# Grazing alters sandy soil greenhouse gas emissions in a sand-binding area of the Hobq Desert, China postprint

**Authors:** WANG Bo, LI Yuwei, BAO Yuhai

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## Abstract

Deserts are sensitive to environmental changes caused by human interference and are prone to degradation. Revegetation can promote the reversal of desertification and the subsequent formation of fixed sand. However, the effects of grazing, which can cause the ground-surface conditions of fixed sand to further deteriorate and result in re-desertification, on the greenhouse gas (GHG) fluxes from soils remain unknown. Herein, we investigated GHG fluxes in the Hobq Desert, Inner Mongolia Autonomous Region of China, at the mobile (desertified), fixed (vegetated), and grazed (re-desertified) sites from January 2018 to December 2019. We analyzed the response mechanism of GHG fluxes to micrometeorological factors and the variation in global warming potential (GWP). CO2 was emitted at an average rate of 4.2, 3.7, and 1.1 mmol/(m2 • h) and N2O was emitted at an average rate of 0.19, 0.15, and 0.09 μmol/(m2 • h) at the grazed, fixed, and mobile sites, respectively. Mean CH4 consumption was as follows: fixed site (2.9 μmol/(m2 • h))>grazed site (2.7 μmol/(m2 • h))>mobile site (1.1 μmol/(m2 • h)). GHG fluxes varied seasonally, and soil temperature (10 cm) and soil water content (30 cm) were the key micrometeorological factors affecting the fluxes. The changes in the plant and soil characteristics caused by grazing resulted in increased soil CO2 and N2O emissions and decreased CH4 absorption. Grazing also significantly increased the GWP of the soil ($P<0.05$). This study demonstrates that grazing on revegetated sandy soil can cause re-desertification and significantly increase soil carbon and nitrogen leakage. These findings could be used to formulate informed policies on the management and utilization of desert ecosystems.

# Full Text

## Preamble

### Grazing Alters Sandy Soil Greenhouse Gas Emissions in a Sand-Binding Area of the Hobq Desert, China

WANG Bo, LI Yuwei*, BAO Yuhai
College of Geographical Science, Inner Mongolia Normal University, Hohhot 010010, China

**Abstract:** Deserts are highly sensitive to environmental changes caused by human interference and are prone to degradation. Revegetation can promote the reversal of desertification and the subsequent formation of fixed sand. However, the effects of grazing—which can cause ground-surface conditions of fixed sand to deteriorate further and result in re-desertification—on soil greenhouse gas (GHG) fluxes remain unknown. Herein, we investigated GHG fluxes in the Hobq Desert, Inner Mongolia Autonomous Region of China, at mobile (desertified), fixed (vegetated), and grazed (re-desertified) sites from January 2018 to December 2019. We analyzed the response mechanism of GHG fluxes to micrometeorological factors and the variation in global warming potential (GWP). $CO_2$ was emitted at average rates of 4.2, 3.7, and 1.1 mmol/($m^2 \cdot h$), while $N_2O$ was emitted at average rates of 0.19, 0.15, and 0.09 µmol/($m^2 \cdot h$) at the grazed, fixed, and mobile sites, respectively. Mean $CH_4$ consumption followed the order: fixed site (2.9 µmol/($m^2 \cdot h$)) > grazed site (2.7 µmol/($m^2 \cdot h$)) > mobile site (1.1 µmol/($m^2 \cdot h$)). GHG fluxes varied seasonally, and soil temperature (10 cm) and soil water content (30 cm) were the key micrometeorological factors affecting the fluxes. The changes in plant and soil characteristics caused by grazing resulted in increased soil $CO_2$ and $N_2O$ emissions and decreased $CH_4$ absorption. Grazing also significantly increased the soil GWP ($P<0.05$). This study demonstrates that grazing on revegetated sandy soil can cause re-desertification and significantly increase soil carbon and nitrogen leakage. These findings could be used to formulate informed policies on the management and utilization of desert ecosystems.

