

## Biochar decreased N loss from paddy ecosystem under alternate wetting and drying in the Lower Liaohe River Plain, China

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### ABSTRACT

Biochar addition to soil is widely utilized to enhance carbon sequestration and reduce fertilizer N losses. However, little research has been studied on the effect of biochar on reactive gaseous N losses, N leaching and grain yield in paddy ecosystems under water stress, especially in the Lower Liaohe River Plain with a higher water percolation. Our experiment was carried out in 2020 and 2021 utilizing a split-plot design with continuously flooding irrigation and alternate wetting and drying irrigation as main plots and without biochar addition and with 20 t·ha<sup>-1</sup> rice husk-derived biochar addition as sub-plots. The results showed that alternate wetting and drying irrigation respectively, decreased N leaching and reactive N losses by 15.9 % and 11.3 % but also respectively, increased seasonal cumulative NH<sub>3</sub> volatilization and N<sub>2</sub>O emissions by 5.0 % and 21.0 % on average. Rice husk-derived biochar addition significantly mitigated seasonal cumulative NH<sub>3</sub> volatilization and N<sub>2</sub>O emissions by 8.8 % and 19.7 % in 2020, 20.7 % and 19.2 % in 2021, respectively, and decreased inorganic N leaching and reactive N losses by 8.3 % and 14.1 % in 2021. Biochar addition coupling with alternate wetting and drying respectively, mitigated cumulative NH<sub>3</sub> volatilization and N<sub>2</sub>O emissions by 7.3 % and 19.3 % in 2020, and, 22.7 % and 22.0 % in 2021 as compared to that without biochar. Biochar did not differ from without biochar in inorganic N leaching under alternate wetting and drying irrigation in both years but significantly reduced reactive N losses by 17.8 % in 2021, which efficiently inhibited the alternate wetting and drying induced negative effects on the increase in reactive N losses. Therefore, biochar addition to paddy ecosystems under alternate wetting and drying could realize sustainable utilization of water resources, increase soil N fixation, and mitigate N losses.

### 1. Introduction

Rice paddy ecosystems play a vital role in food security. It also accounts for approximately 19 % of the energy consumed on the planet (Wei et al., 2022). In rice production, many farmers apply a higher amount of nitrogen (N) fertilizer than required to maximize expected grain yield, especially in China (Dey et al., 2023; Huang et al., 2015; Ma et al., 2008). However, excessive and irrational N fertilizer application causes more reactive N losses such as ammonia (NH<sub>3</sub>) volatilization, nitrous oxide (N<sub>2</sub>O) emission, N leaching and runoff losses to the environment (Shang et al., 2024; Zhang et al., 2012). Reactive N losses through NH<sub>3</sub>, the most common source of N losses in paddy fields, accounts for 9–39 % of total N losses (Chen et al., 2015a), always resulting

in soil acidification, air pollution and indirect enhancement of greenhouse gas N<sub>2</sub>O (Aneja et al., 2020). N<sub>2</sub>O is another important greenhouse gas and a major contributor to global warming after methane and carbon dioxide, approximately having 273-fold the global warming potential of carbon dioxide (CO<sub>2</sub>) in a 100-year time period, which stays in the atmosphere for more than 116 years with the biggest human-related threat to the ozone layer (Intergovernmental Panel on Climate Change IPCC, 2023). N leaching accounts for 0.3–3.1 % of farmland N input, increasing farmer expenses and causing groundwater contamination and water eutrophication (Galloway et al., 2008). It is therefore vital to explore solutions to reduce reactive N losses in paddy fields, decrease environmental pollution produced by rice production, and develop efficient and sustainable agriculture.

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Rice is a highly freshwater-consuming crop most of which is managed by continuous flooding irrigation ( $I_{CF}$ ), which requires approximately 60 % of agricultural water in China (Jiang et al., 2017).  $I_{CF}$  requires a substantial amount of freshwater resources, increasingly receiving environmental concerns regarding its sustainable production and the greenhouse effect. Alternate wetting and drying irrigation ( $I_{AWD}$ ) can substantially decrease rice freshwater usage and methane ( $CH_4$ ) emissions without decreasing grain yield (LaHue et al., 2016). However, the nitrification and denitrification processes enhanced through the soil drying-wetting environment under  $I_{AWD}$  stimulate  $N_2O$  emissions, which increased  $N_2O$  emissions by 2–5-fold, compared to  $I_{CF}$  (Lagomarsino et al., 2016). In terms of  $NH_3$  volatilization, frequent dry-wet alternations increased soil pH and the level of  $NH_4^+-N$  in surface water, which emitted higher  $NH_3$  volatilization (Chen et al., 2022). In addition, less irrigation under  $I_{AWD}$  increased the concentration of  $NH_4^+-N$  and  $NO_3^-N$  in leaching, which in turn raises the inorganic N leaching (Liang et al., 2017). Hence, effective strategies are needed to mitigate the negative effects of  $I_{AWD}$  technology on N losses in paddy ecosystems. The Lower Liaohe River Plain is formed by alluvial deposits of the Liaohe River and is one of the key grain production centers in China (Wanguang et al., 2018; Wu et al., 2024). The soil type in the Lower Liaohe River Plain is mostly classified as clay loam on the surface (Zhao et al., 2009), and the 0.5–1 m deep soil is classified as sandy or sandy loam, which resulted in a higher water percolation and nutrient leaching (Zhao et al., 2009, 2015). Recently, our studies also verified that higher potassium (K) leaching in the Lower Liaohe River Plain was consistently observed than other areas (Han et al., 2023). Therefore, a deeper exploration of the relationship between N losses and irrigation regimes is essential for unraveling gaseous and leached N loss dynamics and characteristics under  $I_{AWD}$ , availing local decision makers of a benefit to provide a new strategy for reducing N losses and optimizing rice production in the lower Liaohe Plain.

Biochar is a carbon-based substance produced through the pyrolysis of agricultural waste under anoxic or anaerobic circumstances (Sun et al., 2021). The addition of biochar to paddy fields can absorb  $NH_4^+-N$  in the soil and alter the physical characteristics of the soil, therefore minimizing N losses and increasing N fertilizer efficiency (Yuan et al., 2018). Biochar addition decreased soil nitrification and denitrification substrate concentration, and increased soil pH could mitigate  $N_2O$  emissions (Singh et al., 2010). Moreover, biochar could absorb the  $NH_4^+-N$  and  $NO_3^-N$  in soil and thereby decrease inorganic N leaching (Wang et al., 2017). However, biochar addition to the sandy soil increased N leaching by 57 % (Bruun et al., 2012). Regarding  $NH_3$  volatilization, biochar addition reduced  $NH_3$  volatilization due to increased soil  $NH_4^+-N$  and decreased  $NH_4^+-N$  concentration in surface water (Sun et al., 2019; Mandal et al., 2016). In contrast, Feng et al. (2017) reported that biochar treatment significantly increased soil pH, and emitted 1.4–1.7 times higher  $NH_3$  volatilization compared to biochar-free control. Therefore, the effect of biochar on  $NH_3$  volatilization and N leaching is still debated, and advancing our knowledge in this domain is not only crucial for better understanding of the impacts of biochar on alternate wetting and drying based gaseous and leached N loss dynamics and characteristics, but also for the development of sustainable agricultural practices that can mitigate  $NH_3$  volatilization and N leaching under  $I_{AWD}$  in the Lower Liaohe River Plain, by extension, on broader alluvial plain area. Therefore, we conducted a 2-year field experiment in the lower Liaohe Plain to evaluate the response of gaseous and leached N losses to irrigation management and biochar addition from paddy ecosystems.

## 2. Material and methods

### 2.1. Site description and materials

A 2-year experiment was conducted at the Liaoning Irrigation Experiment Center Station in Huangjia Town, Shenyang, Liaoning

Province (42°08' 59" N, 120°30' 44" E, 47 m) during the 2020 and 2021 rice growth period (May to October). The experimental site was located in the temperate continental monsoon climate region with an average annual temperature of 7.5 °C and an average annual precipitation of 673 mm. The meteorological data from transplanting to harvest in both years are shown in Fig. S1. A Japonica rice (cv. Shennong 9765) was used and manually transplanted with three seeding plants per hill of 30 cm row x 14 cm plant spacing. Rice was transplanted on May 28, 2020, and May 26, 2021, and harvested on October 8, 2020, and October 9, 2021, respectively. Biochar was incorporated into topsoil (0–20 cm) with basal fertilizer. The topsoil is classified as clay loam, and the 1 m deep soil is classified as sandy. The physicochemical properties of topsoil (0–20 cm depth) are listed in Table S1. Biochar was prepared from rice husk at a pyrolysis temperature of 500 °C for 2 h. The properties of biochar are presented in Table S1.

### 2.2. Experimental design

The experiment was a split-plot design with two irrigation regimes, two biochar additions and three replications in 2020 and 2021. Irrigation regimes (I) included  $I_{CF}$  and  $I_{AWD}$ . Within each main plot, sub-plots were subjected to two biochar (B) additions (biochar-free control and 20 t ha<sup>-1</sup> biochar, represented by  $B_0$  and  $B_{20}$ , respectively). Each main plot was divided by 100 cm wide ridges of soil. Each sub-plot was separated by a 40 cm depth polyvinyl chloride barrier, of which 30 cm was inserted in soil to restrict the horizontal flow of nutrients. Each sub-plot was measured 3 m in width and 6 m in length. To control soil moisture, each plot had its water supply line and meter. The experimental field layout is shown in Fig. S2.

Regardless of irrigation regimes, water level was maintained within the range of 10–30 mm in the first 7–10 days following transplanting to aid seedling recovery. Thereafter, a standing water level of 10–70 mm was constantly maintained after transplanting until about 15 days before harvest under the  $I_{CF}$ . While under  $I_{AWD}$ , the plots were re-flooded until the soil water potential was dropped to –10 kPa at the 15 cm soil depth. About two weeks before harvest, the  $I_{AWD}$  cycle stopped. Water level was automatically monitored and recorded using an ultrasonic water level gauge (AODC-SY001-APD2, Fujian AUTEC Factory Automation Equipment CO.Ltd) at a frequency of 1 h. A soil moisture tension meter (Institute of Soil Science, Chinese Academy of Sciences, Nanjing, China) was used to measure soil water potential in each  $I_{AWD}$  plot twice each day (8:00 and 14:00). In this experiment, fertilizer management was in line with the fertilization practices of local farmers. Fertilizers were annually used, including 210 kg ha<sup>-1</sup> N of urea, 60 kg ha<sup>-1</sup> P<sub>2</sub>O<sub>5</sub> of superphosphate, and 75 kg ha<sup>-1</sup> K<sub>2</sub>O of potassium sulfate. Urea was divided into three dressings with 40 % as a base fertilizer on May 28 (2020) and May 26 (2021), 40 % as tiller fertilizer on June 9 (2020) and June 8 (2021), and 20 % as panicle fertilizer on July 12 (2020) and July 12 (2021), respectively. Potassium was divided into two dressings, with 50 % as a basal dressing and 50 % as a top dressing at panicle initiation. All phosphate fertilizers were administered as a basal dressing. The experiment was repeated in 2021 without the addition of biochar. Weed, pesticide, and pest management practices followed the local rice farming guidelines.

