Chin. Geogra. Sci. 2018 Vol. 28 No. 3 pp. 443–455
https://doi.org/10.1007/s11769-018-0961-7

Springer Science Press
www.springerlink.com/content/1002-0063

Effects of Reclamation on Soil Carbon and Nitrogen in Coastal Wetlands of Liaohe River Delta, China

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Abstract: To evaluate the influence of wetland reclamation on vertical distribution of carbon and nitrogen in coastal wetland soils, we measured the soil organic carbon (SOC), soil total nitrogen (STN) and selected soil properties at five sampling plots (reed marsh, paddy field, corn field, forest land and oil-polluted wetland) in the Liaohe River estuary in September 2013. The results showed that reclamation significantly changed the contents of SOC and STN in the Liaohe River estuary (P < 0.001). The SOC concentrations were in the order: oil-polluted wetland > corn field > paddy field > forest land > reed marsh, with mean values of 52.17, 13.14, 11.46, 6.44 and 6.16 g/kg, respectively. STN followed a similar order as SOC, with mean values of 1351.14, 741.04, 632.32, 496.17 and 390.90 mg/kg, respectively. Interaction of reclamation types and soil depth had significant effects on SOC and STN, while soil depth had significant effects on SOC, but not on STN. The contents of SOC and STN were negatively correlated with pH and redox potential (Eh) in reed marsh and corn field, while the SOC and STN in paddy field had positive correlations with electrical conductivity (EC). Dissolved organic carbon (DOC), ammonium nitrogen (NH_4^+-N) and nitrate nitrogen (NO_3^--N) were also significantly changed by human activities. NH_4^+-N and NO_3^--N increased to different degrees, and forest land had the highest NO_3^--N concentration and lowest DOC concentration, which could have been caused by differences in soil aeration and fertilization. Overall, the results indicate that reed harvest increased soil carbon and nitrogen release in the Liaohe River Estuary, while oil pollution significantly increased the SOC and STN; however, these cannot be used as indicators of soil fertility and quality because of the serious oil pollution.

Keywords: coastal wetlands; reclamation; soil carbon; soil nitrogen; Liaohe River Delta

Citation: WAN Siang, MOU Xiaojie, LIU Xingtu, 2018. Effects of Reclamation on Soil Carbon and Nitrogen in Coastal Wetlands of Liaohe River Delta, China. *Chinese Geographical Science*, 28(3): 443–455. https://doi.org/10.1007/s11769-018-0961-7

1 Introduction

Coastal wetlands, which are important transition regions between the ocean and land, comprise some of the most productive ecosystems in the world. Moreover, coastal wetlands serve as reservoirs of biodiversity and carbon sinks and provide important ecological functions such as degradation of pollutants and nutrient cycling (Costanza et al., 1998; Zhang et al., 2014), and 230 water bird species, over 25% of the global total, inhabit China's coastal wetlands (Ma et al., 2014). Because of their geographical conditions, coastal regions, especially estuarine deltas, are areas of high anthropogenic activity. Reclamation of coastal wetlands has become a common practice worldwide to provide more space for anthropogenic activity, and large areas of coastal wetlands are now employed for urban, industrial, shipping and agricultural uses.

Worldwide, the development of coastal areas has entered a peak period since the 1950s. Mangroves are typical tropical and subtropical coastal wetlands; how-

Received date: 2017-06-15; accepted date: 2017-09-12
Foundation item: Under the auspices of National Basic Research Program of China (No. 2012CB956100), National Natural Science Foundation of China (No. 41301085)
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ever, previous studies have indicated that these areas have decreased by 30%–50% over the last 50 years because of agriculture and fish culture (Duke et al., 2007; Donato et al., 2011). The coastline of China is expanding seaward 25–30 cm every year, mainly for agricultural use (Iost et al., 2007; Sun et al., 2011; Li et al., 2013), the reclamation rate has increased dramatically to 40 000 ha/yr during 2006–2010 (Ma et al., 2014), and the trend of coastal wetland reclamation will continue for the foreseeable future (Yin et al., 2016) as a means of relieving population pressure and promoting economic growth (Zhang et al., 2011). However, large-scale reclamation activity has also led to unprecedented interference and damage to coastal wetlands, and therefore poses substantial environmental risks and has become a global concern. For example, reclamation can increase soil fertility, but may also trigger coastal eutrophication (Boesch, 2002). Moreover, the increases in tilled land lead to increased food production; however, they also reduce wildlife habitats, and can even lead to the extinction of species (Benoit and Askins, 2002). Other consequences of reclamation include seawater erosion and soil salinity (Utset and Borroto, 2001).

