

**Greenhouse gas emissions from river-riparian wetlands: An example from the Inner Mongolia grassland region in China**

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**Abstract:** ~~Riparian wetlands play a significant role in regulating carbon and nitrogen cycles.~~

Gradual riparian wetland drying is increasingly sensitive to global warming and contributes to climate change. Riparian wetlands play a significant role in regulating carbon and nitrogen cycles.

In this study, we analyzed the emissions of carbon dioxide (CO<sub>2</sub>), methane (CH<sub>4</sub>), and nitrous oxide (N<sub>2</sub>O) from riparian wetlands in the Xilin River Basin to understand the role of these ecosystems in greenhouse gas (GHG) emissions. Moreover, the impact of the catchment hydrology and soil property variations on GHG emissions over time and space were evaluated. Our results demonstrate that riparian wetlands emit larger amounts of CO<sub>2</sub> (335–2790 mg·m<sup>-2</sup>·h<sup>-1</sup> in wet seasonAugust and 72–387 mg·m<sup>-2</sup>·h<sup>-1</sup> in dry seasonOctober) than CH<sub>4</sub> and N<sub>2</sub>O to the atmosphere due to high plant and soil respiration. The results also reveal clear seasonal variations and spatial patterns along the transects and in the longitudinal direction. N<sub>2</sub>O emissions showed a spatiotemporal pattern similar to that of CO<sub>2</sub> emissions. Near-stream sites were the only sources of CH<sub>4</sub> emissions, while the other sites served as sinks for these emissions. Soil moisture content and soil temperature were the essential factors controlling the GHG emissions, and abundant aboveground biomass promoted the CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O emissions. Moreover, compared to different types of grasslands, riparian wetlands were the potential hotspots of GHG emissions in the Inner Mongolian region. Degradation of downstream wetlands has resulted in the loss of the soil carbon pool by approximately 60%, reducing CO<sub>2</sub> emissions by approximately 35%, and

shifting the CH<sub>4</sub> and N<sub>2</sub>O emissions from the source to the sink. Our study showed that anthropogenic activities have extensively changed the hydrological characteristics of the riparian wetlands and might accelerate carbon loss, which could further affect the GHG emissions.

**Key words:** Riparian wetlands, Grasslands, Greenhouse gas, Spatial-temporal distribution, Impact factor, Xilin River Basin

## 1. Introduction

With the increasing impacts of global warming, the change in the concentrations of greenhouse gases (GHGs) in the atmosphere is a source of concern in the scientific community (Cao et al., 2005). According to the World Meteorological Organization (WMO, 2018), the concentrations of carbon dioxide (CO<sub>2</sub>), methane (CH<sub>4</sub>), and nitrous oxide (N<sub>2</sub>O) have increased by 146%, 257%, and 122%, respectively, since 1750. Despite their lower atmospheric concentrations, CH<sub>4</sub> and N<sub>2</sub>O absorb infrared radiation approximately 28 and 265 times more effectively at centennial timescales than CO<sub>2</sub> (IPCC, 2013). On a global scale, CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O contribute 87% to the GHG effect (Ferrón et al., 2007).

Wetlands are unique ecosystems that serve as transition zones between terrestrial and aquatic ecosystems. They play an important role in the global carbon cycle (Beger et al., 2010; Naiman and Decamps, 1997). Wetlands are sensitive to hydrological changes, particularly in the context of global climate change (Cheng and Huang, 2016). Moreover, wetland hydrology is affected by local anthropogenic activities, such as the construction of reservoirs, resulting in gradual drying.

Although wetlands cover only 4–6% of the terrestrial land surface, they contain approximately 12–24% of global terrestrial soil organic carbon (SOC), thus acting as carbon sinks. Moreover, they release CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O into the atmosphere and serve as carbon sources (Lv et al., 2013).

Wetlands are increasingly recognized as an essential part of the nature, given their simultaneous functions as carbon sources and sinks. Excessive rainfall will cause an expansion in wetland areas and a sharp increase in the soil moisture content—(SMC)—thus enhancing respiration,

methanogenesis, nitrification, and denitrification rates (Mitsch et al., 2009). On the contrary, reduced precipitation or severe droughts will result in a decrease in water levels, causing the wetlands to dry up. The accumulated carbon will be released back into the atmosphere through oxidation. Due to the increasing impact of climate change and human activity, the drying of wetlands has been widely observed in recent years (Liu et al., 2006); more than half of global wetlands have disappeared since 1900 (Mitsch and Gosselink, 2007), and this tendency is expected to continue in the future. The loss of wetlands may directly shift the soil environment from anoxic to oxic conditions, and modify the CO<sub>2</sub> and CH<sub>4</sub> source and sink functions of wetland ecological systems (Waddington and Roulet, 2000; Zona et al., 2013).

The Xilin River Basin in China is characterized by a marked spatial gradient in SMC. It is a unique natural laboratory that may be used to explore the close relationships between the spatiotemporal variations in hydrology and riparian biogeochemistry. Wetlands around the Xilin River play an irreplaceable role with regard to local climate control, water conservation, the carbon and nitrogen cycles, and husbandry (Gou et al., 2015; Kou, 2018). Moreover, the Xilin River region is subjected to seasonal alterations in precipitation and temperature regimes, and construction of the Xilin River Reservoir has resulted in highly negative consequences, such as the drying of downstream wetlands, affecting riparian hydrology as well as microbial activity in riparian soils. GHG emissions in riparian wetlands vary immensely. Understanding the interactions between GHG emissions and hydrological changes in the Xilin River riparian wetlands has thus become increasingly important. Moreover, it is necessary to estimate the changes in GHG emissions as a result of wetland degradation at the local and global scales.

In this work, GHG emissions from riparian wetlands and adjacent hillslope grasslands of the Xilin River Basin were investigated. GHG emissions, soil temperature (ST), and SMC were measured in dry and wet seasons. The main objectives of this study were to (1) investigate the temporal and spatial variations in CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O emissions from the wetlands in the riparian zone, and examine the main factors affecting the GHG emissions, (2) compare the GHG emissions from the riparian wetlands and different types of grasslands, and (3) evaluate the impact of wetland degradation in the study area on GHG emissions.

## **2. Materials and methods**

## 2.1 Study site

The Xilin River is situated in the southeastern part of the Inner Mongolia Autonomous Region in China (E115°00'–117°30', N43°26'–44°39'). It is a typical inland river of the Inner Mongolia grasslands. The river basin area is 10,542 km<sup>2</sup>, the total length is 268.1 km, and the average altitude is 988.5 m. According to the meteorological data provided by the Xilinhot Meteorological Station (Xi et al., 2017; Tong et al., 2004), the long-term annual mean air temperature is 1.7°C, and the maximum and minimum monthly means are 20.8°C in July and –19.8°C in January, respectively. The average annual precipitation was 278.9 mm for the period of 1968–2015. Precipitation is distributed unevenly among the seasons, with 87.41% occurring between May and September.

Soil types in the Xilin River Basin are predominantly chernozems (86.4%), showing a significant zonal distribution as light chestnut soil, dark chestnut soil, and chernozems from the northwest to southeast. Soil types in this basin also present a vertical distribution with elevation. The chernozems are primarily soluble chernozems and carbonate chernozems, distributed at altitudes above 1350 m with a relatively fertile and deep soil layer. Dark chestnut soil, boggy soil, and dark meadow with high humus content are distributed between the altitudes of 1150 and 1350 m. Light chestnut soil, saline meadow soil, and meadow solonchak with low soil humus, a thin soil layer, and coarse soil texture are distributed between the altitudes of 902 and 1150 m (Xi et al., 2017).

## 2.2 Field measurements and laboratory analyses

In this study, five representative transects were selected as the primary measurement sites in the entire Xilin River. Each transect cuts through the riparian wetlands near the river and hillslope grasslands further away from it (Fig. 1).

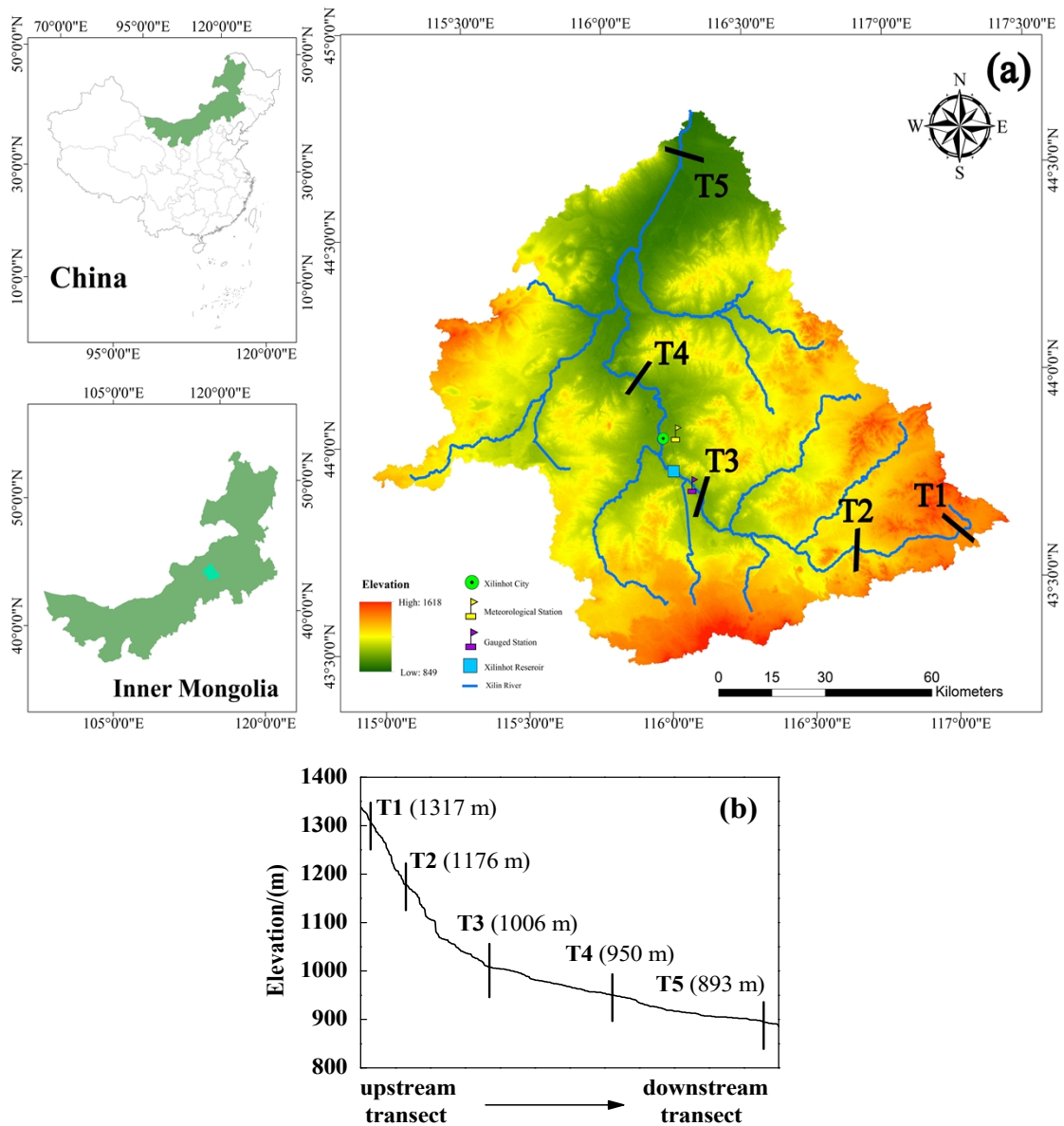


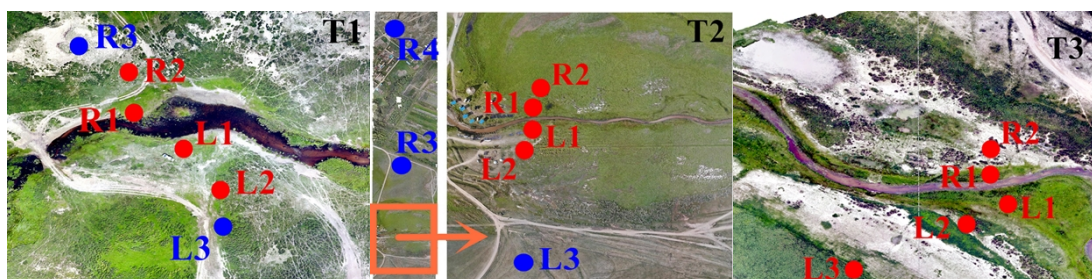
Fig. 1 (a) Location of the Xilin River Basin and distribution of five riparian-hillslope transects (T1–T5). (b) Elevation details of each transect in the Xilin River Basin.

The layout of the sampling points of each transect is shown in Fig. 2. Each sampling point ~~for~~from T1–T5 was extended from the river to both sides, to the grassland on the slopes, using 5–7 sampling points for each transect and resulting in 24 points in total. The sampling sites on the left and right banks were defined as L1–L3 and R1–R4 from the riparian wetlands to the hillslope grasslands. As transect T3 was located on a much wider flood plain, none of its sampling points were located on the hillslope grassland. The last transect (T5) was located downstream in the dry lake and contained seven sampling points. They were defined as S1–S7, where S1, S2, and S7

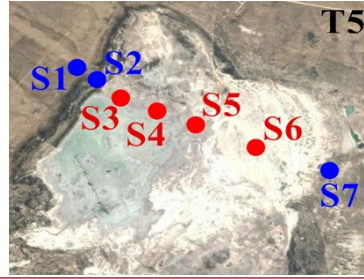
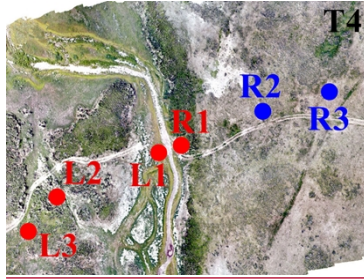
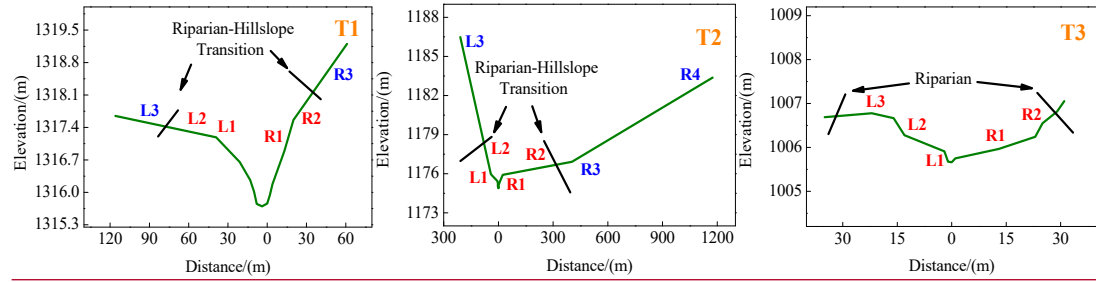
were located along the lake shore (the lakeside zone), and S3–S6 were located in the dry lake bed (S3 and S4 in the mudbank, S5 in saline–alkali soil, and S6 in sand–gravel geology). Moreover, characterizations for T1, T2, and T3 transects were the continuous river flow and T4 and T5 transects were the intermittent river flow.

The CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O emissions from each site were measured in August (wet season) and October (dry season) in 2018 using a static dark chamber and the gas chromatography method. The static chambers were made of a cube-shaped polyvinyl chloride (PVC) pipe (dimensions: 0.4 m × 0.2 m × 0.2 m). A battery-driven fan was installed horizontally inside the top wall of the chamber to ensure proper air mixing during measurements. To minimize heating from solar radiation, white adiabatic aluminum foil was used to cover the entire aboveground portion of the chamber. During measurements, the chambers were driven into the soil to ensure airtightness and connected with a differential gas analyzer (Li-7000 CO<sub>2</sub>/H<sub>2</sub>O analyzer, LI-COR, USA) to measure the changes in the soil CO<sub>2</sub> concentration. The air in the chamber was sampled using a 60 mL syringe at 0, 7, 14, 21, and 28 min. The gas samples were stored in the reservoir bag and taken to the laboratory for CH<sub>4</sub> and N<sub>2</sub>O measurements using gas chromatography (GC-2030, Japan). The measurements were scheduled for 9:00–11:00 a.m. and/or 3:00–5:00 p.m.