### 2.3. Sampling and measurement

#### 2.3.1. $NH_3$ volatilization measurement

$NH_3$  volatilization from paddy fields was collected using the continuous airflow enclosure method (Chen et al., 2021). The chamber was made of polyvinyl chloride with a diameter of 20 cm and a height of 30 cm. Samples were collected from 9:00–11:00 a.m. at the frequency of 1–2 d intervals in the first week after each fertilization, followed by 3–5 d interval for the second week, and then a 7–10 day interval until harvest. To absorb the released alkaline  $NH_3$ , the chamber air was pumped at the rate of 12 L min<sup>-1</sup> into a glass container holding 100 mL of 2 %

H<sub>3</sub>BO<sub>3</sub> adsorbent solution for 2 h (Chen et al., 2022). The concentration of NH<sub>3</sub> in the adsorbent solution was measured by an AA3 continuous flow analyzer (Autoanalyzer 3, Seal Analytical, Germany).

### 2.3.2. N<sub>2</sub>O emission measurement

The N<sub>2</sub>O flux was measured using the static chamber method (Wang et al., 2018). The static chamber was made of non-transparent PVC, which was composed of a base, a standard chamber and an extension chamber, as in detail described by Zhao et al. (2023). The gas samples were collected on the 1st, 4th and 7th days after each fertilizer application, and a sample was collected once every 7 days thereafter. The sampling time was between 9:00 am and 11:00 am. A transparent hose was mounted on top of each chamber to connect a three-way valve, which connected the needle and syringe (100 mL). The gas samples were collected using a vacuum aluminum bag with a metal valve for connecting the needle. Samples were collected from the chamber at 0, 15 and 30 min by syringe, and then transferred to a vacuum aluminum bag by needle. The N<sub>2</sub>O concentration was determined by the thermal conductivity and flame ionization detector on an Agilent 7890B gas chromatograph system (Agilent Technologies, Inc., USA).

### 2.3.3. N leaching measurement

Ceramic suction cups were placed at the 40 cm depth from the soil surface in each plot to collect leachate samples (Fig. 1). To enable automatic water collection in the attached bottles, ceramic tubes were induced with vacuum pressure using a pump 12 hours before water collection (Bajracharya and Homagain, 2006). The sampling frequency was consistent with NH<sub>3</sub> volatilization sampling. Leachate samples were determined by an AA3 continuous flow analyzer (Seal Analytical, Autoanalyzer 3, Germany).

### 2.3.4. Soil sampling and analysis

Three parallel surface soil samples (0–20 cm) were collected every week from each plot after transplanting using an auger and thoroughly mixed. Soil NH<sub>4</sub><sup>+</sup>-N and NO<sub>3</sub><sup>-</sup>-N concentrations were determined by an AA3 continuous flow analyzer (Seal Analytical, Germany). Both soil pH and redox potential at the depth of 10 cm were in situ measured using an automatic redox potential tester (CN61M/FJA3, Nanjing, China) with the same sampling frequency N<sub>2</sub>O sampling.

### 2.3.5. Aboveground dry matter, N uptake and reactive N losses

At harvest, rice plants were collected from three randomly selected hills within a 1 m<sup>2</sup> area from the center of each plot. The plant sample was separated into stem, leaf, and panicle, and then rinsed, oven-dried at 70 °C for 48 h, weighed, and powdered through a 0.15 mm sieve. Nitrogen concentration was measured using a full-automatic Kjelflex k-360 analyzer. (Sun et al., 2019). Reactive N losses are the total amount of NH<sub>3</sub> volatilization, N<sub>2</sub>O emissions and inorganic N leaching (Shang et al., 2024).

### 2.3.6. Calculation and statistic

The NH<sub>3</sub> volatilization fluxes were calculated using Eq. (1).

$$f_{\text{NH}_3\text{-N}} = \frac{48C_{\text{NH}_3}V_{\text{NH}_3}}{A} \# \quad (1)$$

where  $f_{\text{NH}_3\text{-N}}$  is the NH<sub>3</sub> volatilization flux (g m<sup>-2</sup> d<sup>-1</sup>),  $C_{\text{NH}_3}$  is the concentration of NH<sub>3</sub> in H<sub>3</sub>BO<sub>3</sub> adsorbent solution (g mL<sup>-1</sup>),  $V_{\text{NH}_3}$  is the H<sub>3</sub>BO<sub>3</sub> volume of adsorbent solution used to absorb the released alkaline NH<sub>3</sub> (mL), 48 is the conversion coefficient, and  $A$  is the cross-sectional area of the PVC collector (cm<sup>2</sup>).

The cumulative NH<sub>3</sub> volatilization was calculated using Eq. (2).

$$T_{\text{NH}_3} = \sum_{i=1}^N \left[ \frac{f_{\text{NH}_3\text{-N},i} + f_{\text{NH}_3\text{-N},i-1}}{2} T_{i-1} \right] \times 10 \# \quad (2)$$

where  $T_{\text{NH}_3\text{-N}}$  is the cumulative NH<sub>3</sub> emissions (kg ha<sup>-1</sup>),  $f_{\text{NH}_3\text{-N},i}$  and  $f_{\text{NH}_3\text{-N},i-1}$  are the NH<sub>3</sub> volatilization flux (g m<sup>-2</sup> d<sup>-1</sup>) of two adjacent sampling dates,  $T_{i-1}$  is the number of days between  $f_{\text{NH}_3\text{-N},i}$  and  $f_{\text{NH}_3\text{-N},i-1}$  (d),  $N$  is the total number of samples.

The N<sub>2</sub>O fluxes were calculated using Eq. (3).

$$F = \rho h \cdot \frac{dC}{dt} \cdot \frac{273}{273 + T} \# \quad (3)$$

where  $F$  is the flux (mg m<sup>-2</sup> h<sup>-1</sup>),  $\rho$  is N<sub>2</sub>O-N density (1.964 kg m<sup>-3</sup>) at the standard state,  $h$  is the height of static chamber (m),  $dC/dt$  is the N<sub>2</sub>O concentration changes in the chamber during the sampling time,  $T$  is the air temperature in the chamber (°C).

The cumulative N<sub>2</sub>O emissions were calculated using Eq. (4).

$$T_{\text{N}_2\text{O-N}} = \sum_{i=1}^{n-1} \left[ \frac{F_i + F_{i+1}}{2} \times d \times 24 \times 10^{-5} \right] \# \quad (4)$$

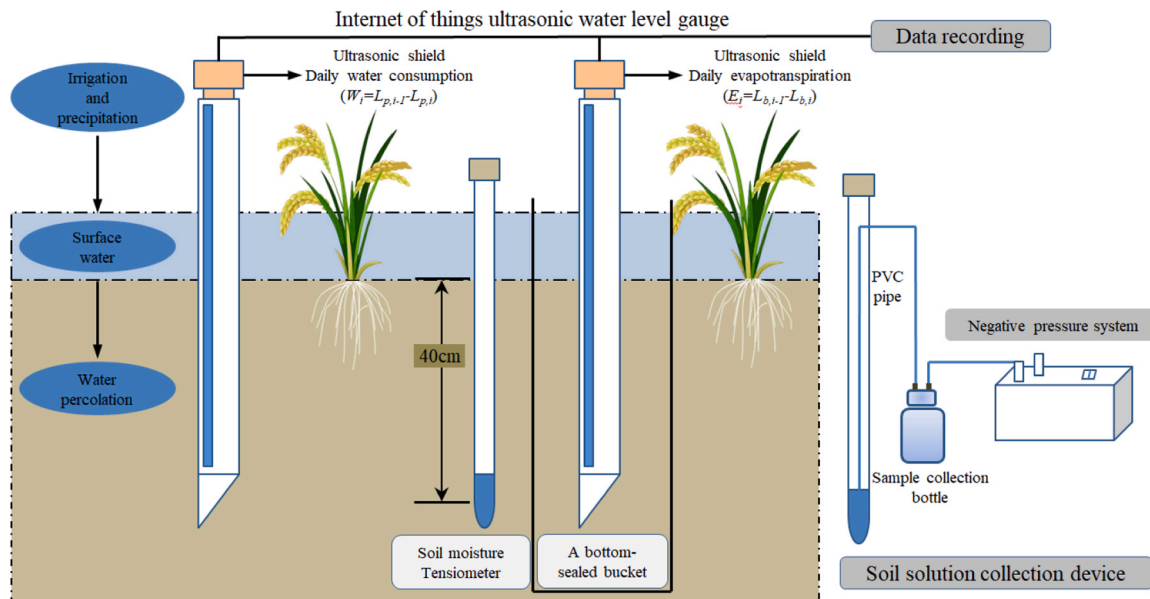


Fig. 1. Water percolation and devices used to determine water percolation, and collect leachate.

where  $T_{N_2O-N}$  is the cumulative  $N_2O$  emissions ( $\text{kg ha}^{-1}$ ),  $d$  is the number of days between two samplings.

The amount of  $\text{NH}_4^+-\text{N}$  or  $\text{NO}_3^--\text{N}$  leaching was calculated using the following Eq. (5).

$$T_{\text{NH}_4^+-\text{N}/\text{NO}_3^--\text{N}} = \sum_{i=2}^N \frac{C_{i-1} + C_i}{2} \times S_{T_i} \times 10^{-3} \# \quad (5)$$

where  $T_{\text{NH}_4^+-\text{N}/\text{NO}_3^--\text{N}}$  is amount of  $\text{NH}_4^+-\text{N}$  or  $\text{NO}_3^--\text{N}$  leaching ( $\text{kg ha}^{-1}$ ),  $C_i$  is the  $\text{NH}_4^+-\text{N}$  or  $\text{NO}_3^--\text{N}$  concentration in leachate at the  $i^{\text{th}}$  sampling ( $\text{mg L}^{-1}$ ), and  $S_{T_i}$  is the amount of water percolation between the  $i-1$ th sampling and the  $i^{\text{th}}$  sampling ( $\text{m}^3 \text{ha}^{-1}$ ) (Han et al., 2023).

The amount of reactive N losses was calculated using the following Eq. (6).

$$T_r = T_{\text{NH}_3} + T_{\text{N}_2\text{O}-\text{N}} + T_{\text{NH}_4^+-\text{N and NO}_3^--\text{N}} \# \quad (6)$$

where  $T_r$  is amount of reactive N loss ( $\text{kg ha}^{-1}$ ),  $T_{\text{NH}_3}$  is the cumulative  $\text{NH}_3$  emissions ( $\text{kg ha}^{-1}$ ),  $T_{\text{N}_2\text{O}-\text{N}}$  is the cumulative  $\text{N}_2\text{O}$  emissions ( $\text{kg ha}^{-1}$ ),  $T_{\text{NH}_4^+-\text{N and NO}_3^--\text{N}}$  is total amount of  $\text{NH}_4^+-\text{N}$  and  $\text{NO}_3^--\text{N}$  leaching ( $\text{kg ha}^{-1}$ ).

