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In situ soil net nitrogen mineralization in coastal salt marshes (*Suaeda salsa*) with different flooding periods in a Chinese estuary

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ABSTRACT

Flooding periods can be one of the most important factors influencing nitrogen (N) biogeochemical processes in wetlands ecosystem. We conducted a field study using *in situ* incubation method to investigate the seasonal dynamics of soil net N mineralization in three coastal salt marshes (*Suaeda salsa*) with different flooding periods (*i.e.*, short-term (STF), seasonal (SF), and tidal (TF) flooding wetland) in the Yellow River Delta. Selected soil inorganic N pools (ammonium, nitrate and inorganic N) and N transformation (mineralization, nitrification and ammonification) rates in the top 0–10 cm soils were repeatedly quantified from April to October. Clear seasonal patterns in inorganic N pools and transformation rates were observed in accord with the seasonal variations of temperature and moisture. Generally, higher levels of soil inorganic nitrogen, ammonium nitrogen (NH_4^+-N) and nitrate nitrogen (NO_3^--N) occurred in the early-growing season (April), and NH_4^+-N contents got a small accumulative peak in midsummer (September). The lower rates (negative) of net mineralization (R_{min}), nitrification (R_{nit}) and ammonification (R_{amm}) were observed in the early-growing season (April–June) and fall (September–October), whereas higher values (positive) in midsummer (August–September). Flooding had a significant influence on inorganic N pools (except for NH_4^+-N) and transformation rates ($p < 0.05$). R_{min} values in SF wetland were significantly higher in the August-September period than those in other incubation periods. R_{nit} values in TF wetland exhibited a small variation and the highest value occured in the June–August period. The results of principal component analysis showed that soil samples were clearly divided into two groups before and after flow-sediment regulation. After flooding events, the R_{min} and R_{amm} values generally increased in the three wetlands, whereas a significant decrease in R_{nit} values was observed in SF wetland ($p < 0.05$), thus the differences in NO_3^--N among these wetlands were eliminated. These results suggested that seasonal variations in temperature and moisture are important factors influencing inorganic N pools and transformation rates.

1. Introduction

Coastal wetlands are the transitional zones between the terrestrial and marine ecosystems, which are characterized by high biodiversity, productivity and susceptibility (Mitsch and Gosselink, 2015), and the "sinks", "sources" and "transformations" of chemical elements (*i.e.*, nitrogen and phosphorous) (Flynn, 2008). Nitrogen often acts as a limiting nutrient for coastal salt marshes, and nitrogen availability is always considered to have considerable impacts on the structure and productivity of plant community (Mitsch and Gosselink, 2015). Nitrogen mineralization is the key process that converts organic nitrogen into inorganic N by soil microorganisms (Niedermeier and Robinson, 2007), which controls the N bioavailability for wetland plants and includes both ammonification and nitrification processes. Ammonification is the process in which organic matter is converted to ammonia by microorganisms. Meanwhile, the subsequent nitrification of NH_4^+-N can also influence N availability by controlling N loss (Groffman and Tiedje, 1989) or by changing the relative availability of NH_4^+-N and NO_3^--N since plants have different capabilities to assimilate inorganic N forms (Mendelssohn, 1979). Therefore, a better understanding of soil nitrogen mineralization in coastal salt marshes can contribute to improving soil fertility and quality management to maintain coastal wetland ecosystem health.

Nitrogen cycle, especially N mineralization and inorganic N pool, is sensitively impacted by wetland hydrology and environmental

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Table 1
In situ incubation periods in three wetlands.

Incubation period	April–June	June–August	August–September	September–October
date of incubation (2008)	4.26–6.7	6.7–8.9	8.9–9.11	9.11–10.14
days of incubation	42	32	34	33