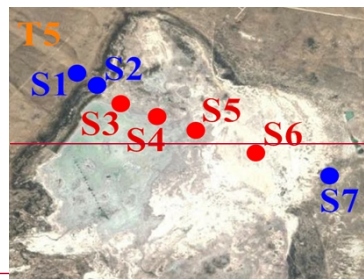
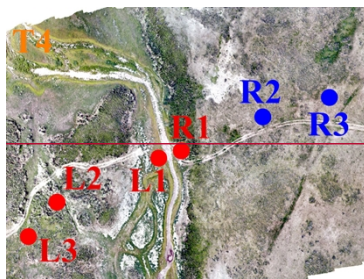
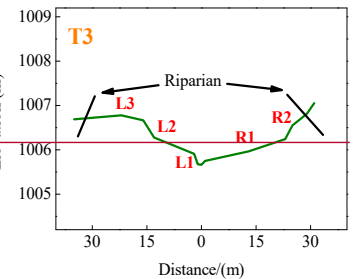
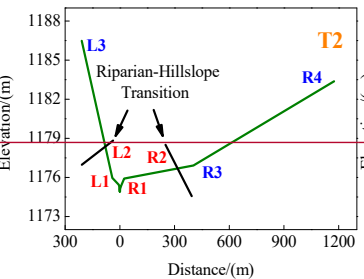
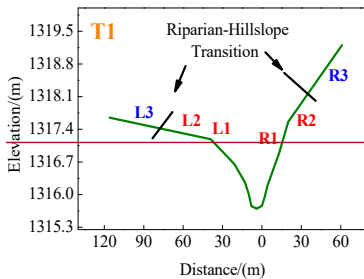
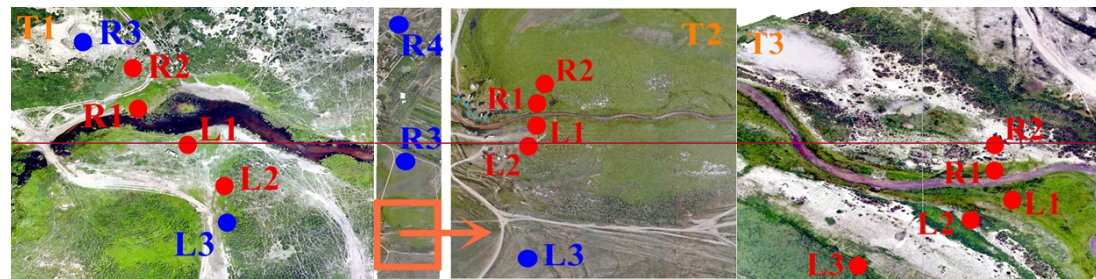
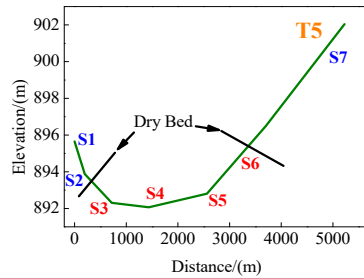
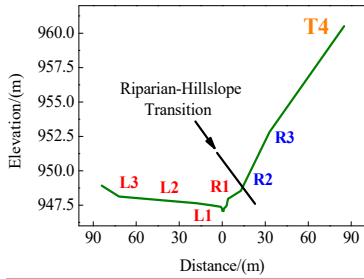
ST was measured at depths of 0–10 cm and 10–20 cm with a geothermometer (DTM-461, Hengshui, China). Plant samples were collected in a static chamber and oven-dried in the laboratory to obtain aboveground biomass (BIO). A 100 cm<sup>3</sup> ring cutter was used to collect surface soil samples at each site, which were placed in aluminum boxes and immediately brought back to the laboratory to measure soil mass moisture content (SMC) and soil bulk density ( $\rho_b$ ) using national standard methods (NATESC, 2006). Topsoil samples were collected, sealed in plastic bags, and brought back to the laboratory to measure soil pH, electrical conductivity (EC), total soil organic carbon (TOC), and soil C:N ratio.







Transect	Riparian	Hillslope
T1	L1, L2, R1, R2	L3, R3
T2	L1, L2, R1, R2	L3, R3, R4
T3	L1, L2, L3, R1, R2	—
T4	L1, L2, L3, R1	R2, R3
T5	S1, S2, S7	S3–S6 in the lake



Transect	Riparian	Hillslope
T1	L1, L2, R1, R2	L3, R3
T2	L1, L2, R1, R2	L3, R3, R4
T3	L1, L2, L3, R1, R2	—
T4	L1, L2, L3, R1	R2, R3
T5	S1, S2, S7	S3–S6 in the lake

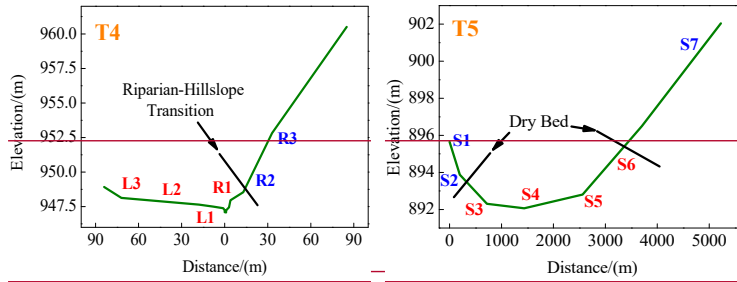


Fig. 2 Distributions of sampling points in transects T1–T5 (The images are authors' own)

Table 1. Physical and chemical properties (Mean  $\pm$  SD (standard deviation)) of soils at various sites within each transect

Transect	Zone	Samples number	Soil C:N	TOC (g·kg <sup>-1</sup> )	BIO (g)	$\rho_b$	pH	EC (μs/cm)	SSM (%)
T1	Riparian	12	12.46 $\pm$ 0.91	30.16 $\pm$ 6.54	14.67 $\pm$ 5.44	1.28 $\pm$ 0.07	7.25 $\pm$ 0.62	154.71 $\pm$ 23.70	47.77
	Hillslope	6	11.41 $\pm$ 0.09	10.77 $\pm$ 4.72	6.70 $\pm$ 1.48	1.45 $\pm$ 0.03	7.22 $\pm$ 0.40	82.02 $\pm$ 16.37	31.02
T2	Riparian	12	11.70 $\pm$ 1.14	19.96 $\pm$ 5.71	24.76 $\pm$ 9.65	1.23 $\pm$ 0.05	8.95 $\pm$ 0.45	303.88 $\pm$ 102.16	51.21
	Hillslope	9	9.77 $\pm$ 0.88	14.87 $\pm$ 11.21	6.10 $\pm$ 3.19	1.38 $\pm$ 0.13	8.10 $\pm$ 0.55	162.97 $\pm$ 128.18	35.09
T3	Riparian	12	15.80 $\pm$ 4.16	22.40 $\pm$ 9.69	26.65 $\pm$ 6.37	1.35 $\pm$ 0.19	9.50 $\pm$ 0.67	1233.20 $\pm$ 829.83	58.05
	L3	3	16.8	36.1	107.75	0.072	8.5	403	100
T4	Riparian	12	12.52 $\pm$ 2.06	9.96 $\pm$ 1.25	11.97 $\pm$ 4.50	1.30 $\pm$ 0.08	8.84 $\pm$ 0.22	461.72 $\pm$ 314.27	44.08
	Hillslope	6	9.97 $\pm$ 0.50	9.65 $\pm$ 1.05	7.84 $\pm$ 2.48	1.30 $\pm$ 0.09	8.23 $\pm$ 0.14	118.5 $\pm$ 8.25	39.43
T5	Lake shore	9	63.74 $\pm$ 12.93	31.41 $\pm$ 6.55	5.48 $\pm$ 2.35	1.16 $\pm$ 0.10	9.88 $\pm$ 0.18	7320.87 $\pm$ 4300.03	58.47
	Dry lake-bed	12	15.92 $\pm$ 4.71	6.35 $\pm$ 1.16	0	1.33 $\pm$ 0.09	9.41 $\pm$ 0.7	281.82 $\pm$ 162.73	37.52
Transect	Zone	Samples number	Soil C:N	TOC (g·kg <sup>-1</sup> )	BIO (g)	$\rho_b$	pH	EC (μs/cm)	SSM (%)
T1	Riparian	12	12.46 $\pm$ 0.91	30.16 $\pm$ 6.54	14.67 $\pm$ 5.44	1.28 $\pm$ 0.07	7.25 $\pm$ 0.62	154.71 $\pm$ 23.70	47.77 $\pm$ 7.04
	Hillslope	6	11.41 $\pm$ 0.09	10.77 $\pm$ 4.72	6.70 $\pm$ 1.48	1.45 $\pm$ 0.03	7.22 $\pm$ 0.40	82.02 $\pm$ 16.37	31.02 $\pm$ 1.32
T2	Riparian	12	11.70 $\pm$ 1.14	19.96 $\pm$ 5.71	24.76 $\pm$ 9.65	1.23 $\pm$ 0.05	8.95 $\pm$ 0.45	303.88 $\pm$ 102.16	51.21 $\pm$ 6.49
	Hillslope	9	9.77 $\pm$ 0.88	14.87 $\pm$ 11.21	6.10 $\pm$ 3.19	1.38 $\pm$ 0.13	8.10 $\pm$ 0.55	162.97 $\pm$ 128.18	35.09 $\pm$ 6.75
T3	Riparian	12	15.80 $\pm$ 4.16	22.40 $\pm$ 9.69	6.37 $\pm$ 2.95	1.35 $\pm$ 0.19	9.50 $\pm$ 0.67	1233.20 $\pm$ 829.83	47.56 $\pm$ 11.65
	L3	3	16.8 $\pm$ 0.58	36.1 $\pm$ 1.84	107.75	0.072 $\pm$ 0.01	8.5 $\pm$ 0.17	403 $\pm$ 57.21	>100



Note: SMC10-V - soil volumetric moisture content in 0-10 cm; SMC20-V - soil volumetric moisture content in 10-20 cm; Soil C:N - soil carbon-nitrogen ratio; TOC - total soil organic carbon; BIO - aboveground biomass;  $\rho_b$  - soil bulk density; pH - soil pH; EC - soil electrical conductivity; SSM - saturated soil moisture.

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Transect	Zone	soil particle composition		
		Clay % (<0.002 mm)	Silt % (0.02~0.002 mm)	Sand (2.0 ~0.02 mm)
T1	Riparian	2.5	2.7	94.8
	Hillslope	9.6	6.1	85.3
T2	Riparian	5.5	5.8	90.7
	Hillslope	10.8	8.6	80.6
T3	Riparian	4.1	1.1	94.8
T4	Riparian	11.4	1.5	87.1
	Hillslope	12.7	5.9	81.4
T5	Lake shore	5.1	2.1	92.8
	Dry lake bed	46.1	4.8	49.1

### 2.3 Calculation of GHG emissions

The CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O emissions were calculated using Eq. 1 (Qin et al., 2016):

$$F = \frac{V}{A} \times \frac{dc}{dt} \times \rho = H \times \frac{dc}{dt} \times \frac{M}{V} \times \left( \frac{273.15}{273.15 + t} \right) \quad (1)$$

Where  $F$  denotes the CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O emissions (mg·m<sup>-2</sup>·h<sup>-1</sup>),  $H$  is the height of the static chamber (0.18 m),  $M$  is the relative molecular weight (44 for CO<sub>2</sub> and N<sub>2</sub>O, and 16 for CH<sub>4</sub>),  $V$  is the volume of gas in the standard state (22.4 L·mol<sup>-1</sup>),  $dc/dt$  is the rate of change of the gas concentration (10<sup>-6</sup> h<sup>-1</sup>), and  $T$  is the temperature in the black chamber (°C).

The annual cumulative emissions were calculated using Eq. 2 (Whiting G and Chanton J., 2001)

$$M = \sum \frac{F_{i+1} + F_i}{2} \times (t_{i+1} - t_i) \times 24 \quad (2)$$

Where  $M$  denotes the total cumulative emissions of CO<sub>2</sub>, CH<sub>4</sub>, or N<sub>2</sub>O (kg·hm<sup>2</sup>),  $F$  is the emission flux of CO<sub>2</sub>, CH<sub>4</sub>, or N<sub>2</sub>O,  $i$  is the sampling frequency,  $t_{i+1}-t_i$  represents the interval between two adjacent measurement dates.

In this study, a 100-year scale was selected to calculate the global warming potential (GWP) of soil CH<sub>4</sub> and N<sub>2</sub>O emissions (Whiting G and Chanton J., 2001):

$$GWP = 1 \times [CO_2] + 25 \times [CH_4] + 298 \times [N_2O] \quad (3)$$

Where 25 and 298 are GWP multiples of CH<sub>4</sub> and N<sub>2</sub>O relative to CO<sub>2</sub> on a 100-year time scale, respectively.

## 2.4 Statistical Analysis

All statistical analyses were performed using SPSS for Windows version 18.0 (SPSS Inc., Chicago, IL, USA). Statistical significance was set at  $P < 0.05$ . Pearson correlation analysis was conducted to estimate the relationships between GHGs fluxes and environmental variables. A Wilcoxon test was used to determine the difference of GHGs fluxes in two seasons.

## 3. Results

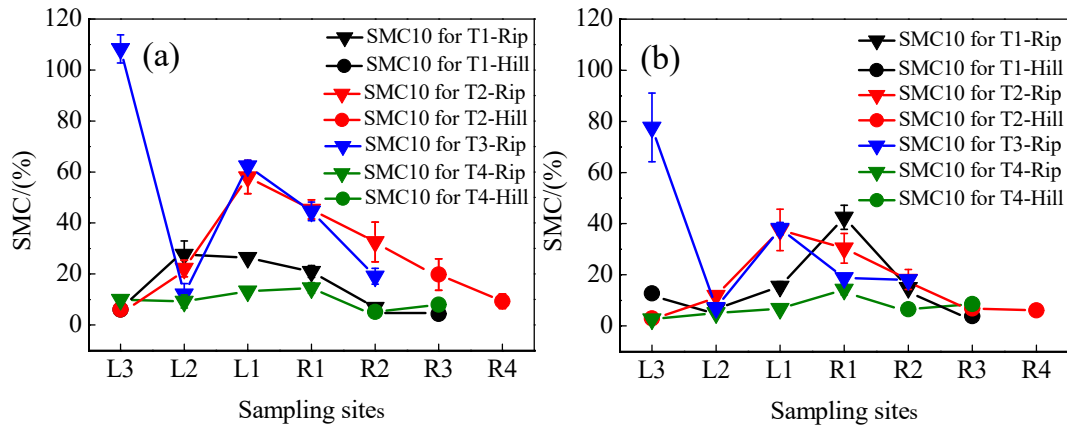
### 3.1 Spatiotemporal patterns of SMC for each transect

The temporal and spatial variations in SMC10 in the following order: wet season > dry season and riparian wetlands > hillslope grasslands (Fig. 3a, c, e). Similar variations were observed in SMC20 (Fig. 3b, d, f). The average SMC10 and SMC20 in the continuous river transects in the riparian zones (37.44% in wet season and 19.40% in dry season; 25.96% in wet season and 17.39% in dry season) were higher than those in the hillslope grasslands (9.12% in wet season and 4.15% in dry season; 6.51% in wet season and 5.96% in dry season). During the study period, both SMC10 and SMC20 changed as the distance from the river increased, and the highest value was observed at the near-stream sites (L1 and R1). SMC10 fluctuations were low in the intermittent transect compared to the upstream transects, with a mean value of 11.79% in wet season and 3.72% in dry season in the riparian areas. The mean SMC10 in the hillslopes was 6.58% in wet season and 2.86% in dry season. SMC20 showed similar fluctuation, 7.22% in wet season and 2.98% in dry season in the riparian areas and 7.56% in wet season and 4.4% in dry season in the hillslopes. In transect T5, average SMC10 and SMC20 at the center of the lake (29.00% in wet season and 13.36% in dry season; 29.30% in wet season and 9.69% in dry season) were higher than those along the lake shore (4.90% in wet season and 3.13% in dry season; 3.34% in wet season and 5.22% in dry season). The spatiotemporal pattern of SMC is illustrated in Fig. 3.