#### 2.4. Statistical analysis

Data were separately analyzed as a split-plot design each year using R software (version 4.3.1). Irrigation regime and biochar addition were considered as fixed effects and replications as random effects. Multiple comparisons were performed using Tukey's HSD test at the  $P < 0.05$  level. Principal component analysis was performed on all traits using Origin 2018. Comparisons of means were only conducted when the differences were significant at the  $P < 0.05$  level unless otherwise indicated (Liu et al., 2019).

### 3. Results

#### 3.1. $\text{NH}_3$ volatilization

Daily  $\text{NH}_3$  volatilization fluxes generally peaked 1–3 days after each split N fertilization, followed by a rapid decline throughout all treatments in two years (Fig. 2). During most of the whole rice growth period,  $\text{NH}_3$  volatilization fluxes tended to decrease with biochar addition under both irrigation regimes. On average, the total amount of  $\text{NH}_3$  volatilization within 7 days after three fertilizations accounted for 62.5 % and 59.8 % of seasonal cumulative  $\text{NH}_3$  volatilization in 2020 and 2021, respectively.

Seasonal cumulative  $\text{NH}_3$  volatilization was significantly impacted by the irrigation regime only in 2021 and biochar addition in both years (Table 1).  $I_{\text{AWD}}$  emitted an 8.9 % higher seasonal cumulative  $\text{NH}_3$  volatilization in 2021 over  $I_{\text{CF}}$ , while did not differ in 2020. In contrast, biochar addition decreased seasonal cumulative  $\text{NH}_3$  volatilization in both years. Compared with  $B_0$ ,  $B_{20}$  reduced seasonal cumulative  $\text{NH}_3$  volatilization by 8.8 % and 20.7 % in 2020 and 2021. Among the treatments,  $I_{\text{AWD}}B_{20}$  did not differ from  $I_{\text{CF}}B_{20}$ , while significantly lowered cumulative  $\text{NH}_3$  volatilization by 7.7 % and 13.7 %, respectively, in 2020 and 2021 as compared to  $I_{\text{CF}}B_0$ . The significant decrease in seasonal cumulative  $\text{NH}_3$  volatilization under  $I_{\text{AWD}}B_{20}$  compared with  $I_{\text{CF}}B_0$  mainly resulted from the 17.9 % and 9.2 % significant decrease in cumulative  $\text{NH}_3$  volatilization at the tiller fertilizer stage in 2020 and 2021, and the 17.5 % decrease at the panicle fertilizer stage in 2021.

#### 3.2. $\text{N}_2\text{O}$ emissions

$\text{N}_2\text{O}$  fluxes under  $I_{\text{AWD}}$  tended to peak approximately 7–15 days after tiller fertilizer and 3–5 days after panicle fertilizer, then decreased rapidly and remained at a low rate until harvest (Fig. 3b, d), while remained relatively lower rate during most of the whole rice growth period under  $I_{\text{CF}}$  (Fig. 3a, c).  $\text{N}_2\text{O}$  fluxes under  $I_{\text{AWD}}$  mainly occurred at

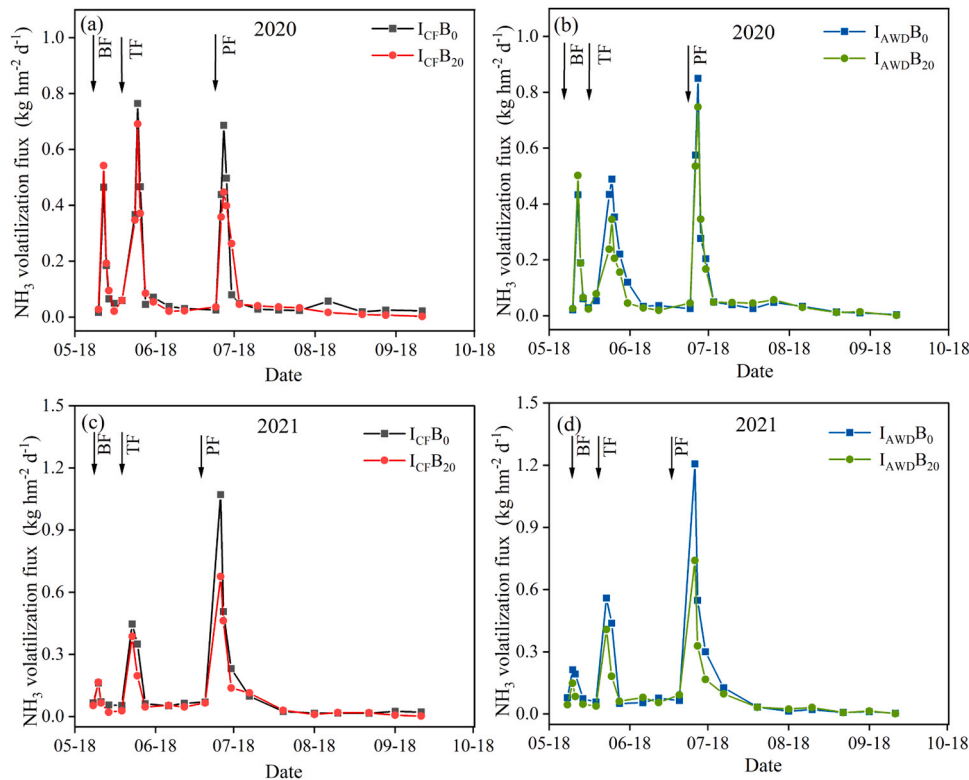
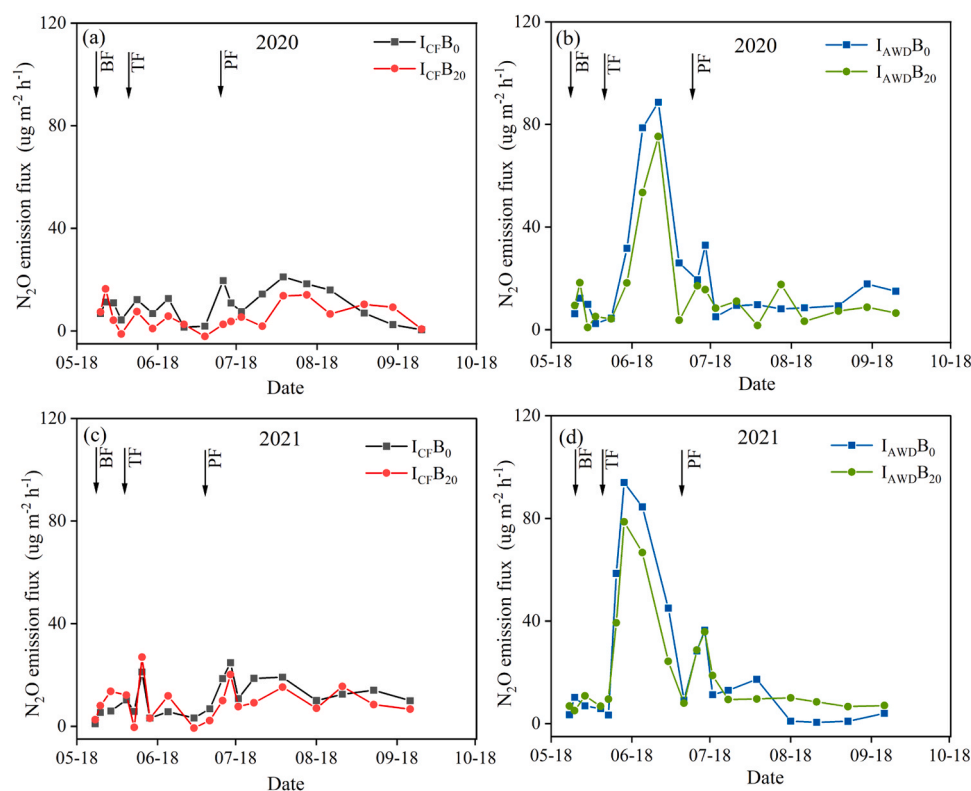


Fig. 2.  $\text{NH}_3$  volatilization fluxes at the base fertilizer stage (BF), tillering fertilizer stage (TF) and panicle fertilizer (PF) stage under different treatments during 2020 and 2021.

**Table 1**Cumulative  $\text{NH}_3$  volatilization at the three fertilization stages and during the whole rice growth period under different treatments in 2020 and 2021.

Treatments	$\text{NH}_3$ volatilization/(kg-N-ha <sup>-1</sup> )							
	Base fertilizer stage		Tiller fertilizer stage		Panicle fertilizer stage		Whole rice growth period	
	2020	2021	2020	2021	2020	2021	2020	2021
I <sub>CF</sub>	1.89 A	1.33 A	2.47 A	2.36B	3.08 A	3.67 A	7.45 A	7.36B
I <sub>AWD</sub>	1.91 A	1.52 A	2.38 A	2.60 A	3.23 A	3.90 A	7.53 A	8.01 A
B <sub>0</sub>	1.89a	1.61a	2.62a	2.72a	3.32a	4.24a	7.83a	8.57a
B <sub>20</sub>	1.92a	1.24b	2.24b	2.23b	2.99b	3.33b	7.14b	6.80b
I <sub>CF</sub> B <sub>0</sub>	1.87a	1.49ab	2.63a	2.60a	3.34a	4.01a	7.84a	8.10b
I <sub>CF</sub> B <sub>20</sub>	1.92a	1.17c	2.32b	2.11ab	2.81b	3.34b	7.05b	6.62c
I <sub>AWD</sub> B <sub>0</sub>	1.91a	1.72a	2.60a	2.83ab	3.30a	4.48a	7.81a	9.04a
I <sub>AWD</sub> B <sub>20</sub>	1.92a	1.32bc	2.16b	2.36b	3.17a	3.31b	7.24b	6.99c
ANOVA								
I	ns	ns	ns	*	ns	ns	ns	*
B	ns	*	**	*	*	**	**	**
I×B	ns	ns	ns	ns	ns	ns	ns	ns

Means followed by different lowercase and uppercase letters within the same column are significantly different at  $P < 0.05$ . I: irrigation regime, B: biochar addition. B<sub>0</sub> and B<sub>20</sub> represent biochar addition rates at 0 and 20 t ha<sup>-1</sup>. \*\*:  $P < 0.01$ ; \*:  $P < 0.05$ ; ns: no significance



**Fig. 3.**  $\text{N}_2\text{O}$  emissions fluxes at the base fertilizer stage (BF), tillering fertilizer stage (TF) and panicle fertilizer (PF) stage under different treatments during 2020 (a) and 2021 (b).

the tiller fertilizer stage, accounting for 59.0 % (2020) and 62.5 % (2021) of the total  $\text{N}_2\text{O}$  emissions, while the  $\text{N}_2\text{O}$  fluxes under I<sub>CF</sub> mainly occurred at the panicle fertilizer stage, accounting for 66.1 % (2020) and 63.7 % (2021) of the total  $\text{N}_2\text{O}$  emissions.