**Keywords:** grazing; revegetation; re-desertification; greenhouse gases; global warming potential; Hobq Desert

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## 1 Introduction

Soil is a major source of atmospheric greenhouse gases (GHGs) in terrestrial ecosystems and can act as either a source or sink in the global carbon cycle

[?, ?]. Land desertification is a crucial ecological issue that threatens sustainable development, affecting approximately 25% of Earth' s total land area. The fragile vegetation and soil systems in these regions affect the stability of the soil carbon pool, leading to the release of sequestered soil carbon. This increases total GHG emissions—predominantly from industrial fossil fuels—and accelerates global warming [?, ?]. Several studies have focused on soil GHG fluxes in forests, wetlands, agricultural lands, and grassland ecosystems [?, ?, ?]. Scholars have increasingly recognized the importance of desert ecosystems in the carbon cycle, with recent research examining seasonal changes in GHG fluxes [?, ?], environmental and anthropogenic influences [?, ?, ?], and soil respiration components [?, ?] in desert ecosystems. Human intervention can lead to complex changes in fragile sand systems, and while scholars have mainly focused on desertification and vegetation restoration processes [?, ?], the subsequent re-desertification of fixed sandy areas remains poorly understood.

The Hobq Desert is the seventh largest desert in China. The structure and function of its eastern region' s ecosystem have been severely damaged by years of cultivation, grazing, and other human interventions. Since the 21st century, forestry authorities have restored the area through artificial vegetation, gradually fixing the mobile sand. This process developed regional microhabitats, increased vegetation cover, and resulted in ecosystem formation and community succession [?, ?]. Furthermore, soil–vegetation interactions can affect soil physical-chemical traits, root distribution, microbial colonization, and soil fauna activities, promote the development and succession of biological crusts [?, ?], and indirectly change the spatial and temporal patterns of soil GHG fluxes. Previous studies suggest that artificial vegetation substantially alters the soil-vegetation system and water and temperature equilibriums, leading to changed rates of GHG emissions from sandy soil [?, ?, ?].

Grazing in fixed sandy areas with vegetation cover can yield favorable economic benefits. However, environmental heterogeneity, water shortage, and soil infertility are ongoing problems in newly fixed sandy areas. Therefore, the vegetation has limited resistance and resilience to disturbance and is vulnerable to degradation [?, ?]. Grazing on such fragile land could result in a sharp reduction in vegetation cover, fragmentation of surface biocrust, and reappearance of bare sand patches on the land surface, ultimately causing land re-desertification. In typical grasslands, grazing plays an important role in regulating GHG emission and absorption by changing plant community composition [?, ?], reducing litter accumulation and soil carbon storage, and altering the physical and chemical properties of plants and soils. However, unlike typical grasslands, it remains unknown how grazing and subsequent changes in the soil microenvironment affect soil GHG emission patterns in fixed sandy areas.

In this study, we investigated the effects of sandy soil alteration caused by vegetation restoration and grazing on desert ecosystems in a sand-binding area of the Hobq Desert. We selected three sites: mobile (desertified), fixed (vegetated), and grazed (re-desertified). The grazed site had previously been mobile dune sand that was subsequently vegetated (fixed), but re-desertification occurred due to grazing. The background characteristics of landform and soil of the three sites were homologous, but they have been differentiated due to human activity interference, resulting in different landscape features and vegetation types. We aimed to determine the effect of sandy soil changes caused by grazing or revegetation on GHG fluxes, identify seasonal variations in GHG fluxes, and investigate the micrometeorological factors driving changes in GHG fluxes. The results of this study can provide scientific evidence for the rational utilization and management of desert ecosystems in the context of global climate change.

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## 2.1 Study Area and Experimental Site

The study area is located in the southeast of the Hobq Desert, Inner Mongolia Autonomous Region of China, with a total area of approximately 1.0$×10^{4}$ $hm^2$ and an elevation of 1100–1300 m. The region has a temperate continental climate with substantial seasonal variation, characterized by hot, rainy summers and cold, dry winters. The annual average temperature is 8.3°C, and the frost-free period is approximately 135 days. Average annual precipitation is 317 mm, with rainfall mainly occurring from June to September, accounting for 84.4% of the annual total. Mean annual evaporation is 2560 mm, and mean annual wind speed is 3.3 m/s, resulting in strong wind erosion. The soil types are primarily aeolian sandy soil formed by aeolian sand flow deposits. The vegetation is mainly composed of *Salix cheilophila* C.K. Schneid., *Artemisia ordosica* Krasch., and *Caragana korshinskii* Kom.