Soil organic carbon (SOC) and soil total nitrogen (STN) are two of the most important indicators used to measure soil fertility and quality (Andrews et al., 2004; Lal, 2004; Schindler, 2006), and they are also important parts of global C and N cycles. Wetlands are important in global carbon dynamics as they store a third of the globe's terrestrial soil organic carbon of 1395 Gt (Poster et al., 1982), and slight changes in SOC can significantly influence atmospheric CO_2 and CH_4 concentrations (Liu et al., 2011). About 56%–70% of all global Nitrous oxide (N_2O) originates from soil (Syakila and Kroeze, 2011). Many studies have indicated that SOC concentrations might differ throughout a given area due to differences in land use types (Don et al., 2011; Wiesmeier et al., 2015; Deng et al., 2016). Shao et al. (2010) indicated that the sequence of SOC contents followed the order vegetable field > paddy field > forest land > dry land > wild grassland, and that SOC contents of all agricultural land use patterns decreased from the surface to deep layers. Coastal wetlands are embanked, drained and then used for a variety of purposes, giving rise to physical and chemical changes in the affected soils (Santín et al., 2007), which will affect the input and decomposition rate of soil organic matter. The accumulation of SOC and STN was positively correlated with reclamation duration (Cui et al., 2012; Zhang et al., 2016). In some studies, coastal wetlands reclamation has been found to have positive effects on SOC and STN accumulation (Jin et al., 2013). On the contrary, some studies indicated that the SOC and STN of coastal reclaimed soils decreased in certain stages (Fu et al., 2014). The input of C to agricultural soils is often lower than that to natural ecosystems in the early period of reclamation (Huang and Song, 2010); therefore, the SOC is rapidly lost in agricultural soils because of soil organic matter (SOM) decomposition (Boesch, 2002).

However, few studies have addressed the effects of different reclamations on the dynamics of SOC and STN in coastal regions. To gain a better understanding of how coastal wetland ecosystems respond to different anthropogenic disturbances, five typical reclaimed regions of the Liaohe River estuary were investigated to gain insight into the effects of different reclamations on SOC and STN dynamics.

2 Materials and Methods

2.1 Site description

This study was conducted in the southern Liaohe River delta (40°40′N–41°25′N, 121°25′E–123°55′E), Northeast China, which is one of the largest warm temperate coastal wetlands of Asia and an important bird habitat. The annual average temperature of the study area is 8.3℃–8.4℃ (Suo et al., 2010), and the annual average precipitation is 611.6–640.0 mm, most of which occurs in summer (Zhang et al., 2009). Soil types in the region are primarily paddy soil, saline soil, meadow soil and swamp soil (Huang et al., 2000). Dominant plants in this area include reeds (*Phragmites australis*), seepweed (*Suaeda salsa*), typha (*Typha orientalis*) and langsdorff small reed (*Calamagrostis angustifolia)*. The main reclamations include agriculture, aquaculture, oil exploitation, forestry and industrial construction. The area contains a total of 70 000 ha of reed marsh, with an annual output of 300 000 t, most of which is used as papermaking material. Additionally, some irrigation water in the region may contain high levels of petroleum hydrocarbons because of oil exploitation (Leng et al., 2006). According to Chen et al. (2010), the coastal area of the Liaohe River estuary increased by 110–143 km^2 during 1979–2003, with a change rate of 9.5–12.2 km^2/yr. However, the area of natural wetlands has dwindled, primarily in response to engineering projects and agricultural development projects.