factors (Gao et al., 2012). The frequent drying and wetting alternation will stimulate decomposition of soil organic matter (SOM) and increase inorganic N but exacerbated N losses (Birch, 1960; Sahrawat, 1980). Bai et al. (2005, 2007) presented that flooding duration and frequencies could influence soil nitrogen distributions, because drying and wetting cycles can greatly influence soil properties and soil oxic and anoxic periods (Neill, 1995), and then affect SOM decomposition and microbial activities (Turner and Patrick, 1968). Gao et al. (2012) observed that R_{min} increased and NO_3^--N decreased after freshwater flooding events due to the potential for anaerobic nutrient cycling promoted by flooding. Liu et al. (2012) showed higher levels of total nitrogen at 10 cm water level treatment in August and September, and at 0 cm water level in October in the Yellow River Delta. Salinity is another vital factor influencing N mineralization in coastal wetlands. Pathak and Rao (1998) suggested that increasing salinity will suppress N mineralization. However, Gao et al. (2014) demonstrated higher salinity promoted the N mineralization. Zeng et al. (2013) reported the increasing rates of nitrification with increasing salinity under a threshold. Temperature and moisture are often considered to be the two important factor influencing N mineralization (Grenon et al., 2004). In general, the sensitivity of nitrogen mineralization to temperature is maximal at 25 °C and the optimal moisture content is between 50% and 100% of field capacity (Guntiñas et al., 2012). Additionally, plants can affect N mineralization through nutrients uptake and competition with microbes for nutrients (Vitousek, 1982). However, little information is available on the combined influence of seasonal variation and flooding periods on soil inorganic N pools and net N mineralization, nitrification and ammonification rates of *Suaeda salsa* in coastal salt marshes.

Therefore, the primary objectives of this study were: (1) to investigate seasonal variations in soil inorganic nitrogen (NH_4^+-N and NO_3^--N) and *in situ* net nitrogen mineralization, nitrification and ammonification rates in salt marshes with different flooding periods in the Yellow River Delta; and (2) to identify the key factors influencing soil N pools and net nitrogen transformation rates.

2. Materials and methods

2.1. Site description

The study was conducted in *Suaeda salsa* salt marshes in the Yellow River Delta Nature Reserve (37°35′–38°12′N, 118°33′–119°20′E), Shandong province of China, during the period from April to October in 2008, which is an important transit point, wintering habitat and breeding grounds for bird migration in northeast Asia and west Pacific, and has been included in the Ramsar List of Wetlands of International Importance. The Yellow River Delta has a typical monsoon climate with an annual mean air temperature of 11.9 °C and 196 frostless days. The annual mean rainfall is 640 mm and mainly concentrates in summer (from June to August) and the annual mean evaporation is 1962 mm. The Fluvo-aquic soil and saline soil are the main soil types in the study area. The predominant vegetation includes *Suaeda salsa, Phragmites australis, Triarrhena sacchariflora and Tamarix chinensis*. *Suaeda salsa* (*Chenopodiaceae*) is a succulent halophytic herb (Wang et al., 2004), and generally germinates in late April, blooms in July, matures in late September, and completely perishes in November. Complex wetlands hydologcial condition were developed in the study area due to freshwater flooding caused by flow-sediment regulation and the irregular semidiurnal tide. Three *Suaeda salsa* wetlands with different flooding periods were selected in this study. Short-term flooding wetland (STF) can only be flooded for approximately one month after flow-sediment regulation, whereas seasonal flooding wetland (SF) can be flooded and last approximately three to four months, and the freshwater from the Yellow River is their main surface water source. Tidal flooding wetland (TF) can be flooded by tidal seawater twice one day.

2.2. *In situ incubation experimental design*

In situ incubation can reflect the natural state of the soil net nitrogen mineralization, nitrification and ammonification. In this study, we choose polyvinyl chloride (PVC) top-closed pipe *in situ* incubation method to ensure soil in the pipe had the same structure and temperature as the outside soil. Soil sampling plots with three replicates were randomly set in each of three flooding wetlands, and a total of nine sampling sites were assembled. The *in situ* incubation experiment lasted for four periods from April to November of 2008 (Table 1). At the beginning of each incubation period, three incubation tubes (25 cm in length and 5 cm in diameter) (tubes A, B, C) were inserted in each sampling plot after removing plant litter layer on the surface soil, with 10 cm left above away from the surface soil. The incubation tube A was removed immediately, and soil samples were collected for the determination of the initial values of NH_4^+-N, NO_3^--N and other soil properties, *e.g.*, soil SOM, salinity, soil total carbon (TC) and total nitrogen (TN), and the incubation tube B was remained for next 30–40 days incubation period in the field. The incubation tube C was stayed in the soil and sealed using a plastic breathable membrane on the top to ensure the soil aerobic respiration and prevent leaching. Meanwhile, another soil core (100 cm^3) in each sampling plot was collected to determine soil bulk density (BD) and water content (WC). At the end of each incubation period, the incubation tubes B and C were pulled out and the soil in the tubes were simultaneously collected for physical and chemical analysis. Then inserted another three new incubation tubes to achieve the purpose of continuous monitoring in the field.

