., 2009). Groffman and Crawford (2003) reported that CO₂ efflux and denitrification activity can be positively correlated, and hence, increased denitrification activity also increases CO₂ efflux. It is generally accepted that the application of organic fertilizers stimulates soil microbial biomass, basal respiration (Ros et al., 2003), and enzyme activities (Chu et al., 2007). Soil microbial processes also produce gases such as N₂O (IPCC, 2013), and residual organic carbon substrates favor soil microbial denitrification (Robertson & Groffman, 2007). In conclusion, BR application to soil significantly increases CO₂-C and N₂O-N emissions from soils.

4.2 Acidification of BR and the Gaseous Emission

Soil N dynamics in BR-amended soils could depend on the BR composition and stability (Alburquerque et al., 2012). Hence, through changing the soil characteristics, BR may affect the nitrification process and hence the amounts of NO₃⁻ lost by denitrification or leaching (Peter et al., 2004). Among the soil characteristics, pH is a master variable that affect microbial transformations in soils. Nitrification is strongly influenced by pH (Mackens et al., 2021), with maximum...
rates occurring at pH 7.5 (Eric et al., 2002). Oxidation of NH$_4^+$ is completely inhibited at pH 5 and increases with a higher pH (Wang et al., 2018). Our result showed that application of ABR lowered soil pH than unamended soil, but this pH effect was not observed for BR-amended soil (Fig. 4). Accordingly, ABR application resulted in half CO$_2$-C and 60% less N$_2$O-N emissions from the soil compared to BR application (Tables 2 and 3). This is well explained by a slow gradual decline in NH$_4^+$ concentration and a rise in NO$_3^-$ in ABR-amended soil as compared to BR-amended soil (Fig. 3). Thus, it could be inferred that ABR application retarded the nitrification process through a pH-driven shift in the microbial community structure and/or microbial activities (Ottosen et al., 2009). Fangueiro et al. (2013) also reported that lowering the pH led to decreased CO$_2$ emission from soil, and the effect is due to the low microbial activities in the acidified soil (Fangueiro et al., 2015). In addition, since a small amount of H$_2$SO$_4$ could lower the pH of BR from 7.9 to 5.5, the cost of acidification is very low. In conclusion, ABR application to soil significantly decreases CO$_2$-C and N$_2$O-N emissions from soils compared to BR.

4.3 Nitrification Inhibitors and the Gaseous Emissions

Application of NIs is recognized as one of the mitigation strategies that have been proven to be highly effective in reducing N fertilizer losses, increasing N use efficiency, and crop yields under different cropping systems (Moir et al., 2012; Zhang et al., 2015). Therefore, we tested the compatibility of using the most common NIs DMPP and PIADIN with BR and ABR for reducing CO$_2$ and N$_2$O emission from soils. Our result showed that NIs reduced CO$_2$ and N$_2$O emissions in BR- and ABR-amended soil (Figs. 1 and 2; Tables 2 and 3). For reducing CO$_2$ emission, the efficiency of NIs was relatively less in ABR-amended soil than BR-amended soil and DMPP was more effective than PIADIN only in ABR-amended soil (Table 2). However, NIs were equally effective in reducing N$_2$O emission in BR- and ABR-amended soil, and DMPP was relatively more effective than PIADIN under both conditions (Table 3). The application of PIADIN has been reported to reduce N$_2$O emissions by 37–62% during the weeks following biogas residue application to soil (Wolf et al., 2014). In fact, DMPP and PIADIN maintained NH$_4^+$-N concentration at a higher level which showed that NIs retarded the nitrification process and kept NO$_3^-$-N concentration at a low level (Fig. 3). Most NIs retard microbial oxidation of NH$_4^+$ by depressing the activity of nitrifiers in soil (Wolf et al., 2014). Thus, NIs could have inhibited nitrification through suppressing the activity of ammonia-oxidizing bacteria or relevant enzymes, effectively delaying the oxidation process that transforms NH$_4^+$ into NO$_3^-$.

5 Conclusion

Application of both BR and ABR substantially increased soil NH$_4^+$ content, but ABR-amended soil showed much lower CO$_2$-C and N$_2$O-N emissions than BR-amended soil. DMPP and PIADIN were equally effective in reducing CO$_2$-C emission from BR-amended soil but DMPP was more effective than PIADIN in ABR-amended soil. DMPP almost completely diminished the N$_2$O emission from both BR- and ABR-amended soils while the efficacy of PIADIN was relatively lower than DMPP in both cases. Acidification of BR did not further improve the efficacy of NIs, rather it had a slightly negative effect on the performance of PIADIN. Thus, it is concluded that although acidification of BR had an ameliorating effect on CO$_2$ and N$_2$O emissions from soils, it is not required when NIs have already been applied to the soils. However, acidification of BR could be beneficial in lowering N$_2$O emission from the soils.
Acknowledgements  Ahmad Khan is thankful to the Alexander von Humboldt Foundation for a re-invitation grant for a research visit at Kiel University.

Author Contributions  YG: methodology, software, writing—original draft; AA: methodology, writing—review and editing; AK: methodology, writing—review and editing; AN: writing—review and editing; KHM: conceptualization, writing—review and editing, funding acquisition, supervision.

Funding  Open Access funding enabled and organized by Projekt DEAL. This work was supported by the funding of the Ph.D. project by the Society of Energy and Climate (EKSH) of Schleswig–Holstein (14/12–24) and the Alexander von Humboldt Foundation through the grant of George Forster Post-Doctorate Fellowship.

Data Availability  The datasets used and/or analyzed during the current study are available from the corresponding author on reasonable request.

Code Availability  Not applicable.

Declarations

Conflict of interest  The authors declare no competing interests.

Ethical Approval  Not applicable.

Consent for Publication  Not applicable.

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