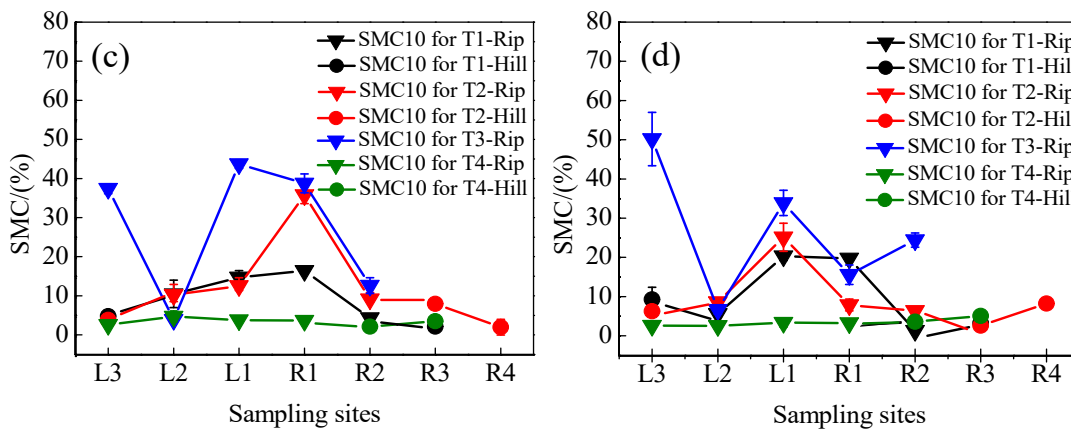
~~The temporal and spatial variations in SMC10. Similar variations were observed between soil depths of 0–10 cm and 10–20 cm in the following order: wet season (August) > dry season (October) and riparian wetlands > hillslope grasslands (Fig. 3a, c, e). Similar variations were observed in SMC20 (Fig. 3b, d, f). We averaged the SMC10 and SMC20 at the same sampling point. The average SMCs in the continuous river transects in the riparian zones (31.70% in wet season/August and 18.40% in dry season/October) were higher than those in the hillslope~~

grasslands (7.82% in wet season August and 5.06% in dry season October). During the study period, the SMC changed as the distance from the river increased, and the highest value was observed at the near-stream sites (L1 and R1). SMC fluctuations were low in the intermittent transect compared to the upstream transects, with a mean value of 9.50% in wet season August and 3.35% in dry season October in the riparian areas. The mean SMC in the hillslopes was 7.07% in wet season August and 3.63% in dry season October. In transect T5, average SMCs at the center of the lake (29.15% in wet season August and 11.52% in dry season October) were higher than those along the lake shore (4.12% in August wet season and 4.18% in dry season October).

#### Wet season



#### Dry season



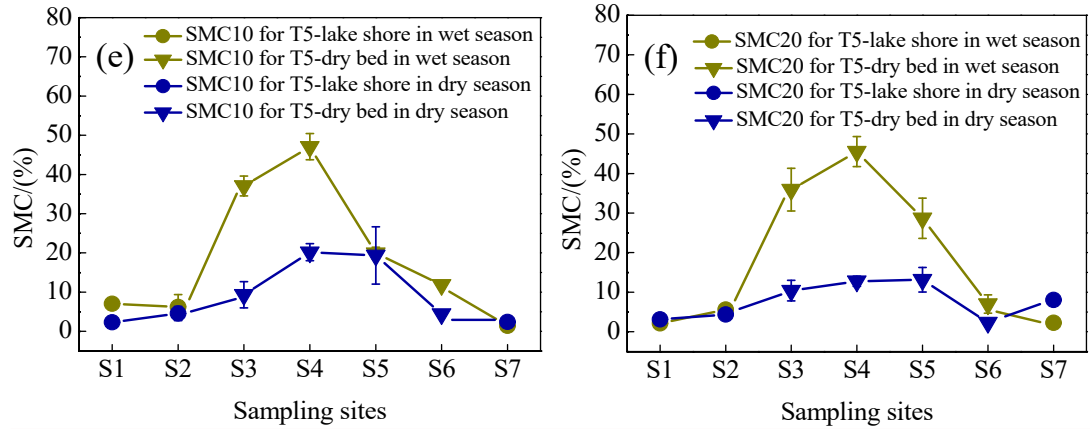
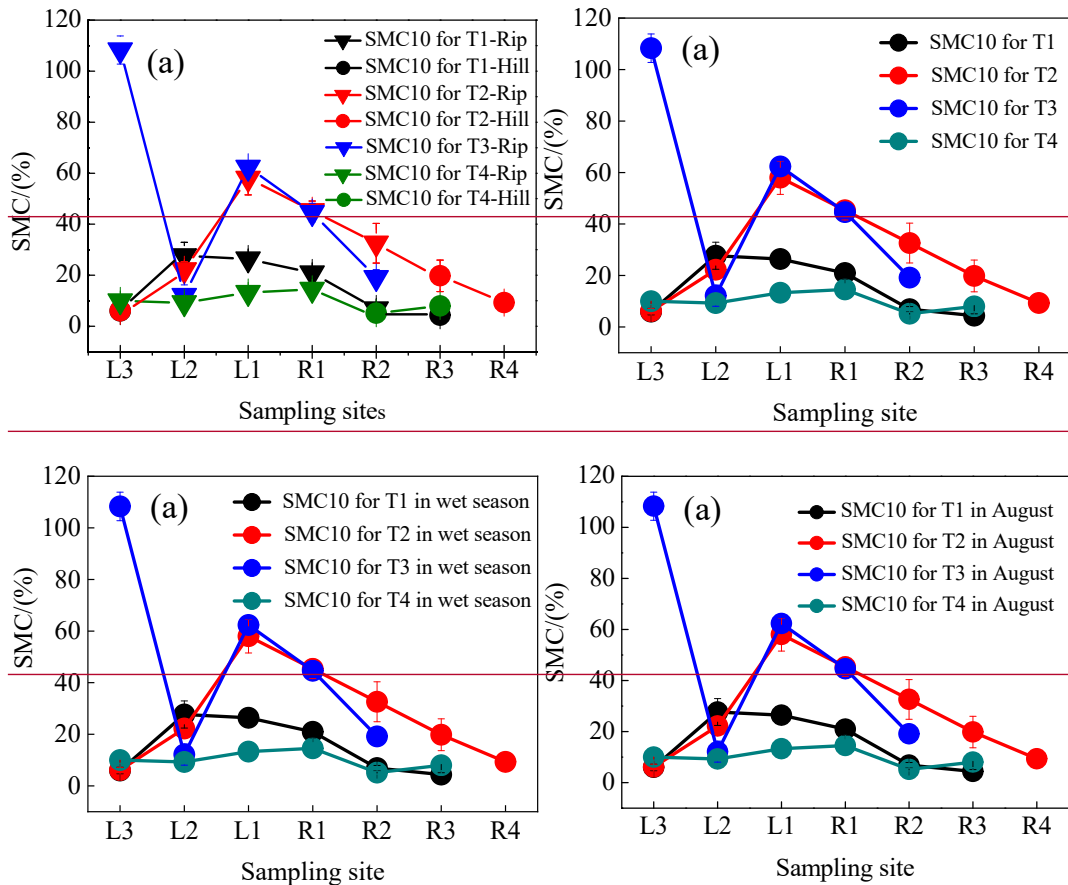
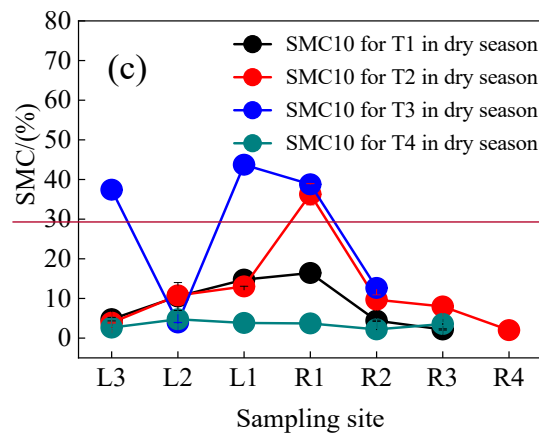
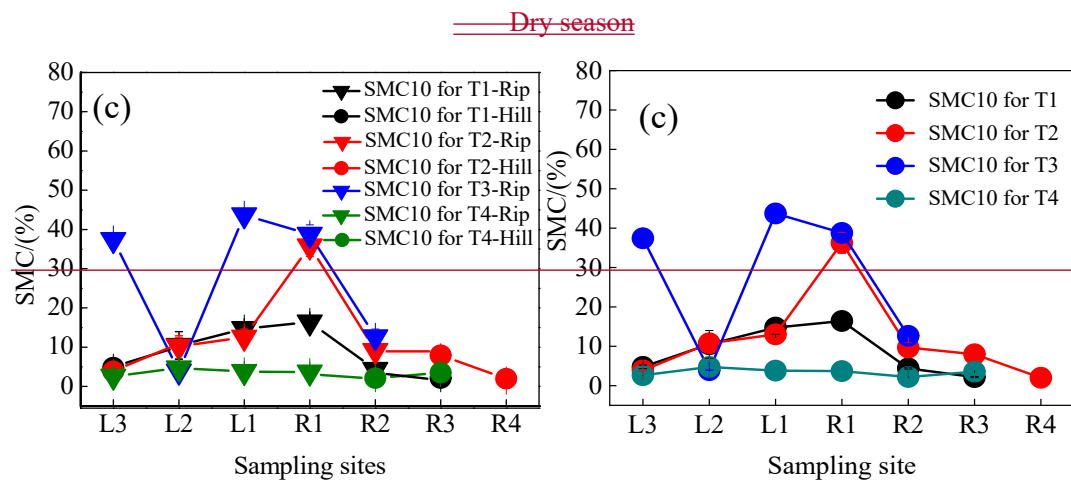
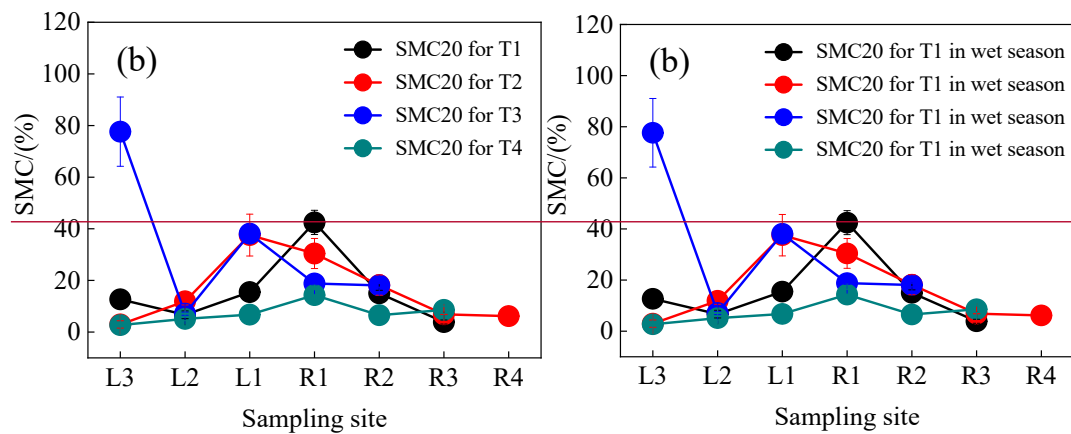
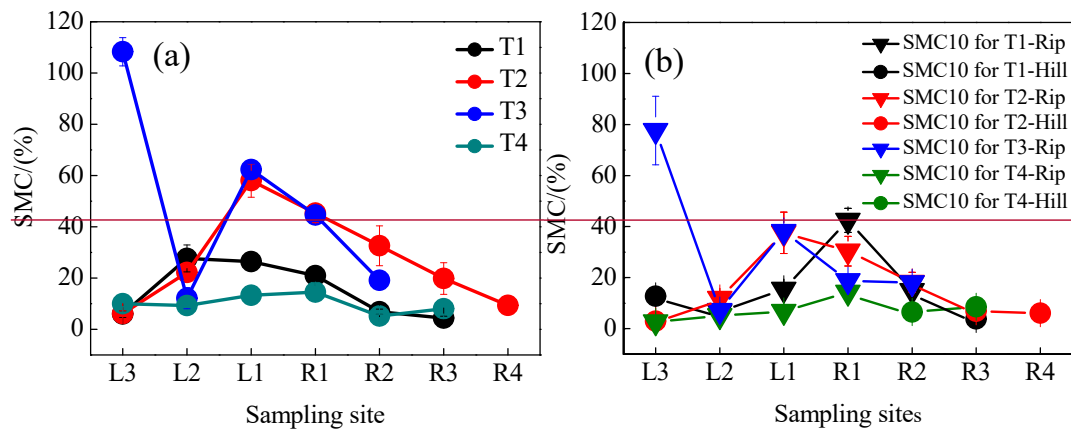


Fig. 3 Soil mass moisture contents (SMCs) at soil depths of 0–10 cm (SMC10) and 10–20 cm (SMC20) for transects T1–T5 in wet season and dry season. Error bars represent the SD about the mean.