Human intervention, such as revegetation and grazing, has resulted in dramatic sandy soil changes. Based on a combination of factors—including the degree of desertification, stage of vegetation recovery, status of surface biocrust development, and degree of interference—we established three experimental sites: mobile, fixed, and grazed. The mobile site (40°05 46 N, 110°53 38 E) had mobile, bare sand with low vegetation cover (only a few annual herbs), severe wind erosion, and no biocrust cover. At the fixed site (40°05 01 N, 110°52 49 E), artificial vegetation was established beginning in 2002 by belt-cutting branches of *S. cheilophila* at 2 m$×1mspacing.Followingnaturalsparsityandartificialrejuvenation, astable∗
S.cheilophila∗communityformed, accompaniedby∗A.ordosica∗, withhighvegetationcover, athicklayeroforga
mm, with scattered patches of bare sand appearing on the surface, indicating that re-desertification was occurring.

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### 2.2.1 Variation Characteristics of $CO_2$, $N_2O$, and $CH_4$ Fluxes

Within each of the three sites, we selected areas with flat terrain and relatively consistent habitats for sampling and established five plots. GHG flux testing was conducted over 24 months from January 2018 to December 2019. At each site, we collected one sample at the beginning, middle, and end of each month, resulting in 360 gas collections per sampling site during the test period. Soil gas samples were collected and $CO_2$, $N_2O$, and $CH_4$ concentrations were determined using static chamber-gas chromatography [**?**, **?**]. The static chamber consisted of a cylindrical top box (320 mm$×$600 mm) and a pedestal. The top wall was equipped with a stirring fan to evenly mix gases within the chamber. The base was placed in clearings not covered by the *S. cheilophila* canopy, at least 1 m away from plant bases. We selected all sampling plots according to specific standards to ensure consistency of basic meteorological elements and eliminate systematic errors caused by varying distances from plants. The base was pre-embedded 20 cm into the soil for stability. For sampling, the chamber was placed in the pedestal groove and water was used to seal it. Gas sampling was performed only on sunny days; if conditions were not sunny, sampling was delayed or advanced by one to three days. Sampling lasted 30 minutes and was conducted between 09:00 and 12:00 (local standard time). Gas samples were taken three times at 0, 15, and 30 minutes. Because sampling plots were scattered, we assigned regular personnel at each site to ensure synchronous gas collection across all three sites, eliminating errors caused by time differences. Samples were collected in 100 mL syringes with three-way valves and stored in aluminum foil airbags for transport to the laboratory, where gas concentrations were determined within one week using an Agilent 4890D gas chromatograph (Agilent, Santa Clara, CA, USA).

Diurnal variation in gas flux was observed 11 times in July and August of both 2018 and 2019. Each diurnal observation began at 06:00, with gas collected every 2 hours until 04:00 the following day, yielding 12 gas samples per day.

We calculated GHG fluxes based on changes in gas concentrations over time [**?**, **?**]. Positive values indicate GHG emission from soil, while negative values represent soil absorption of GHGs from the atmosphere. Gas flux was calculated according to the following formula:

$$F = \rho \times \frac{h \times dC/dt \times 273}{273 + T}$$

where $F$ represents the gas flux of $CO_2$ ($\mathrm{mmol/(m^2 \cdot h)}$), $CH_4$ ($\mathrm{\mu mol/(m^2 \cdot h)}$), and $N_2O$ ($\mathrm{\mu mol/(m^2 \cdot h)}$);  is the gas density under standard conditions ($\mathrm{kg/m^3}$); $h$ is the height of the closed static chamber (m); $dC/dt$ is the rate of change in GHG concentration during measurement; and $T$ is the temperature inside the chamber (°C).

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### 2.2.2 Global Warming Potential (GWP)

Global warming potential (GWP) is commonly used to measure the contribution of different GHG types to global warming. GWP is calculated in terms of $CO_2$ equivalent over a 100-year time scale, where the GWP of $CH_4$ and $N_2O$ is 28 and 265 times that of $CO_2$, respectively (i.e., 1 g $CH_4/m^2$ = 28 g $CO_2/m^2$ and 1 g $N_2O/m^2$ = 265 g $CO_2/m^2$) [?, ?].