2.3. *Soil sample collection and analysis*

All soil samples in the incubation tubes were placed in polyethylene bags and brought to the laboratory. Some fresh soils were sieved to remove the coarse debris and stones and used to determine the NH_4^+-N and NO_3^--N contents. Approximately 10 g (fresh mass) soil samples were extracted with 50 mL of 2 M KCl solution. Extracts were mixed for 1 h on the rotational shaker and conducted on an automated flow injection analysis AA3 (Bran + Luebbe, Germany) to determine soil NH_4^+-N and NO_3^--N contents. Soil inorganic nitrogen (SIN) was expressed as the sum of NH_4^+-N and NO_3^--N. The reminding soil samples were air-dried in the laboratory at room temperature for two or three weeks and ground through a 2-mm sieve to remove the coarse debris and stones for the determination of soil physical and chemical properties. Soil pH and salinity (SAL) were measured using a pH meter and salinity meter, respectively (soil/water, 1:5). SOM was determined using Walkley and Bland method (Walkley and Black, 1934). Soil BD and WC were determined by drying soil samples at 105 °C for 24 h in an oven.

2.4. Data calculation

Net N mineralization, nitrification and ammonification were calculated as the difference between post- and pre-incubation inorganic nitrogen (NH_4^+-N and NO_3^--N), NO_3^--N and NH_4^+-N contents, respectively. Net N mineralization (R_{min}), net nitrification (R_{nit}) and net ammonification (R_{amm}) rates (mg N kg^{-1} d^{-1}) were calculated by following equations:

For a time interval $\Delta t = t_1 - t_0$

$$R_{min} = \left(SIN_{C,1} - SIN_{C,0}\right)/\Delta t$$

$$R_{nit} = \left(NO_3^- - N_{C,1} - NO_3^- - N_{C,0}\right)/\Delta t$$

$$R_{amm} = \left(NH_4^+ - N_{C,1} - NH_4^+ - N_{C,0}\right)/\Delta t$$

Where t_1 and t_0 represent the pre- and post-incubation time, respectively; Δt represent incubation time; $SIN_{C,0}$, NO_3^--$N_{C,0}$ *and* NH_4^+-$N_{C,0}$ represent SIN, NO_3^--N and NH_4^+-N contents at the beginning of incubation, respectively. $SIN_{C,1}$, NO_3^--$N_{C,1}$ *and* NH_4^+-$N_{C,1}$ represent SIN, NO_3^--N and NH_4^+-N contents at the end of the incubation, respectively.

2.5. Statistical analysis

One-way ANOVA analysis was conducted to identify the differences in inorganic N concentrations, the rates of mineralization, ammonification and nitrification among different sampling dates and different wetlands types. The effects of sampling dates and wetland types and their interactions on N mineralization were tested by two-way ANOVA analysis. Principal component analysis (PCA) was performed to analyze the variations of samples using CANOCO for windows 4.5 software. The differences were considered to be significant if $p < 0.05$. Statistical analysis was carried out using SPSS 16.0 software package. Bar and line graphs were performed using Origin 9.0 software package.

3. Results

3.1. Seasonal dynamics of soil inorganic N pools

Both SIN and NO_3^--N contents in the top 10 cm soils in the three wetlands were significantly affected by sampling dates ($p < 0.01$), wetland types ($p < 0.05$) and their interactions ($p < 0.05$), whereas NH_4^+-N contents was only significantly related to sampling dates ($p < 0.01$) (Table 2).