Seasonal cumulative  $\text{N}_2\text{O}$  emission was significantly and consistently affected by irrigation regime, biochar addition and their interaction in both years (Table 2). I<sub>AWD</sub> increased seasonal cumulative  $\text{N}_2\text{O}$  emissions by 115 % and 105 % in 2020 and 2021 as compared to I<sub>CF</sub>, respectively. In contrast, B<sub>20</sub> reduced seasonal cumulative  $\text{N}_2\text{O}$  emissions by 19.7 % in 2020 and 19.2 % in 2021. I<sub>AWD</sub>B<sub>0</sub> increased seasonal cumulative  $\text{N}_2\text{O}$  emissions by 113 % in 2020 and 115 % in 2021 as compared to I<sub>CF</sub>B<sub>0</sub>, while I<sub>AWD</sub>B<sub>20</sub> reduced it by 19.3 % in 2020 and 22.0 % in 2021 as compared to I<sub>AWD</sub>B<sub>0</sub>, respectively. Biochar addition coupled with I<sub>AWD</sub> deceased seasonal cumulative  $\text{N}_2\text{O}$  emissions 19.3 % and 22.0 % in

2020 and 2021 compared with I<sub>AWD</sub> alone, although 72.2 % and 68.0 % increases were observed in 2020 and 2021 over I<sub>CF</sub>B<sub>0</sub>, respectively.

### 3.3. N leaching

$\text{NH}_4^+$ -N leaching mainly occurred at the tiller fertilizer stage, accounting for 44.7–65.5 % of the seasonal cumulative  $\text{NH}_4^+$ -N leaching during the whole rice growth period, while  $\text{NO}_3^-$ -N leaching mainly occurred at the tiller fertilizer stage and panicle fertilizer stage, accounting for 48.4–62.1 % and 27.0–44.8 % of the seasonal cumulative  $\text{NO}_3^-$ -N leaching, respectively (Fig. 4). Seasonal cumulative  $\text{NH}_4^+$ -N leaching was significantly influenced by irrigation regime, but not by biochar addition and their interaction in both years (Table 3 and Table 4). I<sub>AWD</sub> significantly decreased  $\text{NH}_4^+$ -N leaching by 32.1 % in

**Table 2**Cumulative N<sub>2</sub>O emissions at the three fertilization stages and during the whole rice growth period under different treatments in 2020 and 2021.

Treatments	N <sub>2</sub> O emissions /(g·N·ha <sup>-1</sup> )							
	Base fertilizer stage		Tiller fertilizer stage		Panicle fertilizer stage		Whole rice growth period	
	2020	2021	2020	2021	2020	2021	2020	2021
I <sub>CF</sub>	11.4 A	18.7 A	35.7B	49.7B	92.7 A	120 A	139.7B	188.4B
I <sub>AWD</sub>	10.6 A	17.6 A	177.9 A	241.9 A	112 A	126.7 A	300.5 A	386.3 A
B <sub>0</sub>	12a	19.8a	119.9a	166.2a	112.3a	131.8a	244.2a	317.8a
B <sub>20</sub>	9.9b	16.6b	93.7b	125.4b	92.4b	114.9b	196b	256.9b
I <sub>CF</sub> B <sub>0</sub>	12.3a	20.2a	38.0c	53.7c	105.5a	127.8ab	155.8c	201.6c
I <sub>CF</sub> B <sub>20</sub>	10.5b	17.3b	33.3c	45.7c	79.9b	112.3b	123.7d	175.2d
I <sub>AWD</sub> B <sub>0</sub>	11.8a	19.4a	201.8a	278.6a	119a	135.9a	332.6a	433.9a
I <sub>AWD</sub> B <sub>20</sub>	9.4c	15.9c	154b	205.2b	105a	117.5b	268.3b	338.6b
ANOVA								
I	ns	ns	**	**	ns	ns	**	**
B	**	**	**	**	*	*	**	**
I×B	ns	ns	**	**	ns	ns	*	**

Means followed by different lowercase and uppercase letters within the same column are significantly different at  $P < 0.05$ . I: irrigation regime, B: biochar addition. B<sub>0</sub> and B<sub>20</sub> represent biochar addition rates at 0 and 20 t ha<sup>-1</sup>. \*\*:  $P < 0.01$ ; \*:  $P < 0.05$ ; ns: no significance

2020 and 45.3 % in 2021, compared to I<sub>CF</sub>. It is worth noting that B<sub>20</sub> did not significantly differ from B<sub>0</sub>, although a 13.8 % and 9.9 % lower NH<sub>4</sub><sup>+</sup>-N leaching was numerically observed in 2020 and 2021, respectively. Seasonal cumulative NO<sub>3</sub>-N leaching was significantly influenced by the irrigation regime in both years and by biochar in 2021, but not by their interaction at the tiller fertilizer stage in both years. I<sub>AWD</sub> significantly increased NO<sub>3</sub>-N leaching by 33.6 % in 2020 and 34.7 % in 2021, compared to I<sub>CF</sub>, while it decreased by 10.7 % under B<sub>20</sub> over B<sub>0</sub> in 2021. Compared to I<sub>AWD</sub>B<sub>0</sub>, I<sub>AWD</sub>B<sub>20</sub> tended to decrease NH<sub>4</sub><sup>+</sup>-N leaching and NO<sub>3</sub>-N leaching by 12.5–17.2 % and 4.2–19.1 % in two years, respectively.

Seasonal cumulative inorganic N (NH<sub>4</sub><sup>+</sup>-N and NO<sub>3</sub>-N) leaching ranged from 37.92 to 34.55 kg ha<sup>-1</sup> in 2020 and from 21.65 to 31.96 kg ha<sup>-1</sup> in 2021, which accounted for 13.3–16.5 % of the total N fertilizer used in 2020 and 10.3–15.2 % in 2021 (Table 5). Inorganic N leaching was only significantly influenced by the irrigation regime and biochar addition in 2021. I<sub>AWD</sub> significantly reduced inorganic N leaching by 38.9 % at the tiller fertilizer stage, and thereby decreased by 22.4 % over the whole rice growth period, compared to I<sub>CF</sub>. B<sub>20</sub> significantly decreased inorganic N leaching by 8.3 % at the panicle fertilizer stage, and 12.1 % over the whole rice growth period. Among the four treatments, I<sub>AWD</sub>B<sub>20</sub> did not significantly alter inorganic N leaching in 2020, but significantly decreased it by 32.3 % in 2021, as compared to I<sub>CF</sub>B<sub>0</sub>.

### 3.4. Soil environment variables

Soil redox potential (ORP) had no significant difference among the treatments before the late tillering stage, thereafter, ORP was considerably higher under I<sub>AWD</sub> than I<sub>CF</sub> (Fig. 5a, b). ORP was significantly affected by the irrigation regime in both years, I<sub>AWD</sub> increased ORP by 54.1 % in 2020 and 47.6 % in 2021, compared to I<sub>CF</sub> (Fig. 6a, b and Table S3). Soil pH showed a generally consistent trend among the treatments (Fig. 5c, d). Biochar addition significantly affected soil pH in both years (Fig. 6c, d and Table S3), which increased soil pH by 1.0 % in 2020 and 0.6 % in 2021, respectively. In addition, I<sub>AWD</sub> significantly increased soil pH by 1.1 % in 2021.

Soil NH<sub>4</sub><sup>+</sup>-N tended to peak approximately 1–3 days after basal fertilizer and tiller fertilizer application, and then decreased rapidly in both years (Fig. 5e, f). Relative to the soil NH<sub>4</sub><sup>+</sup>-N, soil NO<sub>3</sub>-N was less affected by N fertilizer application (Fig. 5g, h). Soil NH<sub>4</sub><sup>+</sup>-N was significantly affected by biochar addition in both years, while soil NO<sub>3</sub>-N was significantly affected by the irrigation regime in both years (Fig. 6e-h and Table S3). Compared with B<sub>0</sub>, B<sub>20</sub> increased soil NH<sub>4</sub><sup>+</sup>-N by 25.9 % in 2020 and 52.4 % in 2021. I<sub>AWD</sub> increased soil NO<sub>3</sub>-N by 74.0 % in 2020 and 50.3 % in 2021.

### 3.5. Grain yield, aboveground N uptake and reactive N losses

Grain yield and aboveground N uptake were not significantly affected by irrigation regime, biochar addition and their interaction in both years (Table 6), although I<sub>AWD</sub>B<sub>20</sub> numerically had the highest grain yield and aboveground N uptake in both years. Reactive N losses were significantly influenced by irrigation regime and biochar addition in 2021, but not in 2020 (Table 6). Reactive N losses were significantly decreased by 15.7 % under I<sub>AWD</sub> over I<sub>CF</sub> in 2021, and it was significantly decreased by 14.1 % under B<sub>20</sub> in 2021. Among the four treatments, I<sub>AWD</sub>B<sub>20</sub> yielded the lowest reactive N losses (28.98 kg N ha<sup>-1</sup>) in 2021, which was 28.0 %, 17.6 % lower than I<sub>CF</sub>B<sub>0</sub> and I<sub>AWD</sub>B<sub>0</sub>, respectively. In 2020, I<sub>AWD</sub>B<sub>20</sub> numerically had the lowest reactive N losses (35.43 kg N ha<sup>-1</sup>), but there was not a significant difference when compared with the other treatments.

### 3.6. Principal component analysis (PCA) for all traits

The PCA results identified two principal components with their eigenvalues > 1 and contributed to 77.88 % of the total variation of 11 traits (Table 7). The absolute value of all N losses trait coefficients and major of the soil environment traits coefficients were > 0.6. PC1 explained 49.76 % of the total variation, with a coefficient of -0.94–0.97, mainly reflecting N<sub>2</sub>O emissions (0.87), NH<sub>4</sub><sup>+</sup>-N leaching (-0.94), NO<sub>3</sub>-N leaching (0.82), soil ORP (0.95), soil NO<sub>3</sub>-N (0.97) and reactive N losses (-0.72). Accordingly, PC2 reflects a 28.12 % variation in NH<sub>3</sub> volatilization (0.88) and soil NH<sub>4</sub><sup>+</sup>-N (-0.94).