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## 2.3 Micrometeorological Factors

Micrometeorological factors at the mobile, fixed, and grazed sites were monitored continuously from January 2018 to December 2019. Rainfall was recorded every 30 minutes using a small automatic weather station (HOBO U30-NRC, Onset, Bourne, USA). Photosynthetically active radiation (PAR) was measured using a light sensor (HOBO LIA-M003, Onset, Bourne, USA) at 3 m above ground. Wind speed was measured using a wind sensor (HOBO WSB-M003, Onset, Bourne, USA). Air temperature and humidity were measured using sensors (HOBO S-THB-M002 and S-THB-M008, Onset, Bourne, USA) at heights of 2.5 and 0.1 m above ground, respectively. Soil water content was measured using a soil moisture sensor (TDR100, Campbell Scientific, Logan, Germany) at depths of 10, 20, and 30 cm. Soil temperature was measured using a temperature sensor (TCAV, Campbell Scientific, Logan, Germany) at depths of 10, 20, and 30 cm.

Biomass and soil properties of the three sites were investigated at the plot level in August 2018 (Table 1 ). Three 10 m\$×10*mshrubplotswereestablisheddiagonallyineachstandardlandunit*, *and*
m herbaceous plots were randomly selected within each shrub plot to investigate vegetation community characteristics (plant species, height, coverage, and density). The soil profile was collected from the center of each shrub quadrat, with three soil samples taken at 0–20 cm depth and mixed to form a single composite sample. Samples were air-dried, crushed, and sieved in the laboratory. Soil organic carbon was determined using the external potassium dichromate–sulfuric acid oxidation heating method. Soil total nitrogen content was determined using the Kjeldahl method. Soil total phosphorus content was determined using an elemental analyzer (FP-LIBS, FELLES, Tianjin, China). Soil bulk density was determined using the ring-knife method. The quantity of soil microorganisms (bacteria, actinobacteria, and fungi) was analyzed using an UltraClean DNA Isolation Kit (Mo Bio Laboratories, Solana Beach, CA, USA) to extract total microbial DNA, with amplification performed using a PCR instrument (C1000™ Touch Thermal, Bio-Rad, CA, USA). Microorganism quantity was estimated via fluorescence ratio PCR (CFX96, Bio-Rad, CA, USA).

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## 2.4 Statistical Analysis

Excel and Origin were used for data processing and visualization, while SPSS 20.0 software was used for statistical analysis. One-way analysis of variance (ANOVA) and least significant difference (LSD) tests were used to determine differences in GHG fluxes among the mobile, fixed, and grazed sites (P=0.05). Two-way ANOVA was used to determine the effects of sandy soil changes and season on GHG fluxes. Pearson' s correlation coefficient was used to investigate the effects of micrometeorological factors on GHG fluxes. Redundancy analysis (RDA) was used to analyze the main factors affecting GHG fluxes during the growing season (May-October), non-growing season (November-April of the following year), and the entire study period. A total of 1080 samples of each gas flux were obtained from the three sites during the study period, with 540 samples each in both the growing and non-growing seasons.

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## 3.1 Environmental Variables in the Study Area

The average air temperature in the study area was 8.4°C in 2018 and 8.3°C in 2019. Cumulative precipitation was 333.2 mm in 2018 and 369.6 mm in 2019. Soil temperature (0-20 cm) was 0.2°C lower in 2018 than in 2019, while soil water content was 5.7% higher in 2018 than in 2019 (Fig. 1 [Figure 1: see original paper]). Significant differences in soil temperature and water content were detected among the three sites. Soil temperature at the mobile site was higher than at the fixed and grazed sites by 36.5% and 19.3%, respectively. The mobile site also exhibited the highest soil water content, which was 17.8% and 27.3% higher than that of the fixed and grazed sites, respectively.