The NH_4^+-N contents in STF, SF, and TF wetlands in incubation periods ranged from 1.06 to 6.27 mg/kg, from 0.95 to 6.45 mg/kg, and from 0.97 to 5.18 mg/kg, respectively (Fig. 1A). In TF wetland soils, higher NH_4^+-N contents were observed in April and September, and lower levels were observed in June, August, and October ($p < 0.05$). In STF wetland soils, higher NH_4^+-N levels only appeared in April ($p < 0.05$). During the same incubation period, the differences in NH_4^+-N contents among the three wetlands were not significant ($p > 0.05$). During the different incubation periods, NH_4^+-N contents in SF wetland soils showed no significant differences ($p > 0.05$), but changed significantly in both STF and TF wetland soils ($p < 0.05$).

The NO_3^--N contents in STF, SF and TF wetland soils ranged from 0.77 to 8.4 mg/kg, from 0.65 to 4.04 mg/kg, from 0.86 to 2.33 mg/kg, respectively (Fig. 1B). The differences in NO_3^--N contents in soil among the three wetlands during the same incubation period were significant before flow-sediment regulation ($p < 0.05$) and no significant differences were observed after flow-sediment regulation ($p > 0.05$). During the different incubation periods, the NO_3^--N contents in both STF and SF wetland soils showed significant differences ($p < 0.05$), whereas no significant differences were observed in TF wetland soils ($p > 0.05$). The maximum values were observed in STF wetland soils in April and in SF wetland soils in June.

The SIN contents in STF, SF and TF wetlands showed significant differences with increasing incubation periods ($p < 0.05$) (Fig. 1C). During the same incubation period, significant differences were only observed in April ($p < 0.05$). In STF wetland, SIN contents exhibited the peak values in both April and August. Comparatively, the maximum SIN value appeared in April and showed a small peak in August in TF wetland.

In STF and SF wetlands, SIN was mainly composed of NO_3^--N (about 50–60% of SIN), especially higher percentage in April and September ($p < 0.05$). However, in TF wetland, NO_3^--N accounted for approximately 38% of SIN and the maximum value was observed in April.

3.2. Seasonal dynamics of net nitrogen mineralization, nitrification and ammonification rates

Both R_{min} and R_{nit} in the top 10 cm soils were significantly influenced by sampling dates, wetland types and their interactions ($p < 0.01$), whereas the R_{amm} was only significantly affected by sampling periods and their interactions ($p < 0.01$) (Table 3).

In three wetlands, R_{min} were significantly different during different incubation periods ($p < 0.01$) (Table 3, Fig. 2). R_{min} in three wetlands showed similar changing trend and ranged from −0.23 to 0.08 mg kg^{-1} d^{-1} in STF wetland, from −0.14 to 0.04 mg kg^{-1} d^{-1} in TF wetland, and from −0.14 to 0.24 mg kg^{-1} d^{-1} in SF wetland, respectively (Fig. 2A). The R_{min} values during the April–June, June–August and September–October incubation periods were negative in addition to the positive values during the August–September period. The maximum R_{min} values were observed in mid-summer (August–September), whereas the minimum R_{min} values appeared in both April–June and September–October incubation periods. In the April–June incubation period, the R_{min} values in STF wetland were significantly lower than those in SF and TF wetlands ($p < 0.05$). In contrast, the R_{min} values in TF wetland were significantly higher than those in STF and SF wetlands in the August–September incubation period ($p < 0.05$).

The R_{nit} and R_{min} values showed the similar changing tendency in STF wetland (Fig. 2B). In SF wetland, R_{nit} showed the positive maximum value in the August–September incubation period, with a small positive peak value in the April–June and September–October incubation periods and the minimum value in the June–August incubation period. However, the R_{nit} value in TF wetland was positive only in the June–August incubation period, which was significantly higher than those in other incubation periods ($p < 0.05$). Generally, the R_{amm} and R_{min} values in the three wetlands nearly exhibited the similar changing tendency (Fig. 2C).

3.3. Principal component analysis

Principal component analysis (PCA) was performed to reveal the relationships between N transformation rates and soil physical-chemical properties (Fig. 3). These variables were grouped into four principal components with eigenvalue >1, interpreting 84.88% of the total variance (Table 4). The first PCA axis, explaining 40.09% of the total variation, exhibited high factor loadings for TN, TC, and SOM. The second PCA axis loaded heavily on NH_4^+-N, Amin and WC, and explained 24.29% of the total variance. The third and fourth PCA axises were closely related to SOM, SAL, pH and NO_3^--N, respectively.