The PCA scatter plot and biplot are shown in Fig. 7. The length of each load arrow represents the correlation coefficient between each trait and PCs. For instance, NH<sub>3</sub> volatilization was positively loaded on PC1 and PC2, and the correlation coefficients were 0.09 and 0.88, respectively, while reactive N losses were negatively loaded on PC1 (-0.72) and positively loaded on PC2 (0.51). The strong correlation between irrigation regimes and PC1 suggests that the N<sub>2</sub>O emissions and NO<sub>3</sub>-N leaching were negatively affected and NH<sub>4</sub><sup>+</sup>-N leaching, reactive N losses, soil ORP and soil NO<sub>3</sub>-N were positively affected by changing from CF to AWD (Fig. 7). Accordingly, the strong negative correlation between biochar addition and PC2 suggests that the NH<sub>3</sub> volatilization and soil NH<sub>4</sub><sup>+</sup>-N were positively affected by biochar. In addition, our result further suggested that reactive N losses were positively correlated with NH<sub>4</sub><sup>+</sup>-N leaching and NH<sub>3</sub> volatilization, while negatively associated with soil NH<sub>4</sub><sup>+</sup>-N and soil NO<sub>3</sub>-N (Fig. 7). This indicates that biochar addition coupling with alternate wetting and drying had a positive influence on reducing reactive N losses and enhancing soil N fixation.

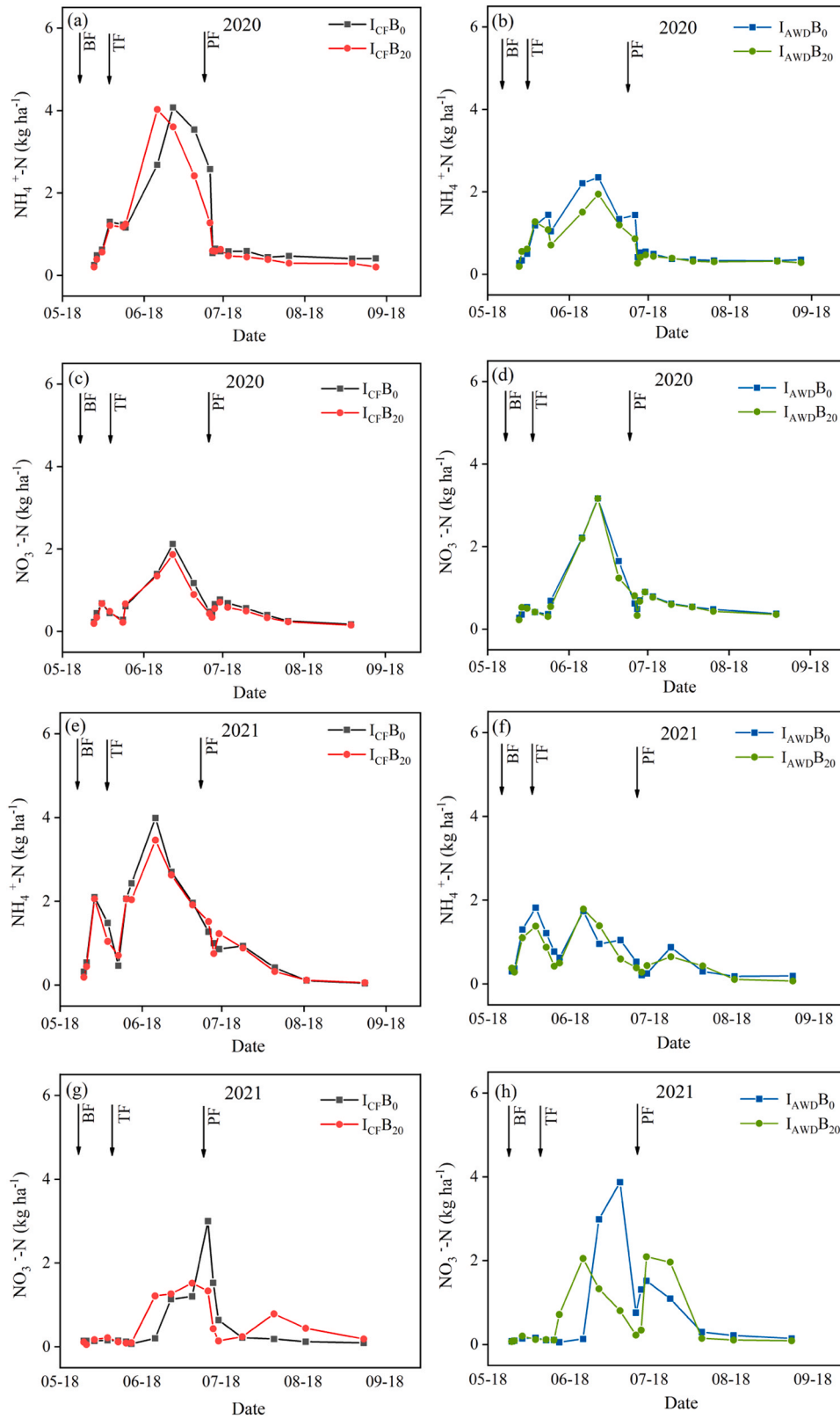


Fig. 4.  $\text{NH}_4^+-\text{N}$  and  $\text{NO}_3^--\text{N}$  leaching at the base fertilizer stage (BF), tillering fertilizer stage (TF) and panicle fertilizer (PF) stage under different treatments during 2020 and 2021.

**Table 3**NH<sub>4</sub><sup>+</sup>-N leaching at the three fertilization stages and during the whole rice growth period under different treatments in 2020 and 2021.

Treatments	NH <sub>4</sub> <sup>+</sup> -N leaching /(kg·ha <sup>-1</sup> )							
	Base fertilizer stage		Tiller fertilizer stage		Panicle fertilizer stage		Whole rice growth period	
	2020	2021	2020	2021	2020	2021	2020	2021
I <sub>CF</sub>	3.73 A	4.68 A	13.30 A	14.02 A	4.31 A	3.05 A	21.34 A	21.75 A
I <sub>AWD</sub>	3.72 A	4.51 A	7.31B	5.38B	3.48B	2.00 A	14.50B	11.89B
B <sub>0</sub>	3.82a	4.96a	11.21a	10.05a	4.22a	2.69a	19.25a	17.70a
B <sub>20</sub>	3.63a	4.24a	9.40a	9.35a	3.57a	2.36a	16.60a	15.95a
I <sub>CF</sub> B <sub>0</sub>	3.90a	4.91a	14.03a	14.43a	4.70a	3.37a	22.63a	22.71a
I <sub>CF</sub> B <sub>20</sub>	3.56a	4.45a	12.57a	13.61a	3.93a	2.73ab	20.06a	20.79a
I <sub>AWD</sub> B <sub>0</sub>	3.73a	5.00a	8.39a	5.67b	3.75a	2.01b	15.87a	12.68b
I <sub>AWD</sub> B <sub>20</sub>	3.71a	4.02a	6.22a	5.09b	3.21a	1.99b	13.14a	11.10b
ANOVA								
I	ns	ns	*	**	*	ns	*	**
B	ns	ns	ns	ns	ns	ns	ns	ns
I×B	ns	ns	ns	ns	ns	ns	ns	ns

Means followed by different lowercase and uppercase letters within the same column are significantly different at  $P < 0.05$ . I: irrigation regime, B: biochar addition. B<sub>0</sub> and B<sub>20</sub> represent biochar addition rates at 0 and 20 t ha<sup>-1</sup>. \*\*:  $P < 0.01$ ; \*:  $P < 0.05$ ; ns: no significance

**Table 4**NO<sub>3</sub>-N leaching at the three fertilization stages and during the whole rice growth period under different treatments in 2020 and 2021.

Treatments	NO <sub>3</sub> -N leaching /(kg·ha <sup>-1</sup> )							
	Base fertilizer stage		Tiller fertilizer stage		Panicle fertilizer stage		Whole rice growth period	
	2020	2021	2020	2021	2020	2021	2020	2021
I <sub>CF</sub>	2.00 A	0.70 A	5.49B	5.55 A	3.80B	2.51B	11.30B	8.76B
I <sub>AWD</sub>	1.95 A	0.58 A	8.14 A	6.57 A	5.01 A	4.65 A	15.10 A	11.79 A
B <sub>0</sub>	2.01a	0.65a	7.06a	6.82a	4.61a	3.68a	13.67a	11.15a
B <sub>20</sub>	1.94a	0.63a	6.58a	5.29b	4.20a	3.48a	12.72a	9.41b
I <sub>CF</sub> B <sub>0</sub>	2.09a	0.73a	5.78a	5.74b	4.06ab	2.79b	11.93ab	9.25b
I <sub>CF</sub> B <sub>20</sub>	1.91a	0.68a	5.21a	5.35b	3.55b	2.23b	10.67b	8.26b
I <sub>AWD</sub> B <sub>0</sub>	1.93a	0.57a	8.34a	7.91a	5.15a	4.57a	15.42a	13.04a
I <sub>AWD</sub> B <sub>20</sub>	1.96a	0.59a	7.95a	5.22b	4.86a	4.73a	14.78ab	10.55ab
ANOVA								
I	ns	ns	*	ns	**	**	**	*
B	ns	ns	ns	**	ns	ns	ns	*
I×B	ns	ns	ns	*	ns	ns	ns	ns

Means followed by different lowercase and uppercase letters within the same column are significantly different at  $P < 0.05$ . I: irrigation regime, B: biochar addition. B<sub>0</sub> and B<sub>20</sub> represent biochar addition rates at 0 and 20 t ha<sup>-1</sup>. \*\*:  $P < 0.01$ ; \*:  $P < 0.05$ ; ns: no significance

**Table 5**

Inorganic N leaching at the three fertilization stages and during the whole rice growth period under different treatments in 2020 and 2021.