2.2 Sample collection and analyses

In September 2013, we selected five land use types in the coastal wetland of the Liaohe River estuary, reed marsh(A), paddy field (B), corn field (C), *Ulmu spumila* forest land (D), and reed wetland polluted by oil exploitation (E). Site details are listed in Table 1. Remote sensing data revealed that all sites had a reclamation history of less than 50 years. We collected soil samples to a depth of 100 cm from three soil cores randomly collected in each plot. Soils were sampled at depths of 0–10 cm, 10–20 cm, 20–30 cm, 30–50 cm, 50–70 cm, and 70–100 cm. A total 90 soil samples were collected, placed in polyethylene bags and returned to the laboratory. After removing the plant roots, fauna, and debris by hand, soil samples were air dried at room temperature and then ground until they passed through a 0.149 mm nylon sieve. SOC was measured by the dichromate oxidation method (Lu, 1999), while STN content was analyzed by the Kjeldahl digestion method (Lu, 1999) Additionally, ammonium nitrogen (NH_4^+-N) and nitrate nitrogen (NO_3^--N) was extracted with 2 mol/L KCl and analyzed by a sequence flow analyzer (SKALAR San++, the Netherlands). The dissolved organic carbon (DOC) was determined on a total organic carbon analyzer (TOC-V CPH, Shimadzu). The pH and redox potential (Eh) (soil : water, 1 : 5) was measured with a pH/Eh meter, while electrical conductivity (EC) (soil : water, 1 : 5) was determined using an EC meter.

2.3 Calculation and statistical analyses

Analysis of variance (ANOVA) followed by Duncan's multiple comparison tests ($P < 0.5$) was used to evaluate the effects of land use, soil depth, and their interactions with soil characteristics. Pearson's correlation analysis was used to determine the possible relationships between SOC and STN and other soil properties. For all tests, differences were considered significant at $P < 0.05$. All statistical analyses were conducted using SPSS 18.0, while figures were generated using ORIGIN 8.0.

3 Results

3.1 Effects of wetland reclamation on EC, pH and Eh of soils

There were significant differences in soil EC, pH and Eh among the five reclamation types in the Liaohe River estuary ($P < 0.001$) (Table 2 and Fig. 1). The soil EC in the five plots followed the order oil-polluted wetland > paddy field > reed marsh > forest land > corn field, with mean values of 646.39 µs/cm, 414.22 µs/cm, 305.06 µs/cm, 183.84 µs/cm and 109.21 µs/cm, respectively. The soil EC values in paddy field and oil-polluted wetland were significantly higher than those in the reed marsh ($P < 0.001$), while the soil EC values of corn field and forest stand were significantly lower than those of reed marsh ($P < 0.001$). Soil depth had no significant effect on soil EC, while reclamation type and soil depth had significant interaction effects on EC ($P < 0.05$) (Table 2).

Table 1 Coordinates and reclamation years of sampling site

Sampling site	Coordinates	Reclamation years
Reed marsh	40°55′24.28″N, 121°39′18.26″E	25–30
Paddy field	40°59′01.52″ N, 121°40′15.25″E	10–20
Corn field	40°59′32.22″N, 121°40′21.55″E	10–20
Ulmus pumila forest land	41°01′3.23″ N, 121°40′21.01″E	10–20
Oil-polluted wetland	40°57′46.20″N, 121°40′23.10″E	15–20

Table 2 Results of two-way ANOVA of effects of reclamation type, soil depth and their interaction for nine factors

	SOC	STN	C/N	DOC	NH_4^+-N	NO_3^--N	EC	pH	Eh
Reclamation type	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001
Soil depth	0.003	0.140	0.001	0.742	0.087	0.835	0.255	0.005	0.004
Type × Depth	0.032	0.024	0.015	0.891	0.099	0.367	0.023	0.180	0.173