#### Wet season







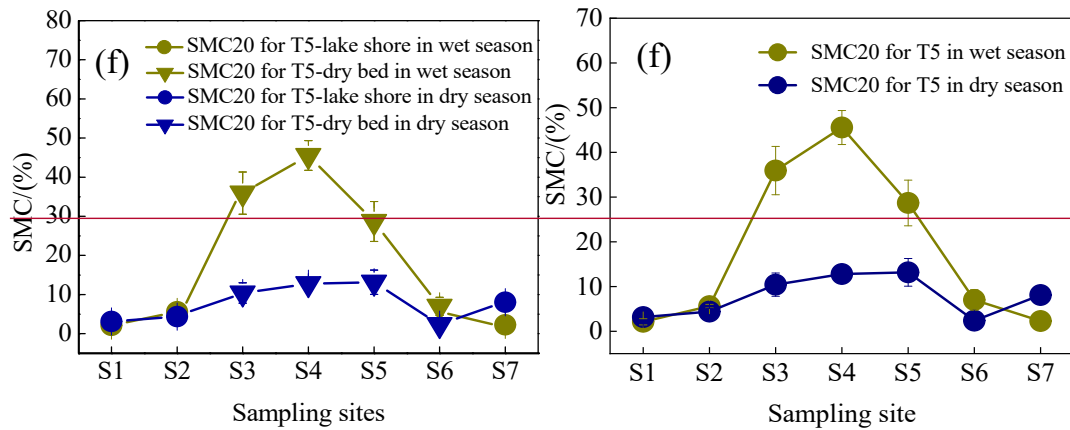
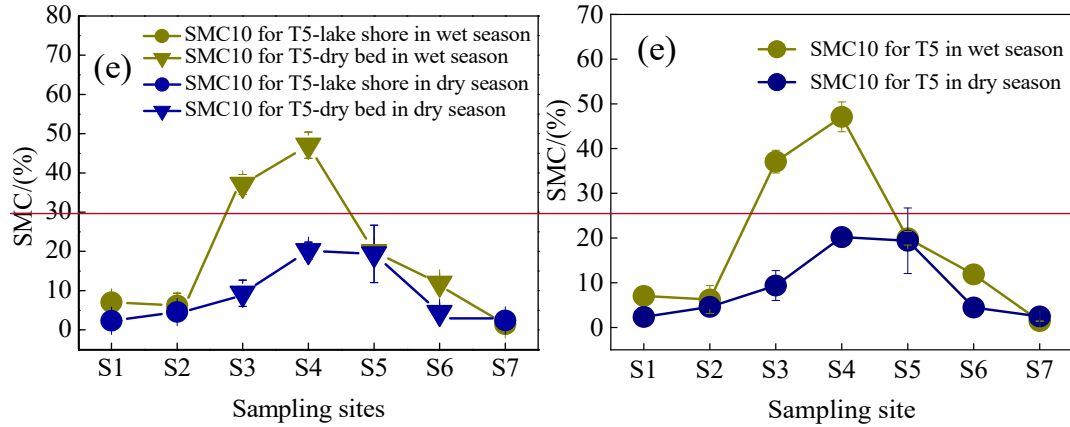
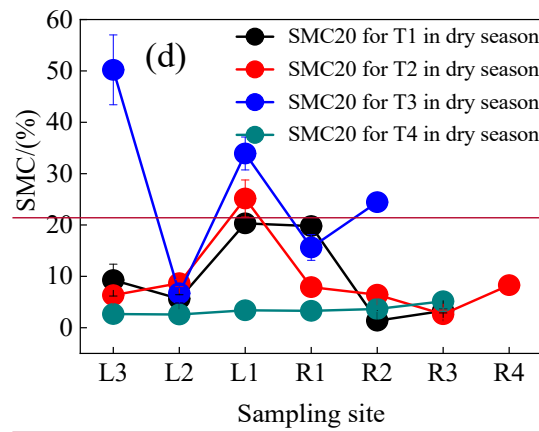
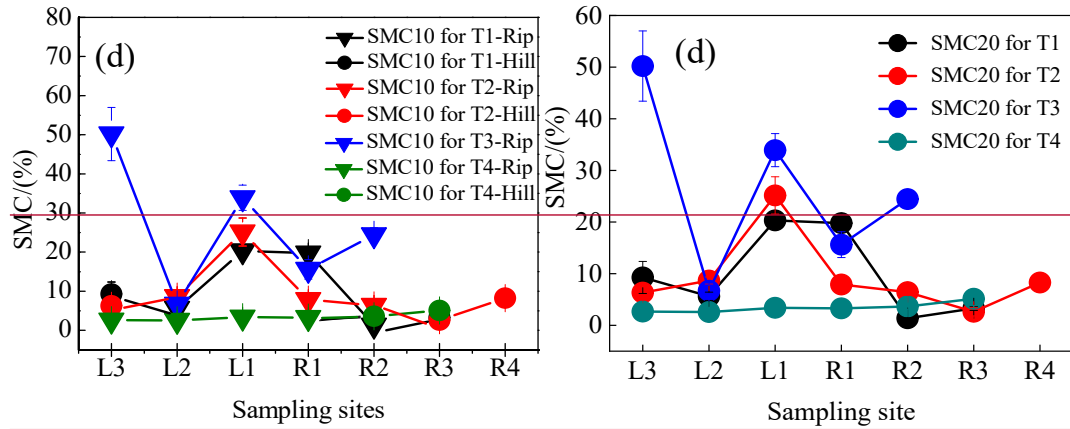
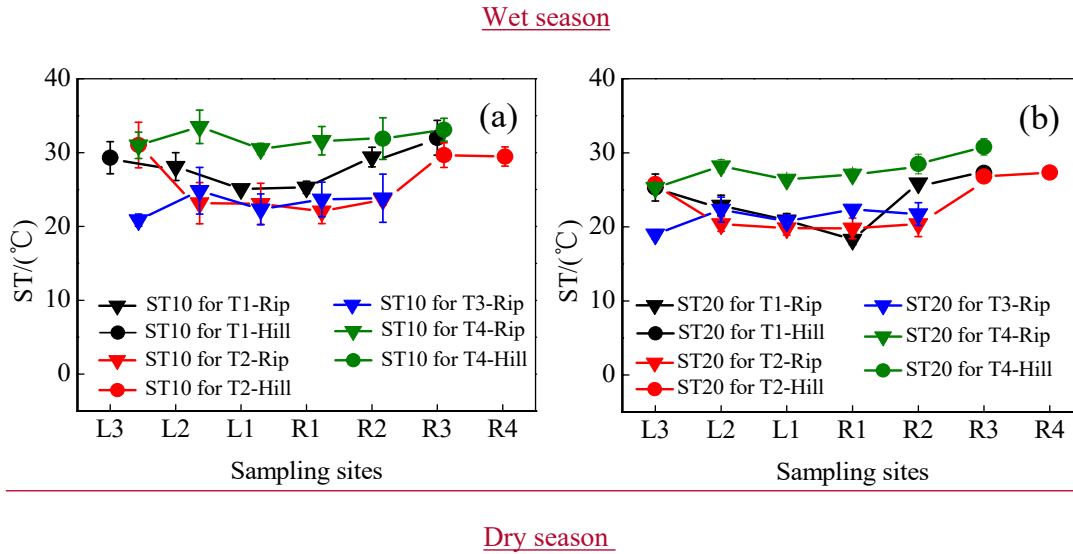
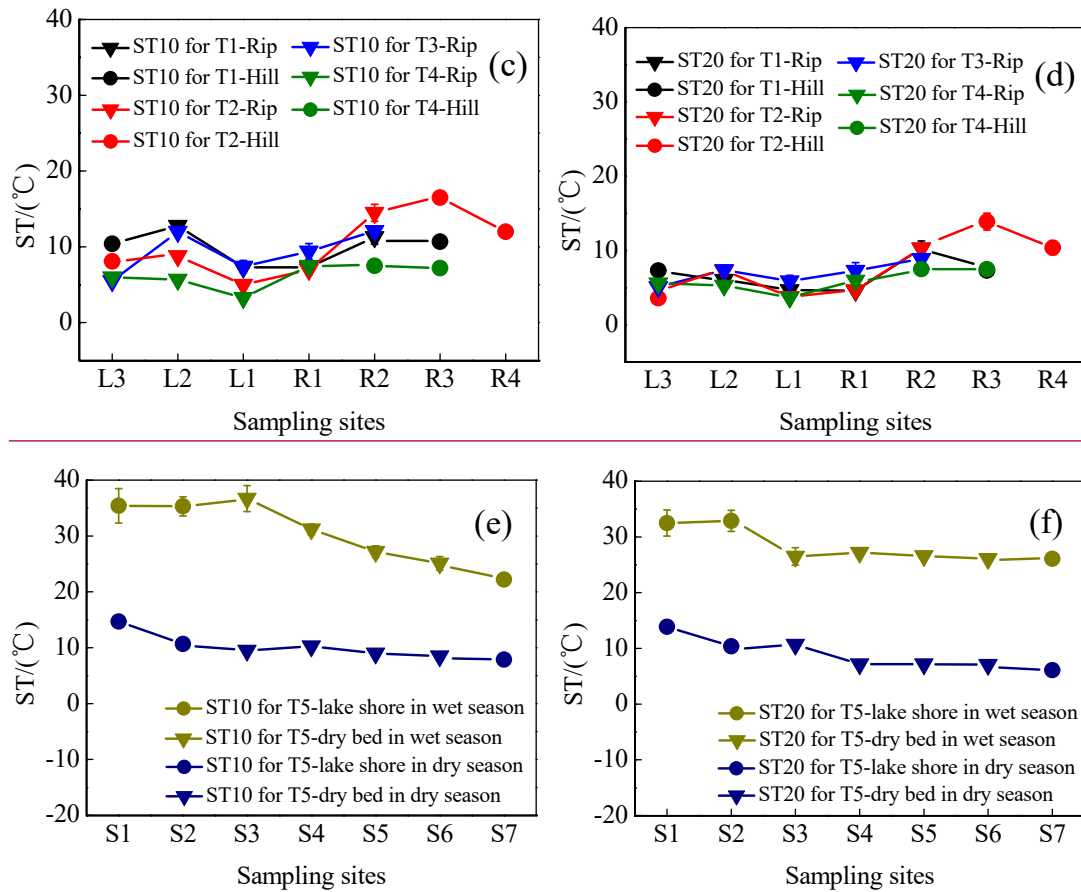


Fig. 3 Soil moisture contents (SMCs) at soil depths of 0–10 cm (SMC10) and 10–20 cm (SMC20) for transects T1–T5 in wet season (August) and dry season (October). Error bars represent the SD about the mean. (a) SMC10 for T1–T4 in August. (b) SMC20 for T1–T4 in August. (c) SMC10 for T1–T4 in October. (d) SMC20 for T1–T4 in October. (e) SMC10 for T5 in August and October. (f) SMC20 for T5 in August and October.

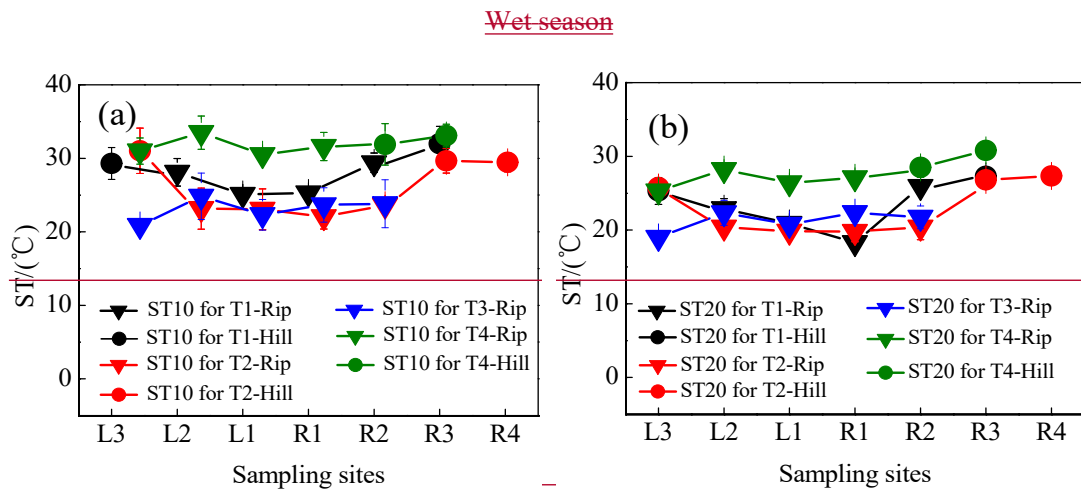
### 3.2 Spatiotemporal patterns of ST in each transect

Spatiotemporal differences in ST during the entire observation period are displayed in Fig. 4. ST variations in wet season (mean value: 27.4°C) were noticeably higher than those in dry season (mean value: 8.97°C). Moreover, ST for riparian sites (mean values: 26.0°C in wet season and 8.41°C in dry season) was slightly lower than that for the hillslope grasslands (mean values: 30.9°C in wet season and 10.3°C in dry season) for the 0–10 cm soil depth, with the exception of transect T5. Similar results were observed for the 10–20 cm soil depth.





**Fig. 4** Soil temperature (ST) at soil depths of 0–10 cm (ST10) and 10–20 cm (ST20) for transects T1–T5 in wet season and dry season. Error bars represent the SD about the mean.



Dry season

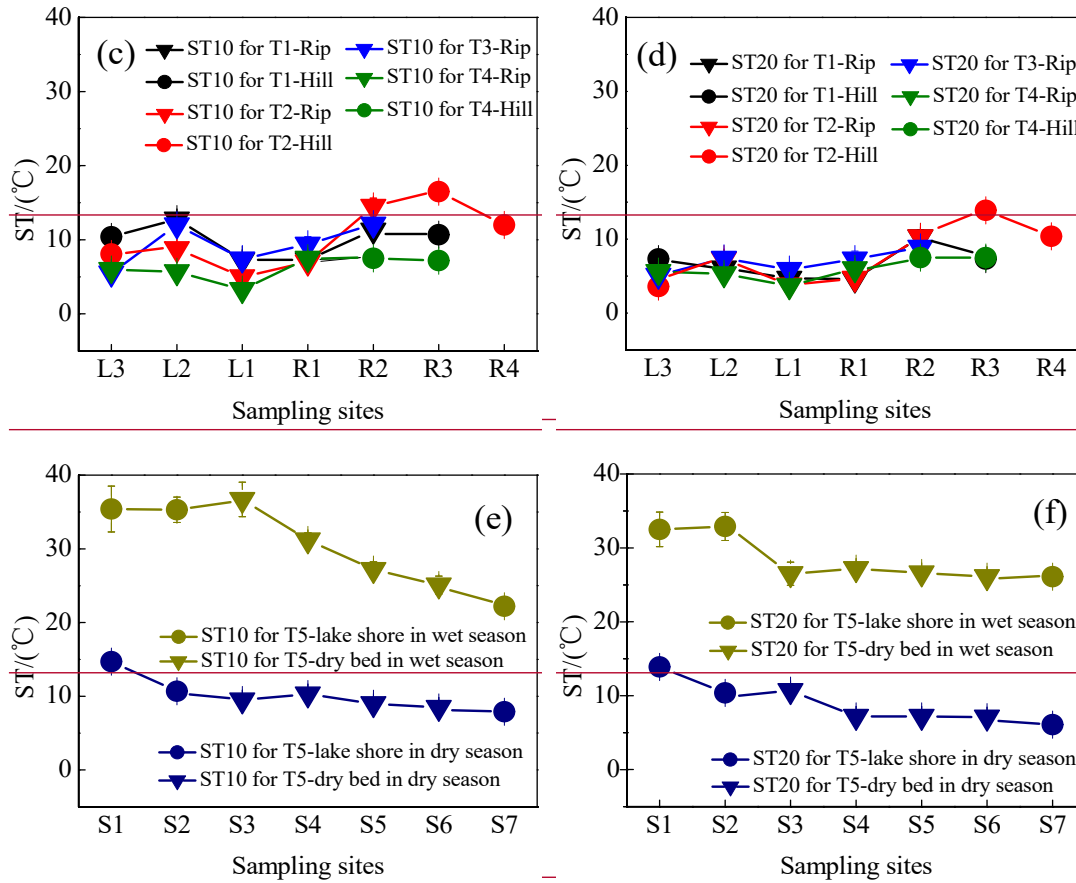


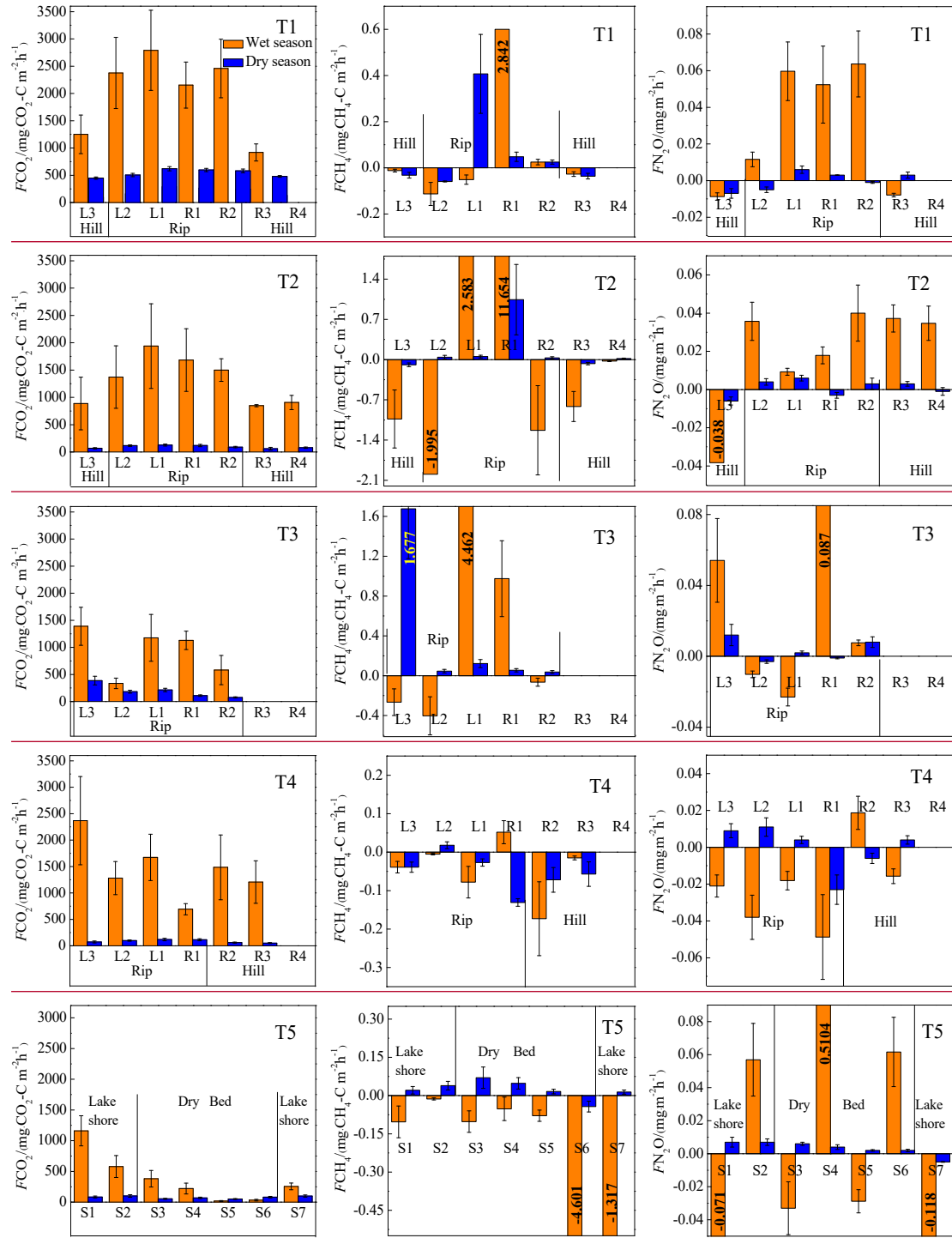
Fig. 4 Soil temperature (ST) at soil depths of 0–10 cm (ST10) and 10–20 cm (ST20) for transects T1–T5 in wet season (August) and dry season (October). Error bars represent the SD about the mean. August and October: (a) ST10 for T1–T4 in August, (b) ST20 for T1–T4 in August, (c) ST10 for T1–T4 in October, (d) ST20 for T1–T4 in October, (e) ST10 for T5 in August and October, (f) ST20 for T5 in August and October.