Treatments	Inorganic N leaching /(kg·ha <sup>-1</sup> )							
	Base fertilizer stage		Tiller fertilizer stage		Panicle fertilizer stage		Whole rice growth period	
	2020	2021	2020	2021	2020	2021	2020	2021
I <sub>CF</sub>	5.73 A	5.38 A	18.80 A	19.57 A	8.11 A	5.56 A	32.64 A	30.51 A
I <sub>AWD</sub>	5.67 A	5.09 A	15.45 A	11.95B	8.48 A	6.65 A	29.60 A	23.68B
B <sub>0</sub>	5.82a	5.60a	18.27a	16.87a	8.83a	6.37a	32.92a	28.84a
B <sub>20</sub>	5.57a	4.87a	15.98a	14.64a	7.77a	5.84b	29.32a	25.35b
I <sub>CF</sub> B <sub>0</sub>	5.99a	5.64a	19.81a	20.16a	8.75a	6.16a	34.55a	31.96a
I <sub>CF</sub> B <sub>20</sub>	5.47a	5.13a	17.78a	18.97a	7.47a	4.96b	30.72a	29.06a
I <sub>AWD</sub> B <sub>0</sub>	5.66a	5.56a	16.73a	13.58b	8.90a	6.58a	31.28a	25.72ab
I <sub>AWD</sub> B <sub>20</sub>	5.68a	4.61a	14.17a	10.32b	8.07a	6.72a	27.92a	21.65b
ANOVA								
I	ns	ns	ns	*	ns	ns	ns	*
B	ns	ns	ns	ns	ns	*	ns	*
I×B	ns	ns	ns	ns	ns	*	ns	ns

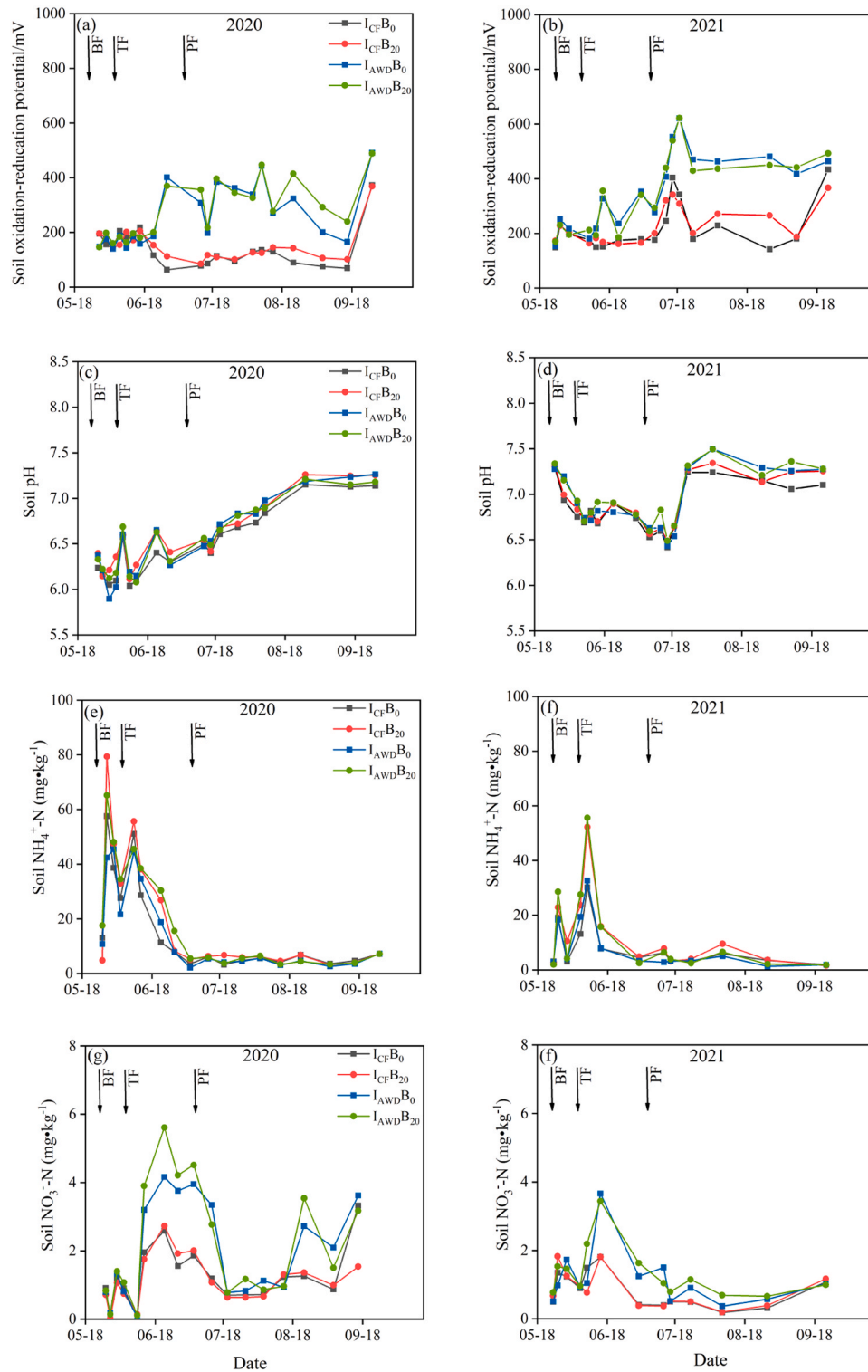
Means followed by different lowercase and uppercase letters within the same column are significantly different at  $P < 0.05$ . I: irrigation regime, B: biochar addition. B<sub>0</sub> and B<sub>20</sub> represent biochar addition rates at 0 and 20 t ha<sup>-1</sup>. \*\*:  $P < 0.01$ ; \*:  $P < 0.05$ ; ns: no significance

## 4. Discussion

### 4.1. NH<sub>3</sub> volatilization

NH<sub>3</sub> volatilization is one of the primary ways of N losses in paddy fields among many N loss ways in paddy fields. In our results, NH<sub>3</sub>

volatilization loss accounted for 3.98–7.32 % of the total N fertilizer application (Fig. 2), which was lower than the 9–39 % of the total N application reported by Chen et al. (2015b). This may be due to higher N leaching caused by the large amount of water percolation during the whole rice growth period in both years (Table S2). The total amount of NH<sub>3</sub> volatilization within 7 days after three fertilizations accounted for

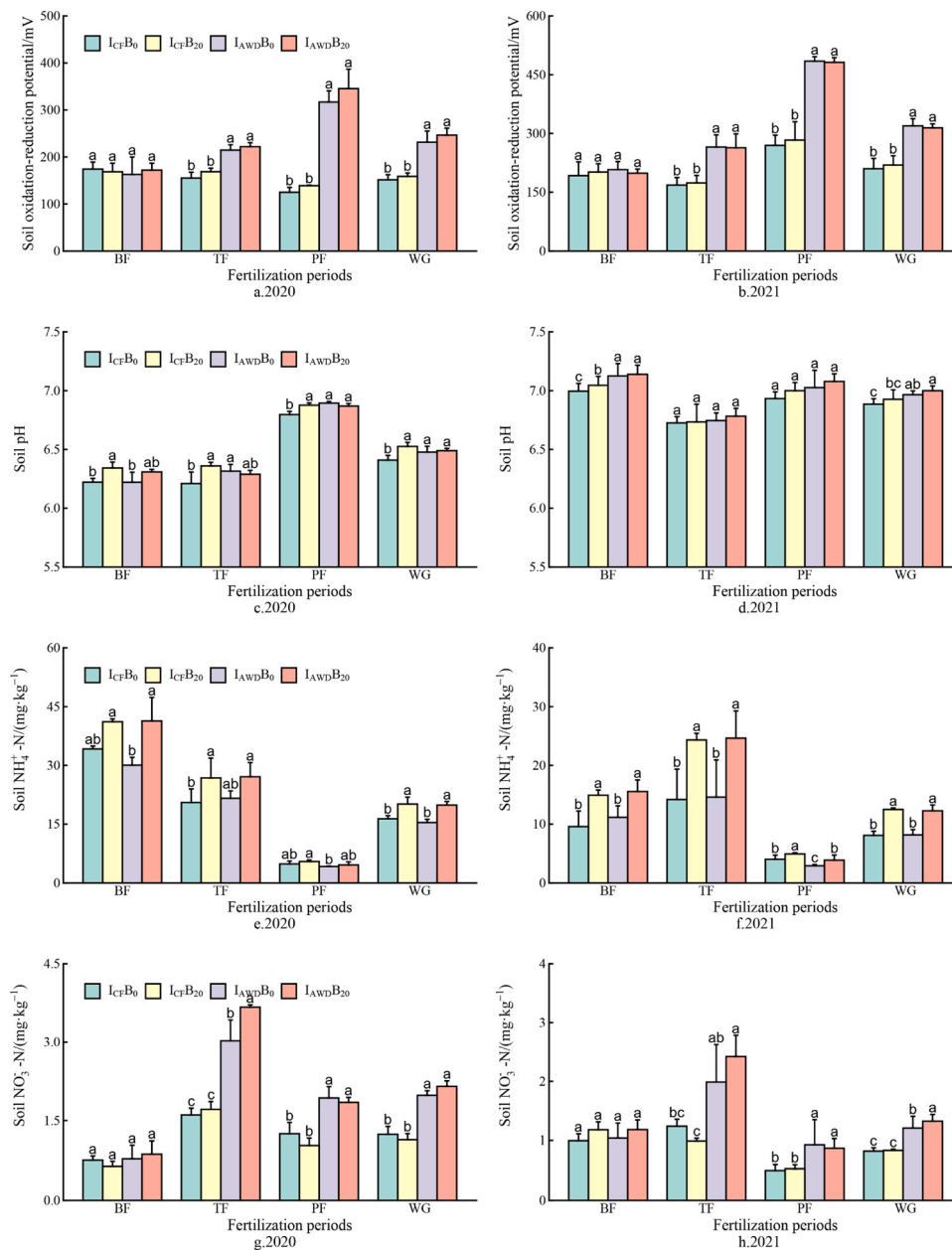


**Fig. 5.** Soil oxidation-reduction potential (a and b), soil pH (c and d), soil  $\text{NH}_4^+\text{-N}$  (e and f), and soil  $\text{NO}_3^-\text{-N}$  (g and h) at the base fertilizer stage (BF), tillering fertilizer stage (TF) and panicle fertilizer (PF) stage under different treatments during 2020 and 2021.

59.8–62.5 % of the seasonal cumulative  $\text{NH}_3$  volatilization (Fig. 2). This is similar to the finding reported by Sun et al. (2019), who estimated that total  $\text{NH}_3$  volatilization within 7–10 days after three fertilizations accounted for over 80 % of the cumulative  $\text{NH}_3$  volatilization.

Conclusions are so far not consistent on  $\text{NH}_3$  volatilization under  $I_{AWD}$ . Some studies reported that  $\text{NH}_3$  volatilization was higher under  $I_{AWD}$  than under  $I_{CF}$  because the higher pH in surface water caused by  $I_{AWD}$  stimulated the transformation of  $\text{NH}_4^+\text{-N}$  to  $\text{NH}_3$  (Dong et al., 2012).

However, Xu et al. (2012) argued that  $I_{AWD}$  decreased  $\text{NH}_3$  volatilization caused by the frequent wetting-drying cycles increased soil adsorption of  $\text{NH}_4^+\text{-N}$ . A recent study reported by Zhao et al. (2023) demonstrated that  $I_{AWD}$  did not significantly affect  $\text{NH}_3$  volatilization due to no significant difference in water conditions after each fertilization. Our present experiment herein indicated that  $I_{AWD}$  did not differ from  $I_{CF}$  in cumulative  $\text{NH}_3$  volatilization in 2020, but significantly increased it in 2021 (Table 1). The unchanged  $\text{NH}_3$  volatilization in 2020 may be due to



**Fig. 6.** Average soil oxidation-reduction potential (a and b), average soil pH (c and d), average soil  $\text{NH}_4^+\text{-N}$  (e and f), and average soil  $\text{NO}_3^-\text{-N}$  (g and h) at the base fertilizer stage (BF), tillering fertilizer stage (TF), panicle fertilizer (PF) stage and during the whole rice growth period (WG) under different treatments during 2020 and 2021.

that frequent and higher precipitation in 2020 overshadowed the differences in water conditions between the two irrigation regimes (Fig. S1). The following increase could be attributed to the increased soil pH caused by the frequent drying-wetting alternations under  $I_{\text{AWDB}}$  increased  $\text{NH}_3$  volatilization (Fig. 6c, d).