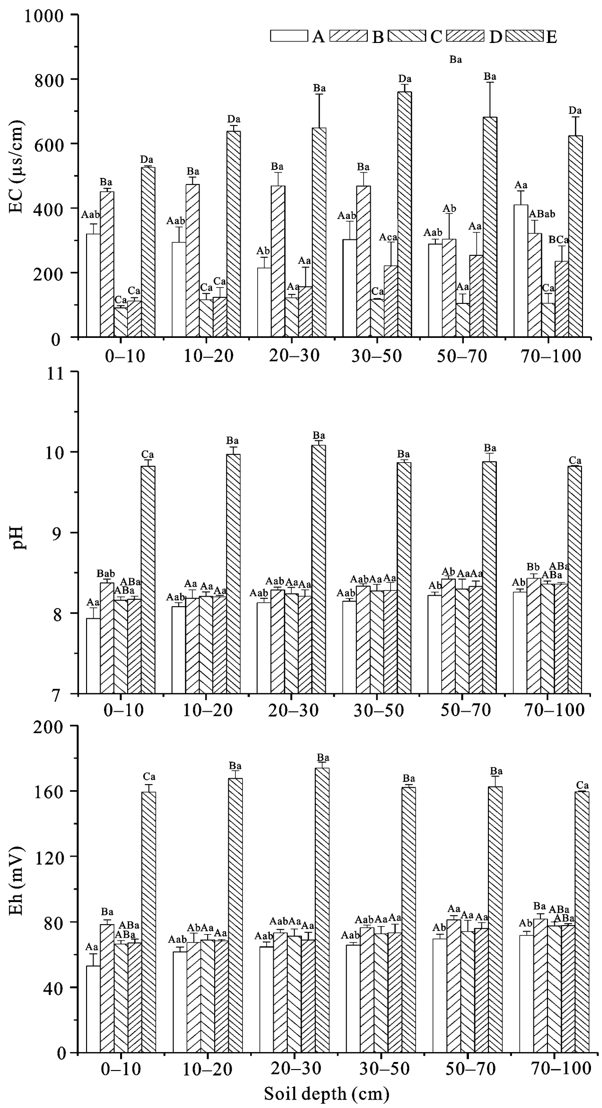

Fig. 1 Effects of wetland reclamation on electrical conductivity (EC), pH and redox potential (Eh) in different soil layers. A: reed marsh; B: paddy field; C: corn field; D: forest land; E: Oil-polluted wetland, similarly hereinafter. Different uppercase letters indicate a significance difference at the same soil depth in different kinds of soil, while different lowercase letters indicate a significance difference at different soil depths in the same kind of soil ($P < 0.05$), similarly hereinafter

The soil pH in the five plots followed the order oil-polluted wetland > paddy field > forest land > corn field > reed marsh, with mean values of 9.900, 8.340, 8.258, 8.257 and 8.130, respectively. The soil Eh in the five plots followed the same order as pH, with mean values of 164.17 mV, 76.39 mV, 71.89 mV, 71.81 mV and 64.39 mV, respectively. The soil pH and Eh of each layer in oil-polluted wetland were significantly higher than those in other plots ($P < 0.01$), while there were no significant differences between the other four plots. Soil depth had significant effects on soil pH and Eh ($P < 0.01$), while the interaction of reclamation type and soil depth did not have significant effects on pH and Eh ($P > 0.05$). There were positive correlations between pH and Eh in all plots ($P < 0.01$) (Table 2), as well as between EC, pH and Eh in forest land ($P < 0.01$).

3.2 Effects of wetland reclamation on SOC, STN and C/N of soils

Significant effects were observed in SOC, STN and C/N among the five types of soils ($P < 0.001$) (Fig. 2 and Table 3). The SOC in the five plots followed the order oil-polluted wetland > corn field > paddy field > forest land > reed marsh, with mean values of 52.17 g/kg, 13.14 g/kg, 11.46 g/kg, 6.44 and 6.16 g/kg, respectively. The SOC values in oil-polluted wetland were significantly higher than those of other types of soils ($P < 0.001$). Additionally, the SOC values of paddy field ($P < 0.05$) and corn field ($P < 0.01$) differed significantly from those of the reed marsh, while the SOC of forest land did not differ significantly from that of reed marsh ($P > 0.05$). Soil depth had significant effects on SOC ($P < 0.01$), and the interaction of reclamation type and soil depth also had a significant effect on SOC ($P < 0.05$). There were positive correlations between SOC and STN in all plots ($P < 0.05$) (Table 3). In reed marsh and corn field, there were negative correlations between SOC and pH, Eh ($P < 0.01$), while there were positive correlations between SOC and EC in paddy field ($P < 0.01$).