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## 3.2 Spatial and Temporal Variations in GHG Fluxes

$CO_2$, $N_2O$, and $CH_4$ fluxes in 2018 and 2019 are shown in Figures 2 and 3. Emissions of $CO_2$ and $N_2O$ were detected at all sites, while $CH_4$ absorption occurred at all locations. However, GHG fluxes varied significantly among sites. The $CO_2$ emission flux followed the order: grazed site ($4.2\ \mathrm{mmol/(m^2 \cdot h)}$) > fixed site ($3.7\ \mathrm{mmol/(m^2 \cdot h)}$) > mobile site ($1.1\ \mathrm{mmol/(m^2 \cdot h)}$). The $N_2O$ emission flux followed the order: grazed site ($0.19\ \mu\mathrm{mol/(m^2 \cdot h)}$) > fixed site ($0.15\ \mu\mathrm{mol/(m^2 \cdot h)}$) > mobile site ($0.09\ \mu\mathrm{mol/(m^2 \cdot h)}$). The $CH_4$ absorption flux followed the order: fixed site ($2.9\ \mu\mathrm{mol/(m^2 \cdot h)}$) > grazed site ($2.7\ \mu\mathrm{mol/(m^2 \cdot h)}$) > mobile site ($1.1\ \mu\mathrm{mol/(m^2 \cdot h)}$).

No significant differences were detected in GHG fluxes between 2018 and 2019, but significant seasonal variations were observed. At all three sites, $CO_2$ emissions were highest in summer and lowest in winter. At the mobile site, $N_2O$ was briefly absorbed during winter. Similar to the other two sites, $N_2O$ emissions were greater in spring and summer, while $CH_4$ absorption was higher in spring and summer but decreased significantly in autumn and winter.

$CO_2$, $N_2O$, and $CH_4$ fluxes exhibited distinct diurnal variation patterns, with all fluxes greater during daytime than nighttime (Fig. 4 [Figure 4: see original paper]). For the mobile, fixed, and grazed sites, $CO_2$ emissions peaked between 10:00 and 12:00. $N_2O$ emissions peaked from 14:00 to 16:00 at the mobile site, from 10:00 to 12:00 at the fixed site, and from 12:00 to 14:00 at the grazed site. $CH_4$ absorption was greatest from 16:00 to 18:00 at the mobile site and from 12:00 to 14:00 at both the fixed and grazed sites.

Two-way ANOVA results revealed that sandy soil changes, season, and their interaction significantly affected $CO_2$ and $CH_4$ fluxes ($P<0.01$; Table 2 ). Partial $^2$ indicated that $CO_2$ flux was more affected by season than by sandy soil changes, whereas the effect of sandy soil changes on $CH_4$ flux was significantly greater than that of season. $N_2O$ flux was not affected by sandy soil changes but exhibited seasonal variations.

## 3.3 Variations in Soil GWP

Significant differences in GHG fluxes among the mobile, fixed, and grazed sites led to varying GWP values. The grazed site exhibited the highest $CO_2$ and $N_2O$ emissions, while the fixed site had the highest $CH_4$ absorption (Fig. 5 [Figure 5: see original paper]). Consequently, the grazed site had the highest GWP (1597.7 g $CO_2/(m^2 \cdot a)$), followed by the fixed site (1401.6 g $CO_2/(m^2 \cdot a)$) and the mobile site (440.8 g $CO_2/(m^2 \cdot a)$).

## 3.4 Correlations Between Micrometeorological Factors and GHG Fluxes

To determine the effects of micrometeorological factors on $CO_2$, $N_2O$, and $CH_4$ fluxes, we performed Pearson' s linear regression analysis (Fig. 6 [Figure 6: see original paper]) and redundancy analysis (Fig. 7 [Figure 7: see original paper]). Linear regression analysis showed that $CO_2$ flux at all sites had significant positive correlations with atmospheric, surface, and soil temperatures at each depth. Soil water content, rainfall, and PAR were also significantly positively correlated with $CO_2$ flux. $N_2O$ flux at the grazed site was significantly positively correlated with rainfall but not with temperature. At the fixed site, $N_2O$ flux was significantly positively correlated with surface soil water content (10 cm) but not with other variables. In contrast, $N_2O$ flux at the mobile site was significantly correlated with temperature, soil water content, and PAR.

At the fixed and grazed sites, $CH_4$ flux was significantly negatively correlated with humidity and atmospheric, surface, and soil temperatures at each depth. $CH_4$ flux at the mobile site was significantly correlated with deep soil water content (30 cm) but not with other variables.