### 3.3 Spatiotemporal patterns of GHG emissions in each transect

Figure 5 shows the spatiotemporal variations in GHG emissions in **August** (wet season) and **October** (dry season) in each transect. CO<sub>2</sub> emissions in each transect were higher in **wet season** **August** than in **dry season** **October**. The average emissions for the riparian wetlands of transects T1–T4 ( $1582.09 \pm 679.34 \text{ mg} \cdot \text{m}^{-2} \cdot \text{h}^{-1}$  in **wet season** **August** and  $163.24 \pm 84.98 \text{ mg} \cdot \text{m}^{-2} \cdot \text{h}^{-1}$  in **dry season** **October**) were higher than those for the hillslope grasslands ( $1071.54 \pm 225.39 \text{ mg} \cdot \text{m}^{-2} \cdot \text{h}^{-1}$  in **wet season** **August** and  $77.68 \pm 25.32 \text{ mg} \cdot \text{m}^{-2} \cdot \text{h}^{-1}$  in **dry season** **October**). Higher CO<sub>2</sub> fluxes occurred in the riparian zones, while lower CO<sub>2</sub> fluxes were observed in the hillslope grasslands in continuous river transects (T1, T2, and T3). Transect T4 exhibited lower



CO<sub>2</sub> emissions in the riparian wetlands near the channel than at sites away from the channel. CO<sub>2</sub> emissions in transect T5 in wet seasonAugust and dry seasonOctober decreased from the lake shore to the lake center.



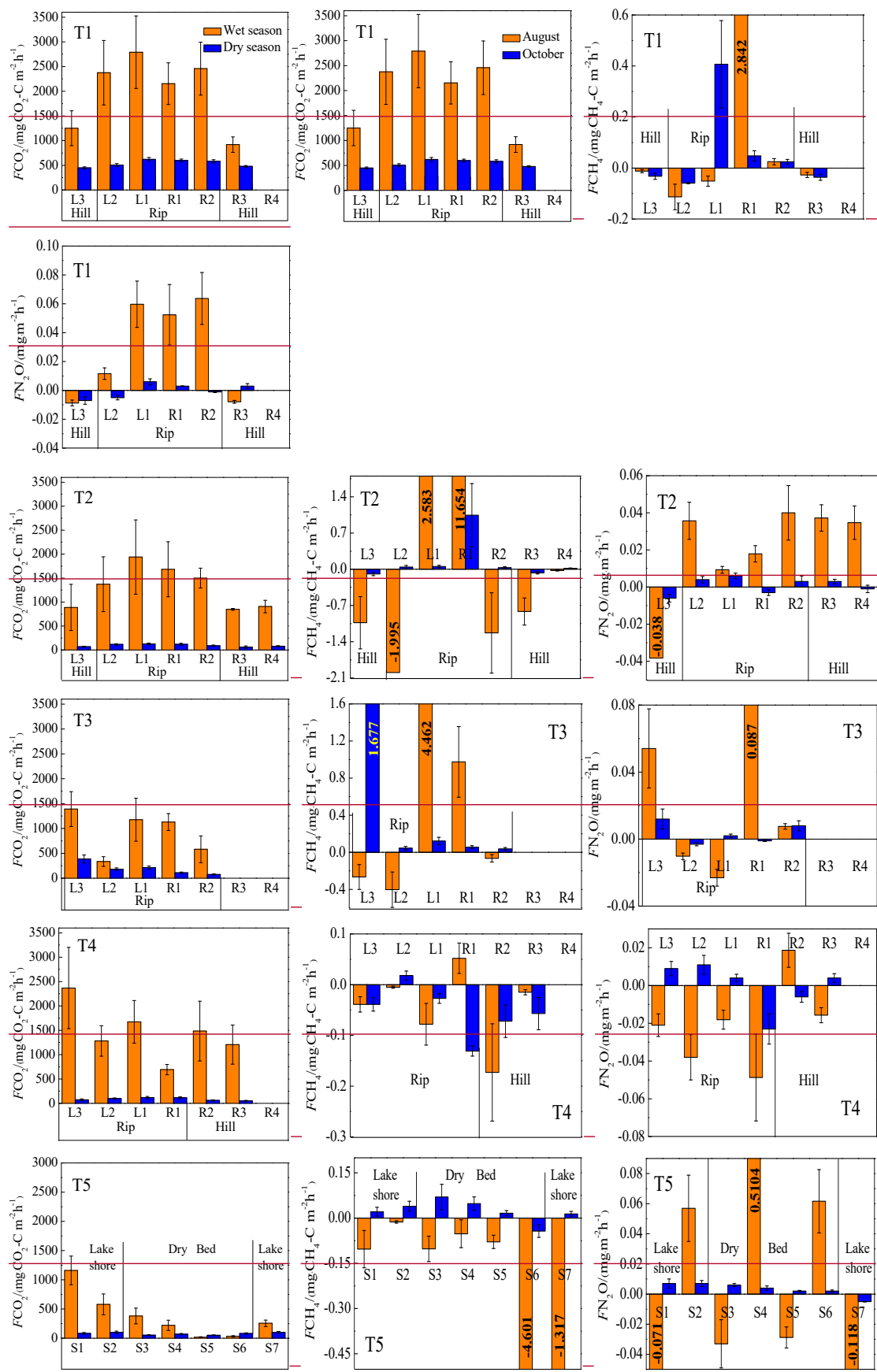


Fig. 5 Spatiotemporal patterns of CO<sub>2</sub> (first column), CH<sub>4</sub> (second column), and N<sub>2</sub>O (third column) emissions (*F*) for each transect. Data are shown for wet seasonAugust (orange) and dry seasonOctober (blue) and error bars are the standard deviations.

CH<sub>4</sub> emissions at the transects with continuous river flow (T1, T2, and T3) varied between wet seasonAugust and dry seasonOctober, except for T4 (characterized by intermittent river flow) and T5 (the dry lake). In wet seasonAugust, the near-stream sites (L1 and R1) in T1, T2, and T3 were characterized as high CH<sub>4</sub> sources (average:  $3.74 \pm 3.81 \text{ mg} \cdot \text{m}^{-2} \cdot \text{h}^{-1}$ ), but the sites located away from the river gradually turned into CH<sub>4</sub> sinks. Moreover, all the sites in transects T4 and T5 were sinks. CH<sub>4</sub> emissions (mean value:  $0.2 \pm 0.45 \text{ mg} \cdot \text{m}^{-2} \cdot \text{h}^{-1}$ ) at the wetland sites were always lower in dry seasonOctober than those in wet seasonAugust. However, the sites on the hillslope grasslands served as CH<sub>4</sub> sinks (mean value:  $-0.05 \pm 0.03 \text{ mg} \cdot \text{m}^{-2} \cdot \text{h}^{-1}$ ). In transect T5, CH<sub>4</sub> emissions revealed the opposite trend; a CH<sub>4</sub> sink was observed in wet seasonAugust, but it was transformed into a CH<sub>4</sub> source in dry seasonOctober.

Similar to the CO<sub>2</sub> and CH<sub>4</sub> emissions, N<sub>2</sub>O emissions showed a distinct spatiotemporal pattern for all the transects. N<sub>2</sub>O emissions in wet seasonAugust were higher than those in dry seasonOctober. These emissions were higher in riparian wetlands than in hillslope grasslands. Moreover, almost all the sites with continuous river flow were N<sub>2</sub>O sources, while more than half of the sites with intermittent river flow were sinks.

Table 3 shows that CO<sub>2</sub> fluxes were significantly correlated between the wet season and dry season, while CH<sub>4</sub> and N<sub>2</sub>O fluxes were not correlated in two seasons.

Table 3 Significant correlations between GHGs fluxes and two seasons (n=31)

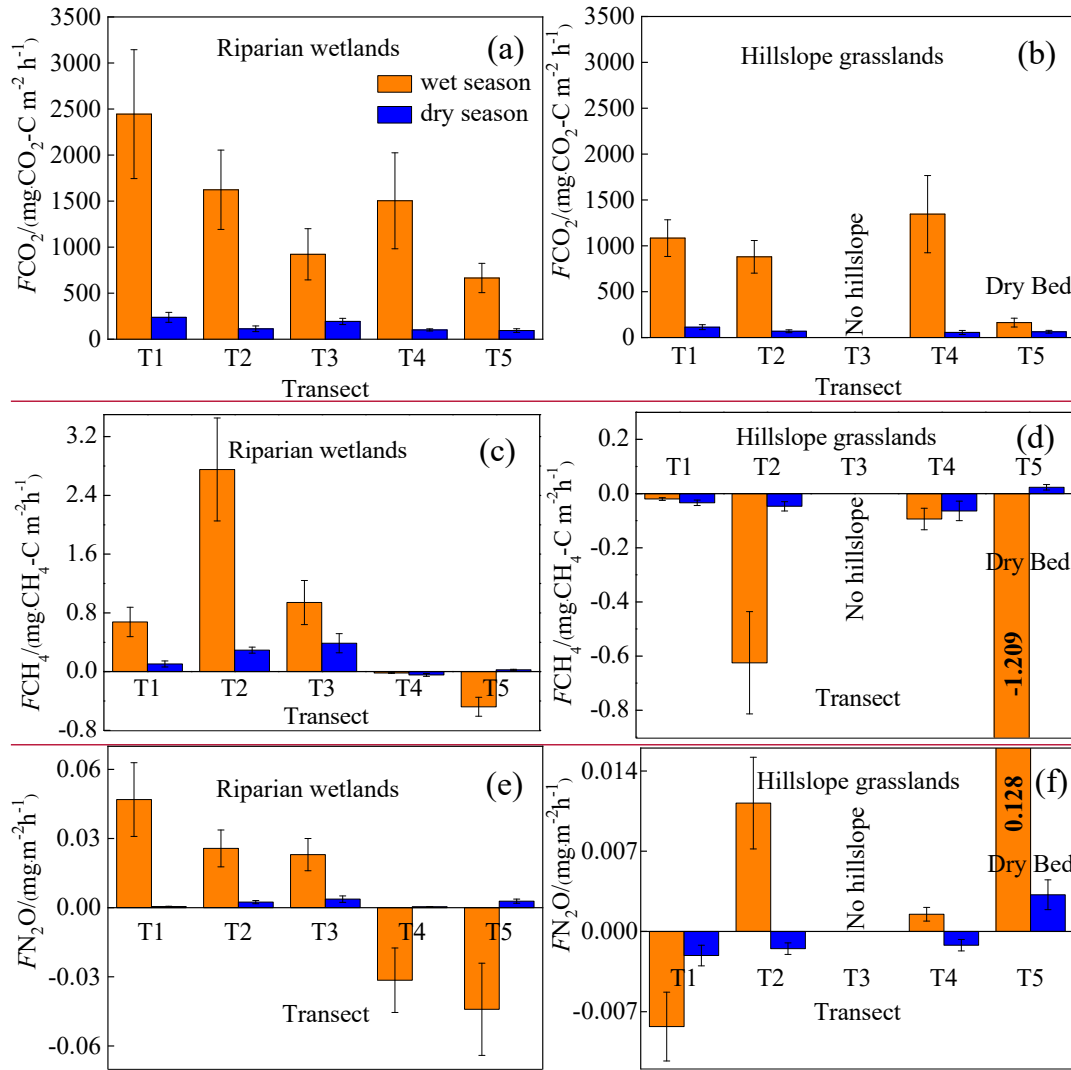
GHG flux	FCO <sub>2</sub> in wet season-FCO <sub>2</sub> in dry season	FCH <sub>4</sub> in wet season-FCH <sub>4</sub> in dry season	FN <sub>2</sub> O in wet season-FN <sub>2</sub> O in dry season
significant correlations (P)	0.000	0.133	0.290

Note: P<0.05 denote significant correlations and P > 0.05 denote no significant correlations

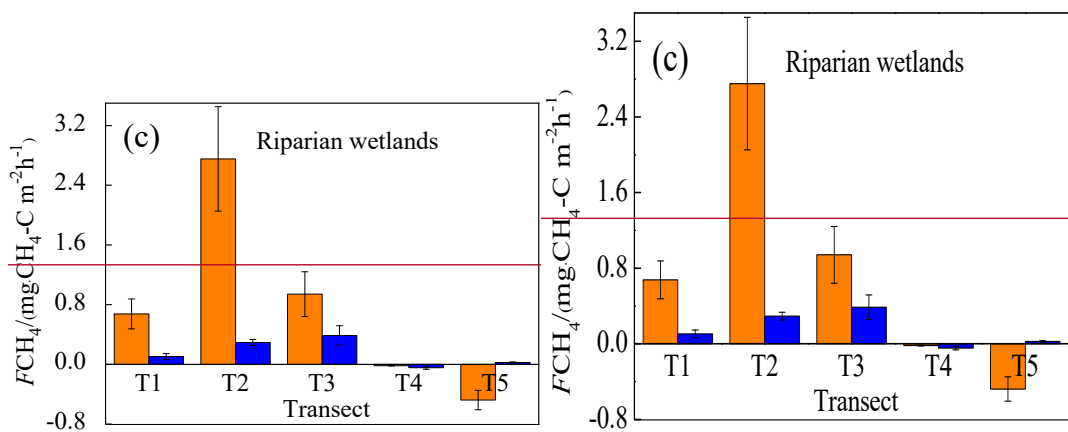
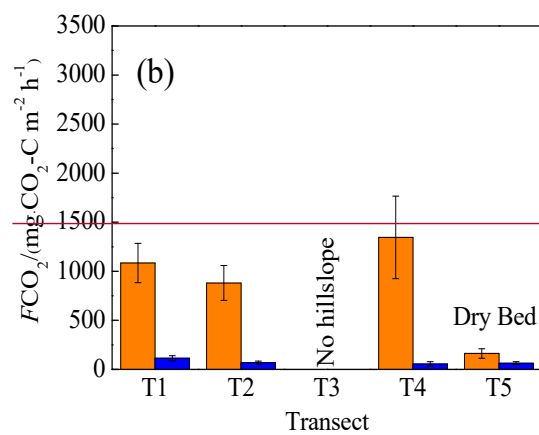
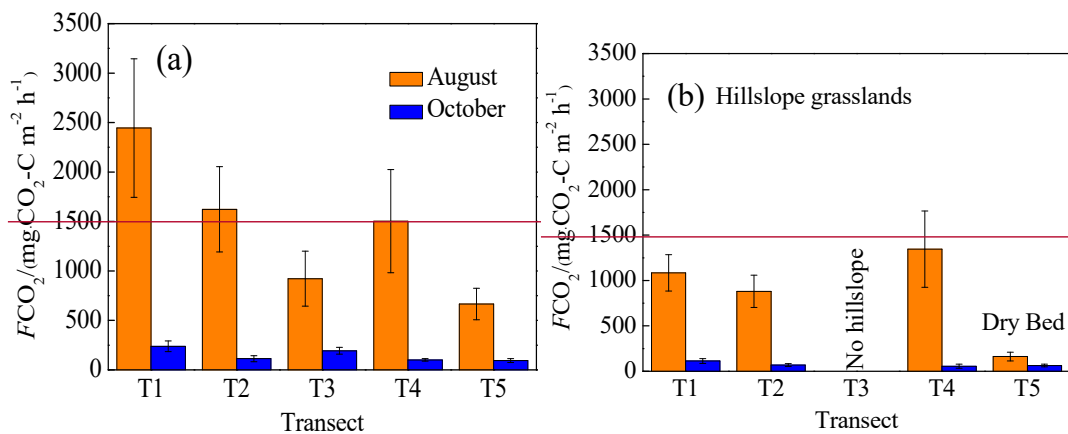
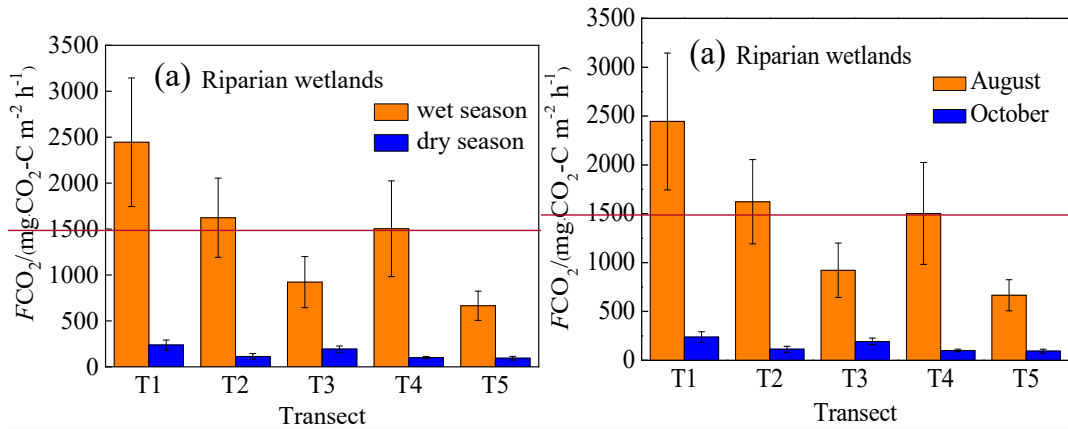
### 3.4 Spatiotemporal patterns of GHG emissions in upstream and downstream areas