Table 2
Two-Way ANOVA analysis results for all soil samples during the study period.

	SIN		NH_4^+-N		NO_3^--N	
	F	P	F	P	F	P
Sampling dates	22.95	0.000**	18.49	0.000**	15.75	0.000**
Wetland types	3.66	0.042*	0.24	0.788	16.29	0.000**
Sampling dates × wetland types	3.23	0.014*	1.43	0.238	6.64	0.000**

* Represent significant level of $p < 0.05$.
** Represent significant level of $p < 0.01$.

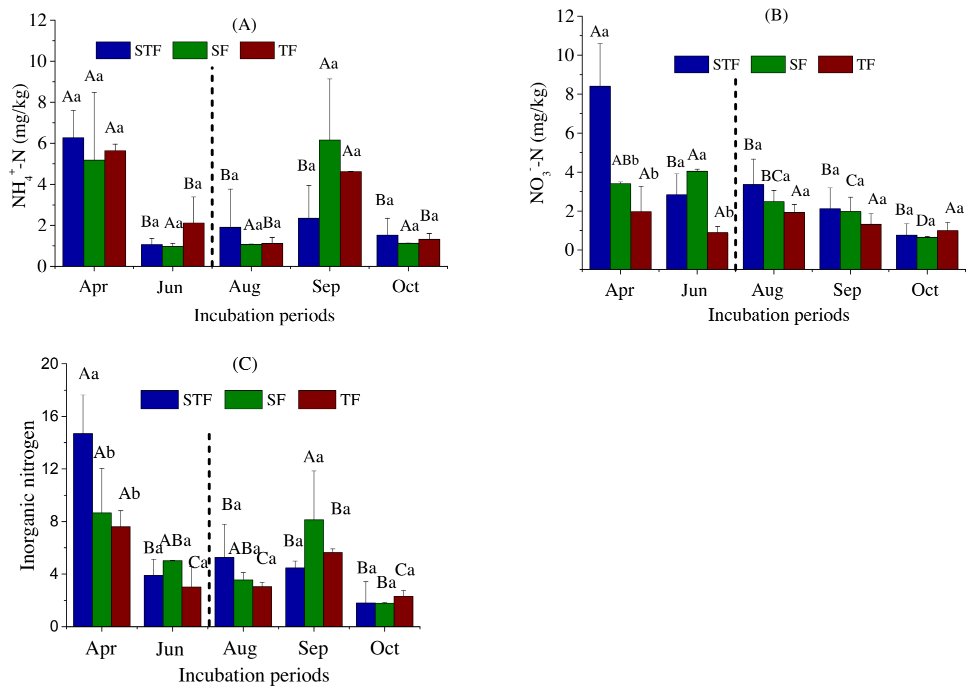

Fig. 1. Seasonal variations in NH_4^+-N (A), NO_3^--N (B) and inorganic N (C) contents (mg/kg) in the top 10 cm soils in SFT, ST, and TF wetlands (ABC represent the differences in organic N levels between different sampling periods in the same wetland; abc represent the differences in organic N levels in the same incubation between different wetlands; the dotted line represents the date when flow-sediment regulation was operated).

Table 3
Two-way ANOVA analysis results for all soil samples during the study period.

	R_{min}		R_{nit}		R_{amm}	
	F	P	F	P	F	P
Sampling dates	76.10	0.000**	17.77	0.000**	143.72	0.000**
Wetland types	6.80	0.002**	9.42	0.000**	1.48	0.234
Sampling dates × wetland types	7.77	0.000**	12.81	0.000**	12.27	0.000**

** Represent significant level of $p < 0.01$.

Table 4
Principal component analysis and factor loadings of soil properties.