RDA results showed that during the growing season, the cumulative contribution of the first (soil water content at 30 cm depth, SWC-30) and second (surface soil temperature, T-10) principal components was 83.2% (Fig. 7a). The former explained 40.5% of variation and the latter explained 24.5%. The mobile site was distinctly separated along RDA1, showing a positive relationship of fluxes with soil water content and moisture in the lower layer. During the non-growing season, the cumulative contribution of the first (SWC-30) and second (PAR) principal components was 87.6% (Fig. 7b). The former explained 51.4% of variation and the latter explained 13.1%. $CH_4$ flux was significantly and negatively correlated with SWC-30 and PAR, consistent with linear correlation analysis results. The three sites had comparable micrometeorological characteristics.

Over the entire study period, the cumulative contribution of the first (T-10) and second (SWC-30) principal components was 81.3% (Fig. 7c). The former explained 40.9% of variation and the latter explained 14.5%. No separation was observed among the three sites.

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## 4.1 Effects of Revegetation on GHG Fluxes

This study demonstrates that revegetation of a sand-binding area can significantly increase soil $CO_2$ emissions. Revegetation resulted in higher coverage of shrubs and grasses and greater above- and belowground biomass (Table 1 ). The synergistic effects of soil–vegetation systems caused substantial spatial heterogeneity of soil $CO_2$ flux in desert soils, with soil respiration significantly enhanced by vegetation restoration and biocrust development. The primary sources of $CO_2$ emissions from sandy soil are plant root respiration, cryptogam respiration in biocrust, soil microbial respiration, and soil animal respiration [?, ?]. Autotrophic respiration is performed by living plant roots, while dead roots serve as substrates for heterotrophic respiration by soil microorganisms; consequently, root growth improves soil aeration [?, ?]. Increased root biomass therefore increases $CO_2$ emissions. Higher aboveground biomass means greater canopy coverage, which effectively regulates microclimate and concentrates soil resources, forming "fertility islands" under shrubs and creating ideal conditions for biocrust formation and colonization [?, ?]. Castillo-Monroy et al. (2011) reported that biocrust autotrophic respiration was the main contributor to respiration in desert soil, accounting for up to 42% of total respiration. In this study, a mixed crust had developed at the fixed site, explaining why $CO_2$ emissions from the fixed site were significantly greater than those from the mobile site.

Soil–vegetation interactions can directly affect soil factors and thus indirectly influence soil $CO_2$ emissions. Soil organic matter and total nitrogen content were significantly higher at the fixed site (Table 1 ), which also explains the higher $CO_2$ emissions. Soil organic carbon is a substrate for biological growth, and soils with high organic carbon contents generally have higher biological activity and thus higher respiration rates [?, ?]. Nitrogen is an important component of amino acids, proteins, nucleotides, and coenzymes, and can directly affect photorespiration and root cell mitochondrial respiration [?, ?]. Soil bacteria, fungi, and actinomycetes use soil organic matter in heterotrophic respiration, and the higher microbial biomass at the fixed site could also explain the higher $CO_2$ emissions.

The air permeability of sandy soil provides an ideal aerobic environment for nitrifying bacteria, explaining why sandy soil is often a source of $N_2O$ emissions. In this study, the $N_2O$ emission rate at the fixed site was greater than at the mobile site. Vegetation restoration reduces soil compactness, creating an ideal environment for microbial proliferation and thereby increasing autotrophic or heterotrophic nitrification and subsequent $N_2O$ emissions [?, ?]. Additionally, soil organic carbon and nitrogen are essential for nitrification; chemotrophic nitrifying bacteria use soil organic carbon as an energy source and oxidize $NH_4^+$ organic compounds to form $N_2O$ and other N-containing gases [?, ?]. Therefore, the increased soil organic carbon and total nitrogen contents at the fixed site also explain the higher $N_2O$ emissions.

All sites in this study showed net $CH_4$ absorption by sandy soil. During vegetation restoration, improvements in micrometeorological and environmental conditions can gradually increase the amount of $CH_4$ absorbed by soil. Methane-oxidizing bacteria generally oxidize and consume $CH_4$ in soil under aerobic conditions, while under anaerobic conditions, fiber-decomposing bacteria and methanogenic bacteria produce $CH_4$ and release it into the atmosphere [?, ?, ?]. At the fixed site, increased aboveground biomass produced better shading effects and promoted development of mixed crusts, resulting in good soil aeration. Meanwhile, increased plant root biomass also increased soil porosity and decreased soil bulk density (Table 1 ). This higher porosity likely led to greater air permeability, thereby promoting oxidation and absorption of $CH_4$.