Figure 6 shows the detailed spatial and seasonal distribution of GHG emissions in wet seasonAugust and dry seasonOctober in the longitudinal direction from the upstream (T1, T2, and

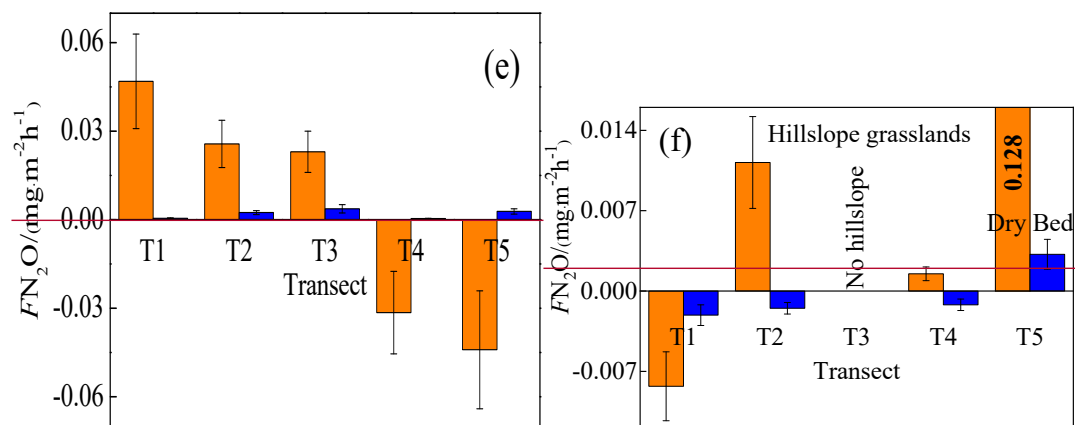
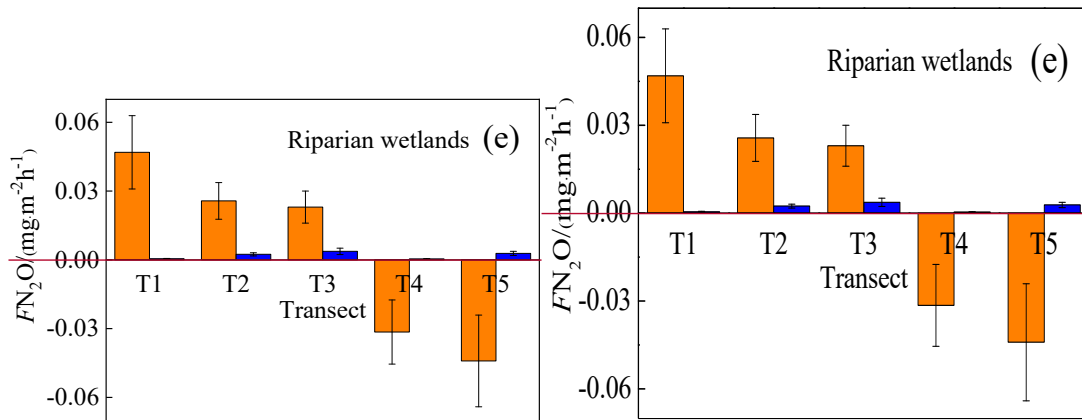
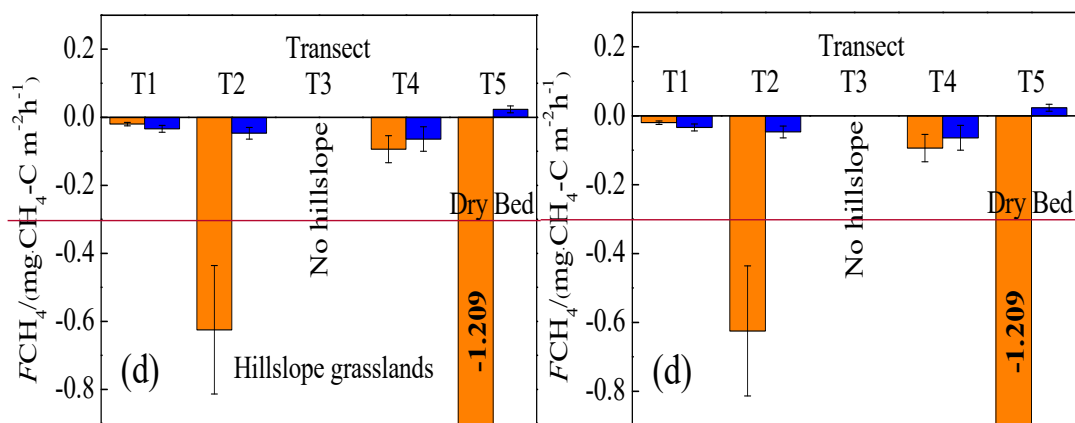
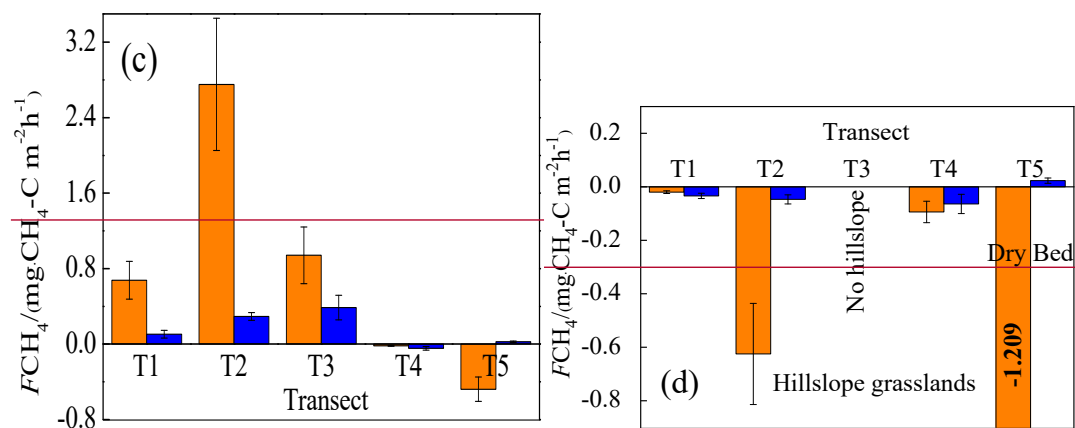
T3) to the downstream areas (T4 and T5). The CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O emissions were calculated from the average values of the respective emissions in the wetlands and hillslope grasslands in each transect.



**Fig. 6** Spatiotemporal patterns of CO<sub>2</sub> (first line), CH<sub>4</sub> (second line), and N<sub>2</sub>O (third line) emissions (F) in the upstream (T1, T2, and T3) and downstream areas (T4 and T5). Bars are the mean values for each transect and error bars are the standard errors.







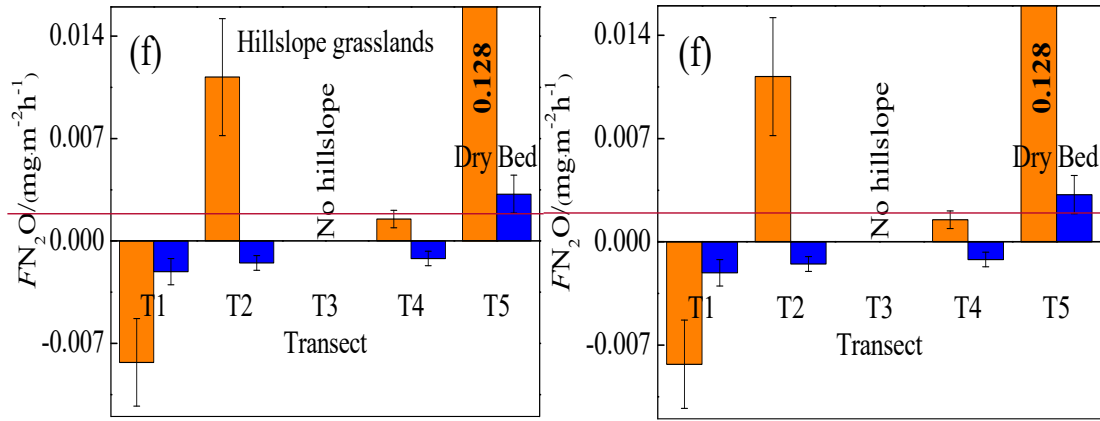


Fig. 6 Spatiotemporal patterns of CO<sub>2</sub> (first line), CH<sub>4</sub> (second line), and N<sub>2</sub>O (third line) emissions ( $F$ ) in the upstream (T1, T2, and T3) and downstream areas (T4 and T5). Data are shown for the riparian wetlands (a, c, e) and hillslope grasslands (b, d, f) in August (orange) and October (blue). Bars are the mean values for each transect and error bars are the standard errors.

CO<sub>2</sub> emissions in riparian wetlands (Fig. 6(a)) in wet seasonAugust decreased from  $2444.69 \pm 228.58 \text{ mg} \cdot \text{m}^{-2} \cdot \text{h}^{-1}$  in the upstream area to  $665.08 \pm 347.57 \text{ mg} \cdot \text{m}^{-2} \cdot \text{h}^{-1}$  downstream, and the corresponding values for dry seasonOctober were  $238.12 \pm 48.20 \text{ mg} \cdot \text{m}^{-2} \cdot \text{h}^{-1}$  and  $94.14 \pm 7.67 \text{ mg} \cdot \text{m}^{-2} \cdot \text{h}^{-1}$ . However, in hillslope grasslands (Fig. 6(b)), CO<sub>2</sub> emissions exhibited no significant seasonality between upstream and downstream areas, with the mean values of  $1103.40 \pm 190.44 \text{ mg} \cdot \text{m}^{-2} \cdot \text{h}^{-1}$  in wet seasonAugust and  $79.18 \pm 24.52 \text{ mg} \cdot \text{m}^{-2} \cdot \text{h}^{-1}$  in dry seasonOctober. In addition, CO<sub>2</sub> emissions in transect T5 were lower for both months, with the averages of  $162.83 \pm 149.15 \text{ mg} \cdot \text{m}^{-2} \cdot \text{h}^{-1}$  and  $63.26 \pm 12.40 \text{ mg} \cdot \text{m}^{-2} \cdot \text{h}^{-1}$  in wet seasonAugust and dry seasonOctober, respectively. The upstream riparian zones exhibited higher CO<sub>2</sub> emissions ( $894.32 \pm 868.47 \text{ mg} \cdot \text{m}^{-2} \cdot \text{h}^{-1}$ ) than their downstream counterparts ( $621.14 \pm 704.10 \text{ mg} \cdot \text{m}^{-2} \cdot \text{h}^{-1}$ ). However, mean CO<sub>2</sub> emissions showed no significant differences in grasslands, averaging  $524.16 \pm 450.10 \text{ mg} \cdot \text{m}^{-2} \cdot \text{h}^{-1}$  upstream and  $508.06 \pm 534.77 \text{ mg} \cdot \text{m}^{-2} \cdot \text{h}^{-1}$  downstream.

CH<sub>4</sub> emissions showed a marked spatial pattern in the riparian zones from upstream to downstream (Fig. 6(c)). The transects with continuous river flow were CH<sub>4</sub> sources in wet seasonAugust and dry seasonOctober, with the average emissions of  $1.42 \pm 3.41 \text{ mg} \cdot \text{m}^{-2} \cdot \text{h}^{-1}$  and  $0.27 \pm 0.49 \text{ mg} \cdot \text{m}^{-2} \cdot \text{h}^{-1}$ , respectively, while those with intermittent river flow served as CH<sub>4</sub> sinks, with the corresponding mean values of  $-0.21 \pm 0.45 \text{ mg} \cdot \text{m}^{-2} \cdot \text{h}^{-1}$  and  $-0.02 \pm 0.05 \text{ mg} \cdot \text{m}^{-2} \cdot \text{h}^{-1}$ .

Moreover, the hillslope grassland sites in all transects were CH<sub>4</sub> sinks (Fig. 6(d)).

N<sub>2</sub>O emissions in riparian wetlands (Fig. 7(e)) showed spatial patterns similar to those of CH<sub>4</sub> emissions. In wet season ~~August~~, the transects with continuous river flow served as N<sub>2</sub>O sources, with the mean value of  $0.031 \pm 0.031 \text{ mg} \cdot \text{m}^{-2} \cdot \text{h}^{-1}$ , while those with intermittent river flow were N<sub>2</sub>O sinks with an average value of  $-0.037 \pm 0.05 \text{ mg} \cdot \text{m}^{-2} \cdot \text{h}^{-1}$ . In dry season ~~October~~, N<sub>2</sub>O emissions occurred as weak sources in the longitudinal transects, averaging  $0.002 \pm 0.007 \text{ mg} \cdot \text{m}^{-2} \cdot \text{h}^{-1}$ . However, N<sub>2</sub>O emissions in hillslope grasslands did not show any spatial pattern (Fig. 7(f)).

## 4. Discussion

### 4.1 Main factors influencing GHG emissions

#### 4.1.1 Effects of SMC on GHG emissions

SMC constituted one of the main factors affecting GHG emissions in wetlands. In this study, transects T1–T4 were characterized by a marked spatial SMC gradient (i.e., a gradual decrease include SMC10 and SMC20 from the riparian wetlands to the hillslope grasslands and from upstream to downstream (Fig. 3)). The CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O emissions showed a similar trend. In Table 4, SMC10 is positive correlated with CO<sub>2</sub> emissions ( $P < 0.05$ ), SMC10 and SMC20 are significantly positive correlated with CH<sub>4</sub> emissions ( $P < 0.01$ ), and SMC10 and SMC20 are highly positive correlated with N<sub>2</sub>O emissions ( $P < 0.05$  and  $P < 0.01$ , respectively). These results indicated the influence of wetland SMC on GHG emissions.

Typically, the optimal SMC values associated with CO<sub>2</sub> emissions in riparian wetlands range from 40 to 60% (Sjögersten et al., 2006), creating better soil aeration and improving soil microorganisms' activity and the respiration of plant roots, thereby promoting CO<sub>2</sub> emissions, whereas excessive SMC reduces soil gas transfer due to the formation of an anaerobic environment in the soil, and microbial activity is lower, favoring the accumulation of organic matter (Hui., 2014). On the contrary, the SMC of hillslope grasslands is less than 10%. Low soil moisture inhibits the growth of vegetation with few vegetation residues and litters. Meanwhile, low soil moisture is not conducive to the survival of soil microorganisms, leading to a decrease in CO<sub>2</sub> emissions than to those in riparian zones (Moldrup et al., 2000; Hui., 2014). Similar results were obtained in our study. The changes in CO<sub>2</sub> emissions in transect T5 were contrary to the change in the SMC10 and SMC20 likely because the optimal range of soil C:N is between 10-12

(Pierzynski et al., 1994), but the value in the dry lake bed of T5 is higher than 60, high soil C:N resulted in nitrogen limitation in the process of decomposition of organic matter by microorganisms. Furtherly, other sediment properties (like Soil pH>9.5) for this transect were not conducive to the survival of microorganisms (Table 1), and the increase in SMC did not increase the respiration activity of microorganisms.

Typically, the optimal SMC values associated with CO<sub>2</sub> emissions in riparian wetlands range from 40 to 60% (Sjögersten et al., 2006). Increasing SMC generally promotes higher soil CO<sub>2</sub> concentrations (Riveros-Iregui et al. 2007) in response to increased production, decreased transport, and excessive SMC reduces soil gas transfer. On the contrary, the SMC of hillslope grasslands is less than 10%, leading to a decrease in CO<sub>2</sub> emissions than to those in riparian zones (Moldrup et al., 2000). Similar results were obtained in our study. The changes in CO<sub>2</sub> emissions in transect T5 were contrary to the change in the SMC<sub>10</sub> and SMC<sub>20</sub> likely because high soil C:N resulted in nitrogen limitation in the process of decomposition of organic matter by microorganisms, and other sediment properties (like Soil pH) for this transect were not conducive to the survival of microorganisms (Table 1), and the increase in SMC did not increase the respiration activity of microorganisms.