The impact of biochar addition on  $\text{NH}_3$  volatilization has been being debated, mainly including the following three views: 1) biochar, especially straw biochar, was rich in salt-based ions such as  $\text{Ca}^{2+}$ ,  $\text{K}^+$ , and  $\text{Mg}^{2+}$ , and these ions from biochar are partially exchanged with  $\text{H}^+$  and  $\text{Al}^{3+}$  in the soil, which in turn increased soil pH and thus increased  $\text{NH}_3$  volatilization (Pan et al., 2021); 2) biochar addition to soils increased the absorptive surface area, microcellular structure, cation exchange capacity (CEC), and functional groups, which enhanced adsorption of  $\text{NH}_4^+$  from the water and storage of  $\text{NH}_4^+$  in the soil, thereby reducing  $\text{NH}_3$  loss (Dawar et al., 2021; Hale et al., 2013); and 3)  $\text{NH}_3$  volatilization was not significantly affected by biochar addition (Feng et al.,

2017). The results from our study showed that biochar addition increased soil pH in both years, but significantly reduced  $\text{NH}_3$  volatilization by 8.8–20.7 % in both years due to the enhanced adsorption capacity of biochar offset the negative effect of pH increase on  $\text{NH}_3$  volatilization (Table S3 and Table 1). This result is similar to those found by Fungo et al. (2019). PCA also showed that soil  $\text{NH}_4^+\text{-N}$  was negatively correlated with the  $\text{NH}_3$  volatilization (Fig. 7). This demonstrated that improving the adsorption capacity of soil to  $\text{NH}_4^+\text{-N}$  is critical to mitigating  $\text{NH}_3$  volatilization. In our results, the reduction in  $\text{NH}_3$  volatilization could be primarily attribute to that the biochar having greater CEC than the soil, and providing greater adsorption sites (Adhikari et al., 2024), which in turn increased the  $\text{NH}_4^+\text{-N}$  level in the surface soil (Fig. 6e, f) and thereby decreased the concentration of  $\text{NH}_4^+\text{-N}$  in surface water (Sun et al., 2019b). As reported, biochar had a higher pH in our result, but the leaching that occurred during the whole rice growth period washed the cations from the biochar to the deeper soil, thereby

**Table 6**

Grain yield, aboveground N uptake, and reactive N losses under different treatments in 2020 and 2021.

Treatments	Grain yield/ (t·ha <sup>-1</sup> )		N uptake/(kg·N ha <sup>-1</sup> )		Reactive N losses (kg·N ha <sup>-1</sup> )	
	2020	2021	2020	2021	2020	2021
I <sub>CF</sub>	9.54 A	8.31 A	106.59 A	105.66 A	40.23 A	38.06 A
I <sub>AWD</sub>	9.7 A	8.73 A	107.21 A	108.35 A	37.43 A	32.08B
B <sub>0</sub>	9.69a	8.22a	106.54a	105.03a	40.99a	37.73a
B <sub>20</sub>	9.56a	8.81a	107.26a	108.98a	36.66a	32.41b
I <sub>CF</sub> B <sub>0</sub>	9.76a	8.08b	106.32a	104.2b	42.55a	40.27a
I <sub>CF</sub> B <sub>20</sub>	9.32a	8.53ab	106.86a	107.12ab	37.90a	35.85a
I <sub>AWD</sub> B <sub>0</sub>	9.61a	8.37ab	106.76a	105.86ab	39.43a	35.19a
I <sub>AWD</sub> B <sub>20</sub>	9.79a	9.10a	107.66a	110.84a	35.43a	28.98b
ANOVA						
I	ns	ns	ns	ns	ns	*
B	ns	ns	ns	ns	ns	**
I×B	ns	ns	ns	ns	ns	ns

Means followed by different lowercase and uppercase letters within the same column are significantly different at  $P < 0.05$ . I: irrigation regime, B: biochar addition. B<sub>0</sub> and B<sub>20</sub> represent biochar addition rates at 0 and 20 t ha<sup>-1</sup>. \*\*:  $P < 0.01$ ; \*:  $P < 0.05$ ; ns: no significance

**Table 7**

Variable loading scores for rice quality and percentage of the variance of each principal component.

Trails	PC1	PC2
N losses traits		
NH <sub>3</sub> volatilization	0.09	0.88
N <sub>2</sub> O emissions	0.87	0.48
NH <sub>4</sub> <sup>+</sup> -N leaching	-0.94	0.04
NO <sub>3</sub> <sup>-</sup> -N leaching	0.82	0.54
Reactive N losses	-0.72	0.51
Soil environment traits		
Soil ORP	0.95	0.17
Soil pH	0.57	-0.34
Soil NH <sub>4</sub> <sup>+</sup> -N	0.10	-0.94
Soil NO <sub>3</sub> <sup>-</sup> -N	0.97	0.08
Yield traits		
Aboveground N uptake	0.48	-0.59
Grain yield	0.51	-0.38
Eigenvalues	5.47	3.09
Percentage of variance (%)	49.76	28.12
Cumulative Percentage of variance (%)	49.76	77.88

Values shaded are the loadings whose absolute value is higher than 0.60.

not significantly affecting soil pH during most of the whole rice growth period (Fig. 5c, d and Table S3) and adsorption capacity of biochar greater influence on NH<sub>3</sub> volatilization (Dey et al., 2023). Therefore, it is critical to balance the pH and adsorption characteristics of biochar to stimulate its potential to reduce NH<sub>3</sub> volatilization in the rice field ecosystem.

#### 4.2. N<sub>2</sub>O emission

I<sub>AWD</sub> technology has been advocated to reduce water use and methane emissions, but a considerable increase in N<sub>2</sub>O emissions caused by its alternating aerobic and anaerobic environment of topsoil after fertilization cannot be ignored (Gao et al., 2019). The increase in N<sub>2</sub>O emission under I<sub>AWD</sub> may be due to the dry-wet cycle promoting nitrogen mineralization in the soil, anaerobic promoting denitrification, and the aeration process promoting nitrification (Li et al., 2020; Seredych et al., 2011). Liao et al. (2020) reported that the N<sub>2</sub>O emissions under mild I<sub>AWD</sub> were 3 times higher than those under I<sub>CF</sub>. In our results, the N<sub>2</sub>O emissions under I<sub>AWD</sub> were 2.15 and 2.05 times higher than those under I<sub>CF</sub> in 2020 and 2021 (Table 2). The higher N<sub>2</sub>O emission of I<sub>AWD</sub> over I<sub>CF</sub> was mainly attributed to the water regulation from the late tillering stage (Sha et al., 2022). In addition, our PCA showed that the

significant increase in N<sub>2</sub>O is closely correlated with the soil NO<sub>3</sub>-N (Fig. 7). This indicated that the dry-wet cycle increased the soil oxygen level and thus promoted the nitrification rate, stimulated the process that produced N<sub>2</sub>O, which in turn increased the soil NO<sub>3</sub>-N levels (Fig. 6a, b, g, h), thereby promoting N<sub>2</sub>O emissions (Liu et al., 2022; Zhou et al., 2020).

Consistent with previous studies (Chen et al., 2022), the result indicated that biochar addition significantly reduced the cumulative N<sub>2</sub>O emissions, especially under I<sub>AWD</sub> (Table 2). The impact of biochar addition on N<sub>2</sub>O emissions can be explained by the following reasons: 1) the pH change caused by biochar addition promoted N<sub>2</sub>O reeducates activity and further converted N<sub>2</sub>O to N<sub>2</sub>, thereby reducing N<sub>2</sub>O emissions (Liu et al., 2022); 2) biochar addition improved the soil water-holding capacity, enhanced the immobilization of NO<sub>3</sub>-N, thereby reducing the substrate availability of denitrification, and thus reducing N<sub>2</sub>O emissions (Karhu et al., 2011) and 3) some harmful substances (ethylene, tar, etc.) in biochar may reduce the activity of nitrification and denitrification colonies in soil in a short time and thus reduced N<sub>2</sub>O emissions (Luz Cayuela et al., 2013). Our study showed that biochar addition increased the concentration of nitrification and denitrification substrates in the topsoil and still mitigated N<sub>2</sub>O emissions because the adsorption of NH<sub>4</sub><sup>+</sup>-N and NO<sub>3</sub>-N by biochar reduced reaction substrates effectiveness (Dey et al., 2023; Lawrinenko et al., 2016). Biochar can be considered as a reducing agent and an electron acceptor, competing with NO<sub>3</sub>-N as an electron sink, reducing the effectiveness of nitrification and denitrification substrates and thereby reducing N<sub>2</sub>O emissions (Nguyen et al., 2016). Therefore, biochar mitigates N<sub>2</sub>O emissions mainly by promoting the reduction of N<sub>2</sub>O in paddy soil and reducing the effectiveness of nitrification and denitrification substrates.

#### 4.3. N leaching

There are many factors affecting N leaching from paddy fields such as soil texture, sampling depth of leaching and fertilization standard (Tan et al., 2013). Bernard-Jannin et al. (2017) indicated that NO<sub>3</sub>-N was the main source of N leaching throughout the whole rice growth period. However, our results indicated that NH<sub>4</sub><sup>+</sup>-N was the main source of N leaching (Table 3). This may be due to the shallow sampling depth (40 cm) and the higher water percolation in our experiment (Yang et al., 2015). The NH<sub>4</sub><sup>+</sup>-N leaching mainly occurred at the base fertilizer and tiller fertilizer stage, possibly due to the 80 % of N fertilizer being applied at these stages. In addition, the leaching in NO<sub>3</sub>-N mainly occurred at the tiller fertilizer and panicle fertilizer stages due to the I<sub>AWD</sub> process during these periods promoting the nitrification rate in soil, which in turn increased NO<sub>3</sub>-N concentration in leachate (Table 4). Relatively higher inorganic N leaching (10.3–16.5 % of N input) over the normal level (0.1–15.0 % of N input) could be explained by the higher water percolation caused by the soil structure in the Lower Liaohe Plain (Wu et al., 2024; Zhu et al., 2000).