	PC1	PC2	PC3	PC4
WC	0.594	0.634	−0.161	0.194
BD	−0.675	0.512	−0.056	0.361
SOM	0.781	0.179	0.515	0.163
SAL	−0.720	−0.312	0.533	−0.004
NH_4^+-N	0.370	−0.847	−0.065	−0.161
NO_3^--N	0.338	−0.552	−0.355	0.467
TN	0.940	0.000	0.252	0.128
TC	0.847	0.316	0.318	−0.029
C/N ratio	−0.747	0.175	0.434	0.016
pH	0.214	0.335	−0.138	−0.813
Amin	−0.135	0.784	−0.225	0.077
% of Variance	40.094	24.287	10.288	10.210

As shown in Fig. 3, soil samples could be clearly divided into two groups based on flow-sediment regulation. Group 1 represents soil samples collected before flow-sediment regulation and Group 2 represents soil samples collected after flow-sediment regulation. Group 1 was mainly affected by the concentrations of initial NH_4^+-N and NO_3^--N and Group 2 was greatly affected by SAL, WC, BD and C/N.

4. Discussion

4.1. Seasonal patterns of nitrogen pools and transformation

This study demonstrated clear patterns of seasonal variations in inorganic N pools and N mineralization, ammonification and nitrification rates in coastal *Suaeda salsa* wetlands. The lowest values of R_{min}, R_{nit} and R_{amm} were observed in the early growing

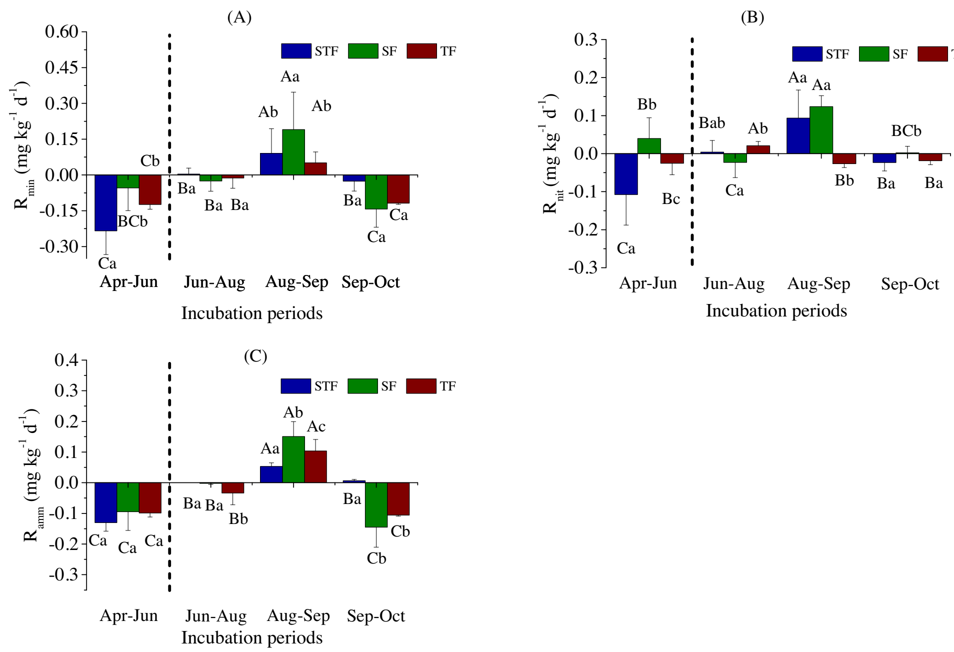

Fig. 2. Seasonal variations in net nitrogen mineralization rates (R_{min}) (A), net nitrification rates (R_{nit}) (B) and net ammonification rates (R_{amm}) (C) in STF, SF, and TF wetlands (ABC represent the differences in R_{min}, R_{nit} and R_{amm} values between different incubation periods in the same wetland; abc represent the differences in R_{min}, R_{nit} and R_{amm} values between different wetlands in the same incubation period; the dotted line represents the date when flow-sediment regulation was operated).