The STN in the five plots followed the same order as SOC, with mean values of 1351.14 mg/kg, 741.04 mg/kg, 632.32 mg/kg, 496.17 mg/kg and 390.90 mg/kg, respectively. The STN values of reed marsh were lower than those of the other four types of soils. Additionally, there were significant differences between oil-polluted wetland ($P < 0.001$), corn field ($P < 0.05$) and reed marsh. Soil depth had no significant effect on STN ($P > 0.05$), while reclamation type and soil depth had significant interaction effects on STN ($P < 0.05$). In reed marsh and corn field, there were negative correlations between STN and pH, Eh ($P < 0.01$), while there were positive correlations between STN and EC in paddy field ($P < 0.01$).

Table 3 Pearson's correlation coefficients of soil characteristics

Parameter	SOC	STN	C/N	DOC	NH_4^+-N	NO_3^--N	EC	pH	Eh
Site A									
SOC	1								
STN	0.993**	1							
C/N	0.421	0.323	1						
DOC	0.212	0.223	0.05	1					
NH_4^+-N	0.816**	0.819**	0.235	–0.194	1				
NO_3^--N	0.533*	0.546*	0.079	0.567*	0.34	1			
EC	0.073	0.088	–0.133	0.321	–0.142	0.083	1		
pH	–0.745**	–0.729**	–0.468	–0.149	–0.653**	–0.358	0.294	1	
Eh	–0.754**	–0.739**	–0.466	–0.152	–0.658**	–0.359	0.279	1.000**	1
Site B									
SOC	1								
STN	0.973**	1							
C/N	0.813**	0.689**	1						
DOC	0.736**	0.639**	0.832**	1					
NH_4^+-N	0.045	0.015	0.11	–0.288	1				
NO_3^--N	0.051	0.053	0.01	–0.385	0.852**	1			
EC	0.791**	0.772**	0.789**	0.742**	–0.087	–0.136	1		
pH	–0.417	–0.402	–0.475*	–0.401	0.083	0.185	–0.403	1	
Eh	–0.426	–0.413	–0.482*	–0.411	0.075	0.174	–0.409	0.999**	1
Site C									
SOC	1								
STN	0.969**	1							
C/N	–0.107	–0.324	1						
DOC	0.36	0.402	–0.3	1					
NH_4^+-N	0.597*	0.625**	–0.313	0.327	1				
NO_3^--N	0.296	0.359	–0.174	0.121	0.195	1			
EC	0.345	0.335	–0.27	0.504*	0.183	–0.225	1		
pH	–0.772**	–0.833**	0.404	–0.328	–0.633**	–0.232	–0.059	1	
Eh	–0.774**	–0.836**	0.409	–0.299	–0.645**	–0.236	–0.066	0.998**	1
Site D									
SOC	1								
STN	0.984**	1							
C/N	0.846**	0.770**	1						
DOC	0.555*	0.549*	0.518*	1					
NH_4^+-N	ND	0.007	0.104	–0.035	1				
NO_3^--N	–0.378	–0.374	–0.256	0.117	–0.118	1			
EC	0.381	0.38	0.374	0.518*	0.228	0.002	1		
pH	0.318	0.283	0.314	0.374	0.222	–0.01	0.878**	1	
Eh	0.298	0.265	0.293	0.376	0.217	0.005	0.878**	0.999**	1
Site E									
SOC	1								
STN	0.570*	1							
C/N	0.185	–0.654**	1						
DOC	0.404	0.439	–0.272	1					
NH_4^+-N	0.368	0.588*	–0.345	0.012	1				
NO_3^--N	–0.333	–0.123	–0.193	–0.243	0.154	1			
EC	0.325	0.325	–0.223	0.417	–0.022	–0.007	1		
pH	–0.108	–0.255	0.125	–0.207	–0.450	0.327	0.270	1	
Eh	–0.094	–0.247	0.129	–0.203	–0.460	0.315	0.278	0.999**	1