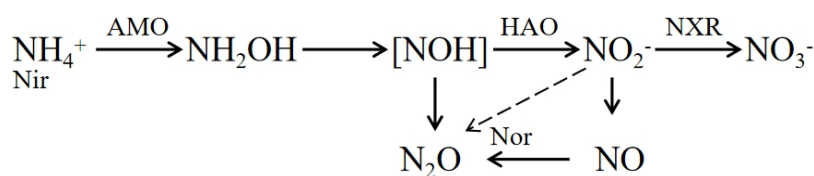
The largest CH<sub>4</sub> emissions were observed at the near-stream sites (i.e., L1 and R1) in T1, T2, and T3, with the average SMC of 30.29%, while the SMC values at the other sites, which were either weak sources or sinks, averaged at 14.57%. These results indicate that a higher SMC is favorable for CH<sub>4</sub> emissions because a higher SMC denotes a soil in a reduced state, which is beneficial for CH<sub>4</sub> production and inhibits CH<sub>4</sub> oxidation. A similar result was reported by Xu et al. (2008). They conducted experiments of CH<sub>4</sub> emissions from a variety of paddy soils in China, and showed that CH<sub>4</sub> production rates increased with the increase in SMC at the same incubation temperature. Meng et al. (2001) also reported that water depth was the main factor affecting CH<sub>4</sub> emissions from wetlands. When the water level dropped below the soil surface, the decomposition of organic matter accelerated, and CH<sub>4</sub> emissions decreased. If the oxide layer is large, the soil is transformed into a CH<sub>4</sub> sink (Meng et al., 2011).

The N<sub>2</sub>O fluxes showed a clear spatial pattern associated with the changes in SMC. The moisture content of wetland soils directly affects the aeration status of the soil. Besides, the aeration status affects the partial pressure of oxygen, which has an important impact on

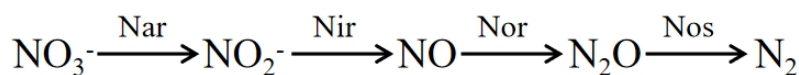
nitrifying/denitrifying bacteria's activity and ultimately affects soil N<sub>2</sub>O emissions (Zhang et al., 2005). Table 234 shows that N<sub>2</sub>O emissions are significantly positively correlated with SMC10 and SMC20 (P < 0.01). Generally, when SMC was below the saturated water content, the microorganisms were in an aerobic environment, and N<sub>2</sub>O mainly came from the nitrification reaction. N<sub>2</sub>O emissions increases with the increase of SMC (Niu et al., 2017; Yu et al., 2006). In our study, the sampling sites with higher SMC (riparian zones and some hillslope grassland zones in the upstream transects) have higher N<sub>2</sub>O emissions. When SMC increases to the saturated water content or is in a flooded state, the system was an anaerobic environment, and the Nos activity was higher due to excessively high SMC, which was conducive to denitrification and eventually produced N<sub>2</sub> (Niu et al., 2017; Yu et al., 2006), such as site L1 in transect T3 in this study. Ulrike et al. (2004) showed that denitrification was the main process under flooded soil conditions in wetland soils, and the release of N<sub>2</sub> exceeds N<sub>2</sub>O. These findings are consistent with those of Liu et al. (2003), who showed that SMC is an essential factor affecting N<sub>2</sub>O emissions.

The N<sub>2</sub>O fluxes showed a clear spatial pattern associated with the changes in SMC. When SMC was below the saturated water content, N<sub>2</sub>O emissions increased with the increase in SMC. In contrast, when soil mass water content SMC was above the saturated water content 100%, N<sub>2</sub>O emissions gradually decreased with the increase in SMC (Niu et al., 2017). In our study, N<sub>2</sub>O emissions in the riparian wetlands with a high SMC10 and SMC20 were higher than those in the hillslope grasslands, and N<sub>2</sub>O emissions in the upstream transects were higher than those downstream. These findings are consistent with those of Liu et al. (2003), who showed that SMC is an essential factor affecting N<sub>2</sub>O emissions.

#### Nitrification:



#### Denitrification:



The enzymes involved in the formula include Ammonia monooxygenase (AMO), Hydroxylamine oxidase (HAO), Nitrite REDOX enzyme (HAO), nitrate reductase (Nar), nitrite reductase (Nir), Nitric oxide reductase (Nor) and Nitrous oxide reductase (Nos).



#### 4.1.2 Effects of ST on GHG emissions

ST was another important factor affecting the CO<sub>2</sub> emissions in this study, as this parameter was significantly correlated with CO<sub>2</sub> emissions ( $P < 0.01$ ) (Table 234). The activity of soil microorganisms increases with rising soil temperatures, leading to increased respiration, and consequently higher CO<sub>2</sub> emissions (Heilman et al., 1999). Previous studies reported that ST partially controls seasonal CO<sub>2</sub> emission patterns (Inubushi et al., 2003). Therefore, CO<sub>2</sub> emissions in wet seasonAugust were significantly higher than those in dry seasonOctober in this study.

CH<sub>4</sub> emissions showed a clear seasonal pattern because high summer temperatures improve the activity of both CH<sub>4</sub>-producing and -oxidizing bacteria (Ding et al., 2010). ~~However, In this study, ST was closely related to CH<sub>4</sub> emissions, especially in the near-stream sites, and CH<sub>4</sub> emissions in August were 59 times higher than those in October. Conversely, seasonal fluctuations in CH<sub>4</sub> emissions at the downstream sampling sites were small.~~ Table 24 indicates that the correlation between CH<sub>4</sub> emissions and temperature is not significant because SMC could be more critical than temperature in our study region with very dry climate.~~the production and emission of CH<sub>4</sub> are significantly affected by water conditions.~~ SMC showed a positive correlation with GHG emissions. In addition, SMC affected ST to a certain extent, while the interactions between SMC and ST had a mutual influence on CH<sub>4</sub> emissions. During the study period, the near-stream sites (L1 and R1) maintained a super-wet state on the ground surface for a long time, which was beneficial for the production of CH<sub>4</sub>. However, the wetlands maintained a state without water accumulation on the soil surface in the growing seasonAugust, which was conducive to the oxidative absorption of CH<sub>4</sub>. SMC thus masked the effect of ST on CH<sub>4</sub> emissions.

Previous studies indicated that temperature is an important factor affecting N<sub>2</sub>O emissions (Sun et al., 2011) through primary mechanisms impacting the nitrifying and denitrifying bacteria in the soil. Table 234 shows that the correlations between N<sub>2</sub>O emissions and ST10 and ST20 are poor ( $P > 0.05$ ). This can be attributed to the wide suitable temperature range for nitrification-denitrification and weak sensitivity to temperature. Malhi et al. (1982) found that the optimum temperature for nitrification was 20 °C, and it will inhibit entirely at 30 °C. However, Brady (1999) believed that the suitable temperature range for nitrification was 25~35°C, and the nitrification inhibits below 5 °C or above 50 °C. It showed that the temperature requirements of

nitrifying microorganisms in wetland soils were different in different temperature belts. The suitable temperature range was the performance of the long-term adaptability of nitrifying microorganisms. Meanwhile, several studies revealed that denitrification could be carried out in a wide temperature range (5 ~ 70 °C), and it was positively related to temperature (Fan., 1995). However, the process will be inhibited when the temperature was too high or too low. The average ST in wet season August was 27.4°C, conducive to the growth of denitrifying microorganisms, while that in dry season October was 8.97°C, and the microbial activity was generally low (Sun et al., 2011). Furthermore, ST fluctuations were low both in wet season August and dry season October. Therefore, the effect of ST on N<sub>2</sub>O emissions was masked by other factors, such as moisture content. Previous studies indicated that temperature is an important factor affecting N<sub>2</sub>O emissions (Sun et al., 2011) through primary mechanisms impacting the activities of the nitrifying and denitrifying bacteria in soil. Table 2 shows that the correlations between N<sub>2</sub>O emissions and ST10 and ST20 are poor ( $P > 0.05$ ). This can be attributed to the wide suitable temperature range for nitrification-denitrification and weak sensitivity to temperature. The average ST in August was 27.4°C, conducive to the growth of denitrifying microorganisms, while that in October was 8.97°C, and the microbial activity was generally low (Sun et al., 2011). Furthermore, ST fluctuations were low both in August and October. Therefore, the effect of ST on N<sub>2</sub>O emissions was masked by other factors, such as moisture content.

#### 4.1.3 Effects of BIO and soil organic matter on GHG emissions

CO<sub>2</sub> and CH<sub>4</sub> emissions were higher in the riparian wetlands than in the grasslands, mainly because of greater vegetation cover. Typically, CO<sub>2</sub> emissions from riparian wetlands originate from plants and microorganisms, with plant respiration accounting for a large proportion in the growing season. Previous studies have shown that plant respiration accounts for 35–90% of the total respiration in the wetland ecosystem (Johnson-Randall and Foote, 2005). Good soil physicochemical properties and high soil total organic carbon (TOC) of riparian wetlands improve the activity of soil microorganisms and plant root respiration. Table 234 shows that BIO is significantly correlated with the CO<sub>2</sub> ( $P < 0.05$ ) and CH<sub>4</sub> ( $P < 0.01$ ) emissions. These results can be attributed to the significant linear positive correlation between the respiration rate and plant biomass (Lu et al., 2007). Higher plant biomass storage can achieve more carbon accumulation

during photosynthesis and higher exudate release by the roots. This, in turn, promotes the accumulation of soil organic matter. Increased amount of organic matter stimulates the growth and reproduction of soil microorganisms, ultimately promoting CO<sub>2</sub> and CH<sub>4</sub> emissions. Moreover, plants act as a gas channel for CH<sub>4</sub> transmission, and a larger amount of biomass promotes CH<sub>4</sub> emissions, given the increased number of channels. In transect T3, high CO<sub>2</sub> emissions observed at site L3 can be attributed to the relatively high levels of SMC, BIO, and soil nutrients~~soil nutrients~~TOC, which stimulate the microbial respiration rates.

BIO had a weak correlation with N<sub>2</sub>O emissions (Table 24), which indicates that plants increase N<sub>2</sub>O production and emissions, although this may not be the most critical factor. Previous studies reported mechanisms where in the plants can absorb N<sub>2</sub>O produced in the soil through the root system before releasing it into the atmosphere. Additionally, the root exudates of plants can enhance the activity of nitrifying and denitrifying bacteria in the soil, ultimately promoting the production of N<sub>2</sub>O. Finally, oxygen stress caused by plant respiration can regulate the production and consumption of N<sub>2</sub>O in the soil, eventually affecting the conversion of nitrogen in the soil (Koops et al., 1996; Azam et al., 2005).

Site L3 in transect T3 was covered by tall reeds, and its BIO was much higher than those of the other sites; thus, the data for this site were excluded from the correlation analysis.

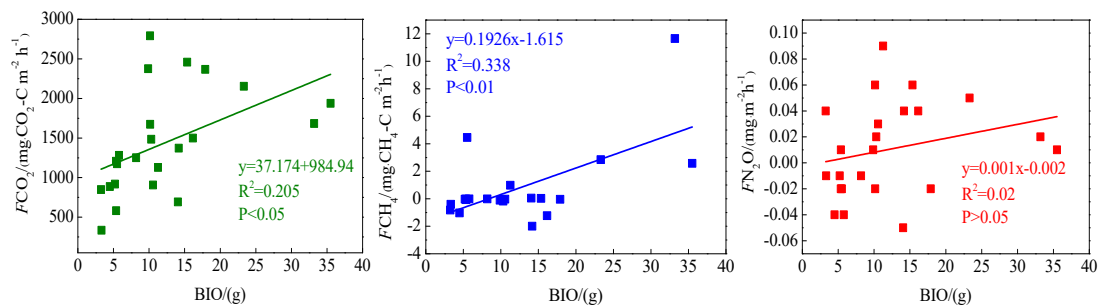


Fig. 7 Correlation between aboveground biomass (BIO) and GHG emissions (*F*)

~~The~~sSoil C:N ratio refers to the ratio of biodegradable carbonaceous organic matter and nitrogenous matter in the soil, and it forms the soil matrix with TOC. TOC decomposition provides energy for microbial activity, while the C:N ratio affects the decomposition of organic matter by soil microorganisms (Gholz et al., 2010). The correlation results (Fig. 8) indicate that

TOC has a weak positive correlation with CO<sub>2</sub> emissions ( $P > 0.05$ ), but soil C:N has a significant negative correlation with CO<sub>2</sub> emissions ( $P \leq 0.05$ ), indicating that nitrogen has a limiting effect on soil respiration by affecting microbial metabolism. Liu et al. (2019) reported that N addition promoted CO<sub>2</sub> emissions from wetlands soil, and the effect of organic N input was significantly higher than those of inorganic N input. Organic carbon provides a carbon source for the growth of plants and microorganisms, which boosts their respiration. Moreover, TOC has a significant correlation with N<sub>2</sub>O emissions ( $P < 0.05$ ). Most heterotrophic microorganisms use soil organic matter as carbon and electron donors (Morley N and Baggs E M., 2010). Soil carbon source has an important influence on microbial activity. Nitrifying or denitrifying microorganisms need organic matter to provide carbon source during the assimilation of NH<sub>3</sub> or NO<sub>3</sub><sup>-</sup>. The high content of organic matter in the soil can promote the abundance of heterotrophic nitrifying bacteria increases, consume dissolved oxygen in the medium, and cause the soil to become more anaerobic, slowing down autotrophic growth nitrifying bacteria. This reduces the nitrification rate, ultimately promoting N<sub>2</sub>O release. —Enwall et al. (2005) studied the effect of long-term fertilization on soil denitrification microbial action intensity. They found that the soil with long-term organic fertilizer application has a significant increase in organic matter content, and consequently, a significant increase in denitrification activity. As the increase of TOC, the abundance of heterotrophic nitrifying bacteria increases, consume dissolved oxygen in the mediums — and cause soils to become more anaerobic, slowing down the growth of autotrophic nitrifying bacteria. This reduces the nitrification rate, ultimately promoting N<sub>2</sub>O release. Typically, low soil C:N ratios are favorable for the decomposition of microorganisms, the most suitable range being between 10 and 12 (Pierzynski et al., 1994). Table 234 shows that N<sub>2</sub>O emissions are significantly related to the soil C:N ratios ( $P < 0.05$ ), which means that denitrifying bacteria will use their endogenous carbon source for denitrification when the external carbon source is insufficient. Moreover, incomplete denitrification leads to the accumulation of NO<sub>2</sub>-N, which is conducive to the N<sub>2</sub>O release. Meanwhile, due to the weak competitive ability of Nos to electrons, low C:N inhibits the synthesis of Nos, which is also a reason for N<sub>2</sub>O release. In this study, all the sites in transects T1–T4 exhibited similar soil C:N ratios in the optimum range (Table 1), which is favorable for microbial decomposition. However, the soil C:N ratios in transect T5 were higher than those in the other transects, especially in the dry lake bed. Therefore, transect T5 showed severe mineralization and

a low microbial decomposition rate.

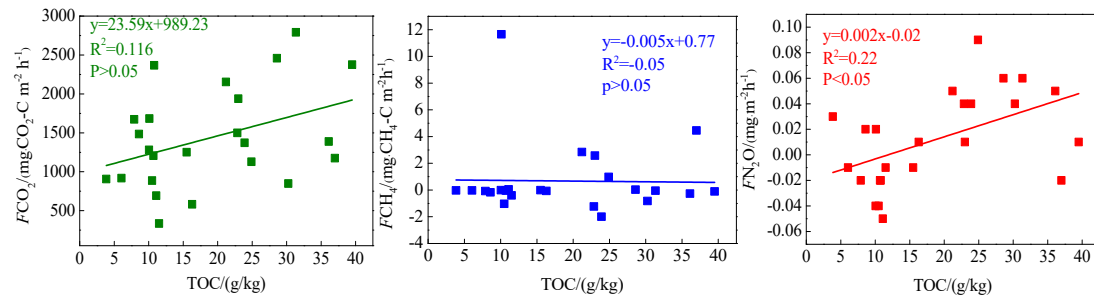


Fig. 8 Correlations between soil organic carbon (TOC) and GHG emissions ( $F$ )

Table 4. Correlations between  $\text{CO}_2$ ,  $\text{CH}_4$ , and  $\text{N}_2\text{O}$  emissions and impact factors ( $n = 62$ )

GHG flux	ST10	ST20	SMC10	SMC20	TOC	$\rho_b$	C:N	pH	EC	BIO
$\text{CO}_2$	0.634**	0.592**	0.307*	0.216	0.393	-0.463**	-0.289*	-0.350**	-0.251*	0.491*
$\text{CH}_4$	-0.029	-0.051	0.346**	0.353**	-0.02	-0.129	-0.156	-0.127	-0.107	0.607**
$\text{N}_2\text{O}$	0.127	0.118	0.304*	0.356**	0.493*	-0.194	0.311*	0.137	0.504**	0.251

Note: 1. The analysis method used in the table is Pearson correlation analysis, and the numbers represent Pearson correlation coefficients.

2. \* and \*\* denote significant and highly significant correlations ( $P < 0.01$  and  $P < 0.05$ ), respectively.