## 4.2 Effects of Grazing on GHG Fluxes

Grazing has two primary effects on GHG regulation: (1) it changes the ecosystem carbon pool and thus directly impacts GHG emissions [?, ?]; and (2) it indirectly affects exchange of GHGs between the ecosystem and atmosphere by altering soil physical, chemical, and biological properties [?, ?]. However, literature review reveals that grazing effects on soil GHG emissions vary with climate conditions and soil physicochemical properties [?, ?, ?, ?, ?, ?, ?]. In this study, grazing increased $CO_2$ emissions. In grasslands, livestock grazing typically does not degrade surface soil but can decrease belowground biomass and reduce soil respiration rates. However, in deserts, the soil surface is fragile and vegetation and biocrust can be easily destroyed by livestock, exposing organic carbon to the surface where it can be oxidized into $CO_2$ and emitted into the atmosphere [?, ?].

The effects of grazing on $N_2O$ flux also vary considerably among previous studies (Table 3 ). Most grassland studies show that grazing does not significantly affect $N_2O$ emissions, and that seasonal dynamics and source-sink patterns do not fluctuate significantly. However, the present study found that grazing increased $N_2O$ emissions from sandy soil. This could be attributed to the loose soil structure, which can be easily compacted under livestock trampling, resulting in higher soil bulk density, reduced soil porosity (Table 1 ), limited oxygen diffusion, and reduced oxygen concentration. These factors could lead to higher denitrification rates under anaerobic conditions [?, ?], thus promoting $N_2O$ emissions. Furthermore, livestock excrement is a primary source of $N_2O$ [?, ?]. Deposition and decomposition of livestock excrement add nitrogen to the soil, and increased $N_2O$ substrates enhance nitrification and subsequent $N_2O$ emissions.

Similar to most previous studies (Table 3 ), we found that grazing reduced soil $CH_4$ absorption. Grazing increased surface soil bulk density, leading to decreased water-holding capacity. This likely increased seepage resistance and decreased gas diffusion, creating anaerobic microsites where $CH_4$ could be produced.

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## 4.3 Factors Controlling GHG Fluxes

Pearson' s linear correlation and redundancy analysis revealed that T-10 and SWC-30 had the most significant influence on GHG emissions. GHG fluxes at all sites were significantly affected by seasonal changes, following trends similar to soil water content and temperature—high in spring and summer but low in autumn and winter. This pattern was consistent across all sites. GHG emissions and absorption are products of soil bio-metabolism and biochemical processes, which are sensitive to water and temperature conditions. Within a specific range, increased soil temperature can accelerate organic matter decomposition, increasing plant root and soil microorganism metabolism rates and thus increasing soil GHG emissions [?, ?]. However, at extremely low (<5°C) or high temperatures (>40°C), root activity, methanogenesis, and nitrification are inhibited [?, ?], explaining the lower GHG fluxes in autumn and winter. In both growing and non-growing seasons, soil water content at 30 cm depth had a significantly greater effect on GHG fluxes than at other depths, likely because the roots of dominant species (*S. cheilophila* and *A. ordosica*) dominated the 20-40 cm soil layer. Soil water content affects soil permeability, pH, microbial activity, and gas diffusion rates, which are closely related to plant root distribution [?, ?]. High soil water content can also alter root structure and increase root exudates, which in turn increase soil carbon and microbial activity [?, ?], thereby increasing GHG emissions.

## 5 Conclusions

In this study, the ecosystem of the sand-binding area remained highly fragile, with human activities such as vegetation restoration and grazing causing dramatic sandy soil changes. Grazing on vegetated sand destroys vegetation and significantly increases soil carbon and nitrogen leakage. The grazed site had the highest $CO_2$ and $N_2O$ emissions and the highest GWP. The mobile site had the lowest GHG fluxes and GWP, while the fixed site had the largest $CH_4$ uptake. Across all three sites, soil temperature at 10 cm depth and soil water content at 30 cm depth were the key micrometeorological factors determining GHG fluxes. These findings illustrate that revegetation and grazing have caused significant sandy soil changes and altered GHG fluxes and GWP. Furthermore, we determined the specific role of re-desertification caused by grazing, highlighting the need for a comprehensive understanding of GHG fluxes in such environments to facilitate rational exploitation of desert land resources.

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