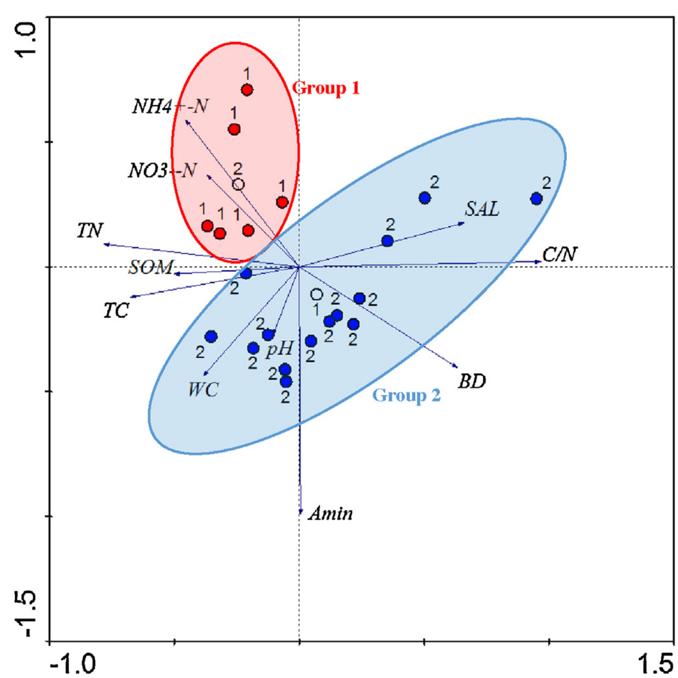

Fig. 3. The ordination diagram from the principal component analysis (PCA) of 11 soil properties indexes and 22 soil samples in the Yellow River Delta. The Group 1 represents soil samples collected before flow-sediment regulation and Group 2 represents soil samples collected after flow-sediment regulation. (TN: total nitrogen; TC: total carbon; SOM: soil organic matter; WC: water content; Amin: the amount of mineralization; BD: bulk density; SAL: salinity; C/N: the ratio of carbon and nitrogen).

season (April–June) and fall (September–October), and the maximum values appeared in midsummer (August–September). Microbial biomass and activities motivated by seasonal air temperature and soil moisture were likely a dominant factor stimulating the processes (Wilson and Jefferies, 1996). More appropriate temperature (almost 25 °C) and higher moisture (26%) in summer could enhance the amount and activities of microorganisms and enzymes, thus stimulating the rates of soil N transformation (Schimel et al., 2004). The increases in available C and N through decomposition motivated by high temperature and moisture might have contributed to N transformation since available C and N can provide substrate and energy for microorganisms (Kushwaha et al., 2015). Gao et al. (2012) observed the higher values of microbial activities and SOM contents in summer in the *Phragmites australis* wetlands. This agrees with the results by Hu et al. (2015), who presented that R_{min} values showed the maximum value in summer with higher temperature and moisture and decreased in the early-growing season and fall. Sierra (1992) reported that the seasonal fluctuations of soil temperature were positively correlated with N mineralization in Pergamino Experimental Station. Comparatively, a different conclusion was obtained by Corre et al. (2002), who presented that nitrification was negatively influenced by soil moisture in growing seasons. These discrepancies might be attributed to the difference in seasonal patterns of temperature and moisture among different study areas. During the early growing season (April–June), the contents of SIN, NH_4^+-N and NO_3^--N exhibited higher values but R_{min}, R_{nit} and R_{amm} were negative, which could be associated with freezing-thawing event which would physically disrupt microbial cells and release substrates containing nutrient being protected by soil aggregates and temperature (Edwards and Cresser, 1992; Ivarson and Sowden, 1970). In addition, the melting of snow and the absence of plant uptake could extend the effect of freezing-thawing events (Zhou et al., 2009).

In growing seasons, NH_4^+-N contents increased and NO_3^--N contents decreased. N leaching and plant uptake could be the probable explanation. Bai et al. (2006) presented that NO_3^--N was much easier to be leached than NH_4^+-N in soil because NO_3^- is negatively charged. This was consistent with the result by Gao et al. (2012), who also observed lower NO_3^--N and higher NH_4^+-N in the *Phragmites australis* wetland soils. We also studied the plant uptake of N under flooding using ^{15}N isotope method in the study area, and reported that the SIN of plant uptake were mainly composed of NO_3^--N in the Yellow River Delta (Unpublished data).