2.3. ST - soil temperature, SMC - soil moisture content,  $\rho_b$  - soil bulk density, soil C:N - soil carbon-nitrogen ratio, pH - soil pH, EC - soil electrical conductivity, BIO - aboveground biomass

Table 2. Correlations between  $\text{CO}_2$ ,  $\text{CH}_4$ , and  $\text{N}_2\text{O}$  emissions and impact factors ( $n = 62$ )

GHG flux	ST10	ST20	SMC10	SMC20	TOC	$\rho_b$	C:N	pH	EC	BIO
$\text{CO}_2$	0.634**	0.592**	0.307*	0.216	0.393	-0.463**	-0.289*	-0.350**	-0.251*	0.491*
$\text{CH}_4$	-0.029	-0.051	0.346**	0.353**	-0.02	-0.129	-0.156	-0.127	-0.107	0.607**
$\text{N}_2\text{O}$	0.127	0.118	0.304*	0.356**	0.493*	-0.194	0.311*	0.137	0.504**	0.251

Note: 1. \* and \*\* denote significant and highly significant correlations ( $P < 0.01$  and  $P < 0.05$ ), respectively.

2. ST - soil temperature, SMC - soil moisture content,  $\rho_b$  - soil bulk density, soil C:N - soil carbon-nitrogen ratio, pH - soil pH, EC - soil electrical conductivity, BIO - aboveground biomass

## 4.2 Riparian wetlands as hotspots of GHG emissions

The results of this study emphasized that CO<sub>2</sub> emissions in the riparian wetlands were higher than those in the hillslope grasslands owing to a variety of factors. ST is an important factor affecting GHG emissions. McLain and Martens (2006) showed that seasonal fluctuations in ST and SMC in semi-arid regions have important effects on CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O emissions in riparian soils. Poblador et al. (2017) studied the GHG emissions in forest riparian zones and suggested that the difference in the CO<sub>2</sub> and N<sub>2</sub>O emissions in these zones is affected by the spatial gradient of the regional SMC. In this study, the upstream riparian wetlands are characterized by higher TOC, lower soil C:N ratio, and abundant BIO than the hillslope grasslands (Table 1). These soil conditions benefited the soil microbial activity, ultimately enhancing respiration as well as CO<sub>2</sub> emissions. However, CO<sub>2</sub> emissions in downstream areas were nearly identical to those in the grasslands because the wetlands gradually evolved into grasslands after their degradation. The N<sub>2</sub>O emissions showed spatial patterns similar to those of the CO<sub>2</sub> emissions because the CO<sub>2</sub> concentrations were closely related to nitrification and denitrification processes. High CO<sub>2</sub> concentrations can promote the carbon and nitrogen cycles in soil (Azam et al., 2005), ~~providing the substrate and energy required for the nitrification and denitrification reactions. increasing below ground C allocation associated with increased root biomass, root turnover, and root exudation in elevated pCO<sub>2</sub> plants provided the energy for denitrification in the presence of high available N, or that there was increased O<sub>2</sub> consumption under elevated pCO<sub>2</sub> (Baggs et al., 2003).~~ Moreover, soil respiration increases during soil denitrification (Liu et al., 2010; Christensen et al., 1990). In this study, a weak correlation was observed between the CO<sub>2</sub> and CH<sub>4</sub> emissions in the riparian zones ( $r = 0.228$ ), but CO<sub>2</sub> emissions were significantly correlated with N<sub>2</sub>O emissions ( $r = 0.322$ ,  $P < 0.05$ ). The soil became anaerobic in the riparian areas as the SMC increased, and this was conducive to the survival of CH<sub>4</sub>-producing bacteria and denitrification reactions, eventually leading to an increase in CH<sub>4</sub> and N<sub>2</sub>O emissions. Jacinthe et al. (2015) reported that inundated grassland-dominated riparian wetlands were CH<sub>4</sub> sinks ( $-1.08 \pm 0.22 \text{ kg} \cdot \text{CH}_4\text{-C ha}^{-1} \cdot \text{yr}^{-1}$ ), ~~but~~ and Lu et al. (2015) also indicated that grasslands were CH<sub>4</sub> sinks. In our study, a marked water gradient across the transects led to the transformation of the soil from anaerobic to aerobic soil, which changed the wetland function as a CH<sub>4</sub> source or sink. Therefore, during the transition from

the riparian wetlands to the hillslope grasslands, CH<sub>4</sub> emissions only appeared as sources in the near-stream sites and sinks at other sites.

Further, we compared the GHG emissions of riparian wetlands and hillslope grasslands around the Xilin River Basin with various types of grasslands (meadow grassland, typical grassland, and desert grassland) in the Xinlingol League in Inner Mongolia (Table 35). The CO<sub>2</sub> emissions in wet season ~~August~~ decreased in the following order: upstream riparian wetlands > downstream riparian wetlands > hillslope grasslands > meadow grassland > typical grassland > desert grassland. Moreover, the upper riparian wetlands acted as ~~the~~-source of CH<sub>4</sub> emissions, while the downstream transects and grasslands served as CH<sub>4</sub> sinks. Similarly, except for the downstream transects, N<sub>2</sub>O emissions occurred as weak sources in different types of grasslands and upstream riparian wetlands. The GHG emissions showed similar spatial patterns in October. Although these estimates were made only in the growing season in August and the non-growing season in October, our results suggest that riparian wetlands are the potential hotspots of GHG emissions. Thus, it is important to study GHG emissions to obtain a comprehensive picture of the role of riparian wetlands in climate change.

Table 35. GHG emission fluxes of riparian wetlands and grasslands

Sample-plot	GHG-emissions-in-August- (mg·m <sup>-2</sup> ·h <sup>-1</sup> )			GHG-emissions-in-October- (mg·m <sup>-2</sup> ·h <sup>-1</sup> )			Reference
	CO <sub>2</sub>	CH <sub>4</sub>	N <sub>2</sub> O	CO <sub>2</sub>	CH <sub>4</sub>	N <sub>2</sub> O	
Wetlands- of upstream- transects- (T1, T2, and T3)	<u>n=13</u>	1606.28	1.417	0.031	182.35	0.272	0.002
Wetlands- of downstrea m- transects- (T4 and- T5)	<u>n=7</u>	1144.15	=0.215	=0.037	98.13	=0.015	0.001
Hillslope- grasslands	<u>n=7</u>	1103.40	=0.246	0.001	79.18	=0.048	=0.002

This study



of all transects							
Meadow grassland	166.39	=0.038	0.002	-	-	-	
Typical grassland	240.32	=0.042	0.037	-	-	-	Guo et al., 2017
Desert grassland	107.59	=0.036	0.003	-	-	-	
Typical grassland	520.25	=0.102	0.007	88.34	=0.099	0.005	Zhang, 2019
Typical grassland	232.42	=0.090	0.004	-	-	-	Chao, 2019
Typical grassland	265.23	=0.185	0.005	189.41	=0.092	0.004	
Meadow grassland	553.85	=0.163	0.003	47.73	=0.019	0.011	Geng, 2004
Typical grassland	308.60	=0.105	0.002	70.25	=0.029	0.007	
Sample plot	GHG emissions in August (mg·m <sup>-2</sup> ·h <sup>-1</sup> )			GHG emissions in October (mg·m <sup>-2</sup> ·h <sup>-1</sup> )			Reference
	CO <sub>2</sub>	CH <sub>4</sub>	N <sub>2</sub> O	CO <sub>2</sub>	CH <sub>4</sub>	N <sub>2</sub> O	
Wetlands of upstream transects (T1, T2, and T3)	n=13 1606.28 ± 697.78	1.417 ± 3.41	0.031 ± 0.03	182.35 ± 88.26	0.272 ± 0.49	0.002 ± 0.005	
Wetlands of downstream transects (T4 and T5)	n=7 1144.15 ± 666.50	-0.215 ± 0.45	-0.037 ± 0.05	98.13 ± 15.11	-0.015 ± 0.05	0.001 ± 0.01	This study
Hillslope grasslands of all transects	n=7 1071.54 ± 225.39	-0.300 ± 0.40	0.003 ± 0.03	77.68 ± 25.32	-0.048 ± 0.03	-0.002 ± 0.005	
Meadow grassland	166.39 ± 45.89	-0.038 ± 0.009	0.002 ± 0.001	=	=	=	
Typical grassland	240.32 ± 87.56	-0.042 ± 0.025	0.037 ± 0.034	=	=	=	Guo et al., 2017
Desert grassland	107.59 ± 54.10	-0.036 ± 0.015	0.003 ± 0.001	=	=	=	
Typical grassland	520.25 ± 59.07	-0.102 ± 0.012	0.007 ± 0.001	88.34 ± 9.84	-0.099 ± 0.003	0.005 ± 0.001	Zhang, 2019
Typical grassland	232.42 ± 18.90	-0.090 ± 0.005	0.004 ± 0.001	=	=	=	Chao, 2019
Typical grassland	265.23 ± 31.43	-0.185 ± 0.018	0.005 ± 0.001	189.41 ± 28.96	-0.092 ± 0.012	0.004 ± 0.001	

<u>Meadow grassland</u>	<u>553.85</u>	<u>-0.163</u>	<u>0.003</u>	<u>47.73</u>	<u>-0.019</u>	<u>0.011</u>
<u>Typical grassland</u>	<u>308.60</u>	<u>-0.105</u>	<u>0.002</u>	<u>70.25</u>	<u>-0.029</u>	<u>0.007</u>

Geng, 2004

We roughly estimated the annual cumulative emissions of CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O from riparian wetlands and hillslope grasslands around the Xilin River Basin, and further calculated its global warming potential. Table 6 indicated that annual cumulative emissions of CO<sub>2</sub> and CH<sub>4</sub> decreased in the following order: upstream riparian wetlands > downstream riparian wetlands > hillslope grasslands, and N<sub>2</sub>O in the following order: upstream riparian wetlands > hillslope grasslands > downstream riparian wetlands. In this study, we used the static dark-box method to measure CO<sub>2</sub> emissions, which does not consider the absorption and fixation of CO<sub>2</sub> by plants' photosynthesis. Therefore, the total annual cumulative CO<sub>2</sub> emissions are high. This result clearly showed that CO<sub>2</sub> contributed more than CH<sub>4</sub> and N<sub>2</sub>O to global warming. The GWP depends on the cumulative emissions of the GHGs. GWP is shown as (Table 6): upstream riparian wetlands (13474.91 kg/hm<sup>2</sup>) > downstream riparian wetlands (8974.12 kg/hm<sup>2</sup>) > hillslope grasslands (8351.24 kg/hm<sup>2</sup>). Therefore, both riparian wetlands and grasslands are the “sources” of GHGs on a 100-year time scale. The source strength of wetlands is higher than grasslands, further indicating that riparian wetlands are the hotspots of GHG emissions.

Table 6 Cumulative annual emission flux and global warming potential of GHGs in riparian wetlands and grasslands

<u>Sample plot</u>	<u>CO<sub>2</sub>/kg/hm<sup>2</sup></u>	<u>CH<sub>4</sub>/kg/hm<sup>2</sup></u>	<u>N<sub>2</sub>O/kg/hm<sup>2</sup></u>	<u>GWP/CO<sub>2</sub> kg hm<sup>2</sup></u>
<u>Wetlands of upstream transects (T1, T2, and T3)</u>	<u>13092.8±5378.16</u>	<u>12.36±26.40</u>	<u>0.25±0.23</u>	<u>13474.91±5828.68</u>
<u>Wetlands of downstream transects (T4 and T5)</u>	<u>9093.47±4831.82</u>	<u>-1.68±3.23</u>	<u>-0.26±0.40</u>	<u>8974.12±4912.75</u>
<u>Hillslope grasslands of all transects</u>	<u>8412.26±1614.26</u>	<u>-2.55±3.12</u>	<u>0.01±0.20</u>	<u>8351.24±1648.22</u>

#### 4.3 Effects of riparian wetland degradation on GHG emissions

–The hydrology and soil properties showed more evident differences among the transects

because the downstream zone was dry all year due to the presence of the Xilinhote Dam (Fig. 1). The dam caused the degradation of the riparian wetlands, resulting in reduced GHG emissions. The average CO<sub>2</sub> emissions amounted to 1663 mg·m<sup>-2</sup>·h<sup>-1</sup> in the riparian wetlands in the upstream transects (T1, T2, and T3), while the downstream transects (T4 and T5) recorded an average of 1084 mg·m<sup>-2</sup>·h<sup>-1</sup>, 35% lower than the value in the upstream transects. The N<sub>2</sub>O emissions from the riparian wetlands were lower in the downstream transects.

The wetland degradation first resulted in the continuous reduction of SMC, which led to the deepening of the wetland's aerobic layer thickness. Besides, SMC could affect ST's change and thus transformed CH<sub>4</sub> emissions from a source to a sink by affecting methanogens' activity (Yan et al., 2018). Secondly, the reduction of SMC impeded aboveground plants' physiological activities and inhibited related enzymes' activities in the respiration process. Meanwhile, various enzyme reactions of underground microorganisms under water stress influence and reduced CO<sub>2</sub> emissions (Zhang et al., 2017). Finally, after wetland degradation, long-term drought caused too low SMC, which was not conducive to the growth of nitrifying and denitrifying bacteria, which caused the transformation of N<sub>2</sub>O emissions from source to sink (Zhu et al., 2013). When the soil environment became less moist and more aerobic, the CH<sub>4</sub> and N<sub>2</sub>O emissions changed as the environment was transformed from a source to a sink, and CO<sub>2</sub> emissions decreased. Table 1 shows that soil TOC in the upstream transects (average: 25.1 g·kg<sup>-1</sup>) is higher than that in the downstream transects (average: 8.41 g·kg<sup>-1</sup>). The relatively low SMC and the aerobic environment were conducive to the mineralization and decomposition of TOC. The degradation of plants in the wetlands led to the gradual reduction of BIO. Ultimately, the plant carbon source input of the degraded wetlands decreased, and the bare land temperature increased due to the reduced plant shelter. This accelerated the decomposition of TOC, leading to its decrease. This result indicates that wetland degradation caused the soil carbon pool's loss and weakened the wetland carbon source/sink function. These results are in agreement with those of Xia (2017).

~~The relatively low SMC and the aerobic environment were conducive to the mineralization and decomposition of TOC. The degradation of plants in the wetlands led to the gradual reduction of BIO. Ultimately, the plant carbon source input of the degraded wetlands decreased and the bare land temperature increased due to the reduced plant shelter. This accelerated the decomposition of TOC, leading to its decrease.~~

The degraded wetlands also caused soil desertification and salinization, leading to a decline in the physical protection afforded by organic carbon and a reduction in soil aggregates. Thus, the preservation provided by organic carbon declined. Soil TOC and SMC in the dry lake bed in transect T5 were relatively high, but GHG emissions were very low along this transect because soil pH values increased after the degradation of the lake soil, exceeding the optimum range required for microorganism activity. The soil C:N ratio was very high, resulting in severe mineralization and a low microbial decomposition rate, hence affecting the GHG emissions.

## 5. Conclusions

The riparian wetlands in the Xilin River Basin constitute a dynamic ecosystem. The present spatial and temporal transfers in the studied biogeochemical processes were attributed to the changes in SMC, ST, and soil substrate availability. Our simultaneous analysis of CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O emissions from riparian wetlands and hillslope grasslands in the Xilin River Basin revealed that the majority of the GHG emissions occurred in the form of CO<sub>2</sub>. Moreover, our results clearly illustrated a marked seasonality and spatial pattern of GHG emissions along the transects and in the longitudinal direction (i.e., upstream and downstream). SMC and ST were two critical factors controlling the GHG emissions. Moreover, abundant BIO promoted the CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O emissions.

The riparian wetlands were the potential hotspots of GHG emissions in the Inner Mongolian region. However, the degradation of wetlands transformed the area from a source to a sink for CH<sub>4</sub> and N<sub>2</sub>O emissions, and reduced CO<sub>2</sub> emissions, which severely affected the wetland carbon cycle processes. Overall, our study suggests that anthropogenic activities have significantly changed the hydrological characteristics of the studied area, and will accelerate carbon loss from the riparian wetlands and further influence the GHG emissions in the future.

## Author Contributions

Xinyu Liu, Xixi Lu and Ruihong Yu designed the research framework and wrote the manuscript. Xixi Lu and Ruihong Yu supervised the study. Xinyu Liu, Hao Xue, Zhen Qi, Zhengxu Cao and Zhuangzhuang Zhang carried out the field experiments and laboratory experiments analyses. Z.Z. drew GIS mapping in this paper. Tingxi Liu proofread the manuscript.

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### **Competing interests**

The authors declare no conflicts of interest.

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