Jiao et al. (2018) reported that I<sub>AWD</sub> significantly decreased inorganic N leaching by 10.5–21.1 % because I<sub>AWD</sub> had lower water percolation. In contrast, previous studies found that although I<sub>AWD</sub> had less water percolation, it raised the levels of NH<sub>4</sub><sup>+</sup>-N and NO<sub>3</sub>-N in leachate, thus increasing inorganic N leaching (Tan et al., 2013). Our results support the first view, and PCA further suggested that I<sub>AWD</sub> did alter soil NH<sub>4</sub><sup>+</sup>-N and soil NO<sub>3</sub>-N, which in turn significantly decreased NH<sub>4</sub><sup>+</sup>-N leaching by 32.1–45.3 % but increased NO<sub>3</sub>-N leaching by 33.6–34.7 % in two years (Fig. 7 and Tables 3, 4). In addition, NH<sub>4</sub><sup>+</sup>-N was the main source of inorganic N leaching in our experiment, thereby inorganic N leaching was significantly reduced by I<sub>AWD</sub>. B<sub>20</sub> significantly decreased the inorganic N leaching by 12.1 % in 2021 (Table 5). He et al. (2020) similarly reported that biochar addition decreased N leaching by 26.9–34.8 %. In contrast, some research indicated that biochar increased N leaching by 57 % (Bruun et al., 2012). The influence of biochar addition on inorganic N leaching can be explained by the following reasons: 1) the negatively charged carboxyl and amino groups

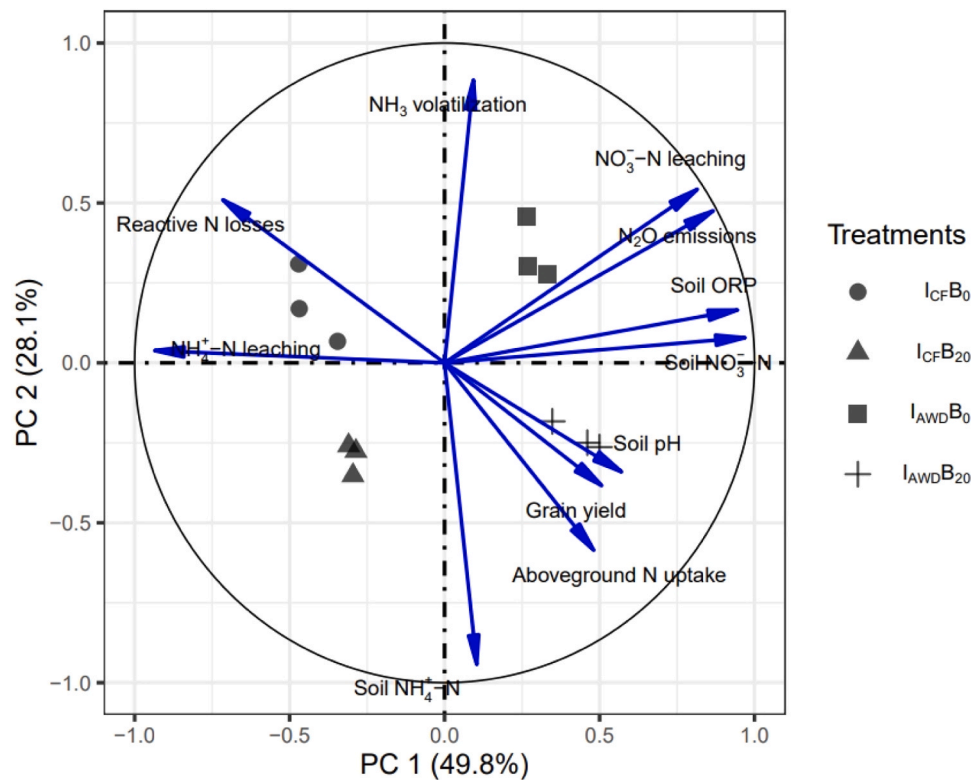


Fig. 7. Principal component analysis (PCA) of variables response to biochar addition (0 and 20 t ha<sup>-1</sup>) and irrigation regimes ( $I_{CF}$  and  $I_{AWD}$ ) throughout the rice growth period.

on biochar surface effectively adsorbed  $NH_4^+$ -N in the topsoil and thereby decreased  $NH_4^+$ -N leaching (Laird et al., 2010); 2) the adsorption capacity of biochar to  $NO_3^-$ -N was weaker than that of  $NH_4^+$ -N (Dong et al., 2020); 3) the biochar addition increased  $NH_3$  volatilization loss, which may decrease inorganic N leaching (Sun et al., 2017); 4) the biochar addition promoted nitrification and increased soil  $NO_3^-$ -N level (Fig. 6g, h), which may increase  $NO_3^-$ -N leaching (Eykelbosh et al., 2015). Our results showed that  $NH_4^+$ -N leaching was not significantly influenced by biochar in both years, which may be due to the higher water percolation in the Lower Liaohe River Plain, which overshadowed the differences in biochar addition. In addition, the results further indicated that biochar did not significantly reduce inorganic N leaching in the first year, which might be related to that biochar contains the amount of 8.61 % g kg<sup>-1</sup> total N inside (Table S1). Biochar additions at the rate of 20 t ha<sup>-1</sup> to rice fields supplied an additional 172.2 kg total N ha<sup>-1</sup> inputs to soil N pool. This caused rises of N leaching for biochar addition, which in turn offsets its effect of reducing N leaching. The reduction of N leaching by biochar addition in the second year which mainly attributed to increased soil  $NO_3^-$ -N adsorption induced by biochar addition (Fig. 6g, h).

#### 4.4. Grain yield, aboveground N uptake and reactive N losses

Our results showed that grain yield and aboveground N uptake were not significantly influenced under  $I_{AWD}$  (Table 6). These are similar to the study of Ye et al. (2013), who reported that  $I_{AWD}$  promoted root growth, but grain yield was not significantly influenced. In our experiment, the influence of grain yield and aboveground N uptake by  $I_{AWD}$  may be related to that  $I_{AWD}$  promoted nitrification and increased soil  $NO_3^-$ -N levels, which was easier to be absorbed by roots and thus improved grain yield and N uptake (Kronzucker et al., 1999). In addition,  $I_{AWD}$  increased  $O_2$  concentration in the rice rhizosphere and further promoted root growth and density, thereby slightly increasing grain yield and N uptake (Sha et al., 2022). However, the negative effects of

water stress on grain yield under  $I_{AWD}$  cannot be ignored. Moreover, we found that weeds are prone to occur in the  $I_{AWD}$  paddy system, especially at the tiller fertilizer stage. Although we cleaned up in time, the competition of weeds for water and nutrients may have a negative influence on grain yield and N uptake.

In our study, the main components of reactive N losses were  $NH_3$  volatilization,  $N_2O$  emissions and N leaching.  $I_{AWD}$  decreased inorganic N leaching but stimulated  $NH_3$  volatilization and  $N_2O$  emissions due to low water depth and frequent dry-wet alternation. Biochar addition increased the average soil  $NH_4^+$ -N levels by 25.9 % (2020) and 52.4 % (2021) in both years (Fig. 6e, f), demonstrating that biochar prolonged the soil mineral N retention time and therefore decreased the gaseous and leached N losses from paddy ecosystems by continuously adsorbing inorganic N in topsoil, thus mitigated the negative effects of  $I_{AWD}$ . Among the four treatments,  $I_{AWDB_{20}}$  decreased reactive N losses by 16.7–28.0 %, as compared to  $I_{CFB_0}$ . Therefore, biochar addition at the rate of 20 t ha<sup>-1</sup> can be used as an efficient management technique to improve rice yield and reduce the reactive N losses in AWD paddy systems.

#### 4.5. Implication and future research direction

Rice cultivation consumes large amounts of water and N fertilizer resources, resulting in water shortage and environmental pollution. This phenomenon is particularly serious in Liaohe Plain where water percolation is large and water resources are becoming increasingly scarce.  $I_{AWD}$  can significantly reduce rice water consumption in rice paddy ecosystems but stimulates the gaseous and leached N losses. Biochar addition is an effective measure for reducing N fertilizers such as  $N_2O$  emissions,  $NH_3$  volatilization and N leaching. Chen et al. (2022) reported that biochar addition into AWD ecosystems had a positive influence on mitigating  $N_2O$  emissions,  $NH_3$  volatilization and enhancing soil N fixation over three years. In this experiment, we further found that biochar addition into rice paddy fields had an immense potential for

reducing N<sub>2</sub>O emissions and NH<sub>3</sub> volatilization as well as mitigating the high N leaching caused by large water percolation. Biochar addition coupled with I<sub>AWD</sub> suppresses the negative effect of I<sub>AWD</sub> alone on N<sub>2</sub>O emissions and NH<sub>3</sub> volatilization, achieving significant benefits in mitigating gaseous and leached N losses, and therefore solving the shortage of water resources and environmental pollution and promoting green and sustainable development of agriculture.

Different regions, climates and different soil types can contribute to different results, and this experiment was based on a 2 years field in Northeast China. An in-depth mechanistic study on the long-term (5 years and 10 years) impacts of biochar addition on gaseous and leached N losses in an I<sub>AWD</sub> paddy ecosystem merits further exploration.

## 5. Conclusions

I<sub>AWD</sub> significantly decreased N leaching and reactive N losses in 2021, although increased NH<sub>3</sub> volatilization in 2021 and cumulative N<sub>2</sub>O emissions in two years as compared to I<sub>CF</sub>. Meanwhile, biochar addition mitigated NH<sub>3</sub> volatilization and cumulative N<sub>2</sub>O emissions in both years and decreased N leaching and reactive N losses in 2021. Therefore, biochar addition could weaken the negative impact of I<sub>AWD</sub> on NH<sub>3</sub> volatilization and N<sub>2</sub>O emissions while enhancing soil N fixation, and thus decreasing reactive N losses. In addition, I<sub>AWD</sub>B<sub>20</sub> reduced cumulative NH<sub>3</sub> volatilization, N leaching and reactive N losses by 7.7–13.7%, 19.2–32.3% and 16.7–28.0%, although increased N<sub>2</sub>O emissions by 68.0–72.2% as compared with I<sub>CF</sub>B<sub>0</sub> in two years. In conclusion, biochar combined with I<sub>AWD</sub> is a potential and sustainable strategy for conserving water, enhancing soil N fixation and mitigating reactive N losses in paddy fields. The long-term observations of biochar addition on reactive N losses and grain yield in I<sub>AWD</sub> paddy fields remain to be further explored in the future.

## CRedit authorship contribution statement

**Hongyuan Zhu:** Software, Investigation. **Taotao Chen:** Supervision, Project administration, Funding acquisition. **Feng Zhang:** Writing – review & editing, Writing – original draft, Investigation, Formal analysis. **Guimin Xia:** Supervision, Investigation. **Daocai Chi:** Investigation. **Wanning Dai:** Investigation, Data curation, Conceptualization. **Wanting Zhang:** Investigation. **Zhe Wang:** Software, Investigation.

## Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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## Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at [doi:10.1016/j.agwat.2024.109108](https://doi.org/10.1016/j.agwat.2024.109108).

## Data Availability

Data will be made available on request.

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