4.2. Hydrological effects on nitrogen mineralization and nitrification

The results of ANOVA showed that wetland types had significant influence on inorganic N pool (except for NH_4^+-N), R_{min} and R_{nit} values in the Yellow River Delta ($p < 0.05$). The results of PCA exhibited clear differences in soil samples before and after flow-sediment regulation. These results indicated that freshwater input and flooding might be vital factors controlling the N transformation processes in this region. However, the effects of flooding on inorganic N pool and transformation rates differed from the three wetlands with different flooding periods, which could be induced by the synthetic effects of flooding, seasonal variation in environmental factors, and complex interactions.

Before flow-sediment regulation, the R_{min} values in STF wetland was significantly lower than those in SF and TF wetlands ($p < 0.05$), and the differences in R_{min} values among wetlands were eliminated after flow-sediment regulation. Because N mineralization is a process controlled by soil microorganisms, longer drying period in STF wetland would cause the death of microorganisms and result in lower R_{min} values (Iovieno and Baath, 2008). In addition, drying could destroy the pore structure and amplify the negative effect (Zheng, 2013). In the August–September incubation period, SF wetland was continuously flooded but not flooded in STF wetland, and the R_{min} values exhibited higher values in SF wetland than those in STF wetland, indicating that flooding triggered R_{min} in growing seasons. This is in agreement with the result by Neill (1995), who presented that the net N mineralization rates in flooding marsh were up to 4 times greater than that in nonflooding marshes. Our results also supported the findings of Ono (1989), who found flooding would increase soil pH and promote the N mineralization. Miller et al. (2005) studied the effects of flooding frequency on N transformation and reported that the N mineralization rates under a four-week drying-wetting circulation were significantly lower compared with a two-week drying-wetting circulation. However, lower R_{min} values in TF wetland with higher flooding frequency were observed in the August–September incubation period. This might be associated with higher salinity in TF wetland, which could suppress soil microbial communities and their activities (Yuan et al., 2007), and reduce N mineralization (Walpola and Arunakumara, 2010). However Gao et al. (2014) observed that high salinity enhanced soil N mineralization in tidal freshwater marshes.

A significant decrease in R_{nit} values was observed in SF wetland after flow-sediment regulation ($p < 0.05$). This was associated with the fact that inundation condition inhibited nitrobacteria from changing NH_4^+-N into NO_3^--N, whereas had a slight influence on the N mineralization that can occur under both aerobic and anaerobic conditions (Wang et al., 2001). The R_{nit} values in TF wetland exhibited small variations and appeared the maximum in the June–August incubation period. Generally, TF wetland soils showed a significantly lower R_{nit} value in the August–September incubation period than STF and SF wetland soils, indicating that higher flooding frequencies could inhibit N nitrification processes because nitrifying microorganisms were sensitive to moisture change (Stark and Firestone, 1995). Franzluebbers et al. (1994) also reported that frequent drying-rewetting cycles would reduce nitrification. However, Fierer and Schimel (2002) presented that the autotrophic nitrifying bacteria increased in oak woodland and grassland soils with frequent drying-rewetting. Therefore, further studies are still needed to testify the hydrological effects in different ecosystems. In addition, the alleviation of salinity stress due to freshwater input affected by the flow-sediment regulation could also offset the negative effects, leading to higher R_{nit} values in TF wetland in the June–August incubation period.

5. Conclusions

The inorganic N pools and rates of mineralization, ammonification and nitrification in coastal *Suaeda salsa* wetlands had clear seasonal patterns. Generally, N mineralization rates began to greatly increase from the April–June incubation period to the August–September incubation period and undergo a change from nitrogen immobilization to nitrogen release, and SIN gradually accumulated. The maximum N mineralization rates appeared in August–September, whereas maximum nitrogen immobilization occurred in October. Flow-sediment regulation can bring a large amount of freshwater into the Yellow River Delta and obviously influence inorganic N pools and N transformations due to the changes in the hydrological conditions in the Yellow River Delta. Generally, soil samples can be clearly divided before and after flow-sediment regulation. Flooding can improve N mineralization, especially frequent drying and rewetting can stimulate N mineralization and reduce N nitrification. Therefore, flooding frequencies and flow-sediment regulation should be taken into account for coastal wetland conservation and restoration. We can modify the key influencing factors before and after flow-sediment regulation to control nitrogen mineralization in order to improve wetlands productivities.

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