Note: *: P value is significant at <0.05; **: P value is significant at <0.01; ND: no data

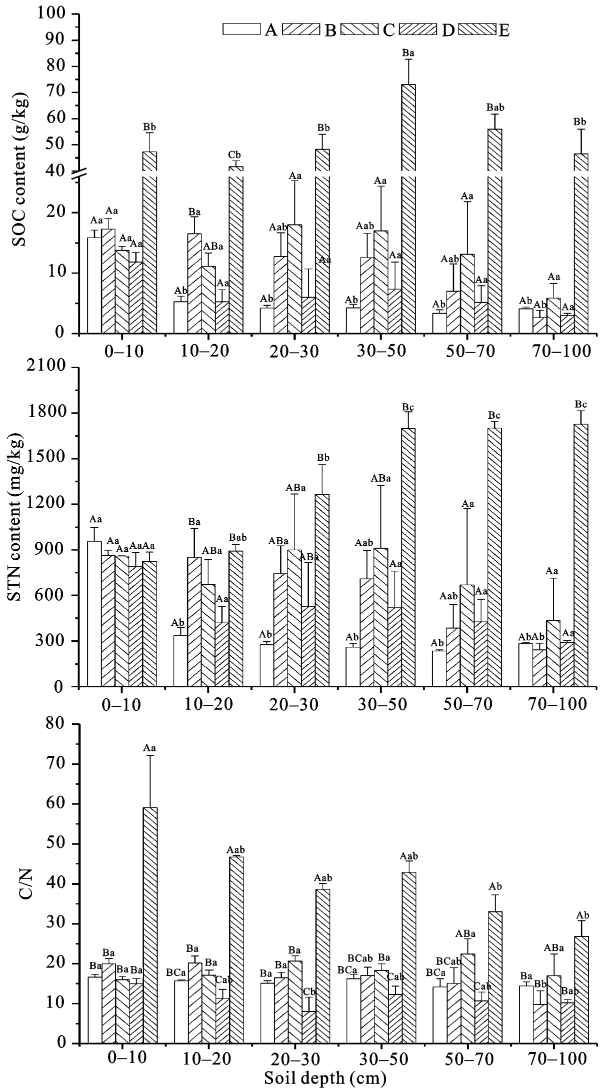

Fig. 2 Effects of wetland reclamation on soil organic carbon (SOC), soil total nitrogen (STN) and C/N ratio in different soil layers

The C/N in the five plots followed the order oil-polluted wetland > corn field > paddy field > reed marsh > forest land, with mean values of 41.18, 18.58, 16.41, 15.38 and 11.26, respectively. The soil C/N of oil-polluted wetland was significantly higher (P < 0.01), while that of forest stand was significantly lower (P < 0.05) than those in the reed marsh. Additionally, the soil C/N of paddy field did not differ significantly from that of reed marsh (P > 0.05). Soil depth and interaction between reclamation type and soil depth had significant effects on soil C/N (P < 0.05). In paddy field, there were positive correlations between C/N and EC (P < 0.05) (Table 3) and negative correlations between C/N and pH, Eh (P < 0.05).

3.3 Effects of wetland reclamation on DOC, NH_4^+-N and NO_3^--N of soils

Soil DOC, NH_4^+-N and NO_3^--N were significantly influenced by reclamation activities (P < 0.001) (Fig. 3 and Table 3). The soil DOC in the five plots followed the order oil-polluted wetland > paddy field > forest land > reed marsh > corn field, with mean values of 1314.71, 162.91, 121.54, 107.33 and 80.99 mg/kg, respectively. The soil DOC values in oil-polluted wetland were significantly higher than those in reed marsh (P < 0.001), while the soil DOC of paddy field, corn field and forest stand did not differ significantly from that of reed marsh (P > 0.05). There were positive

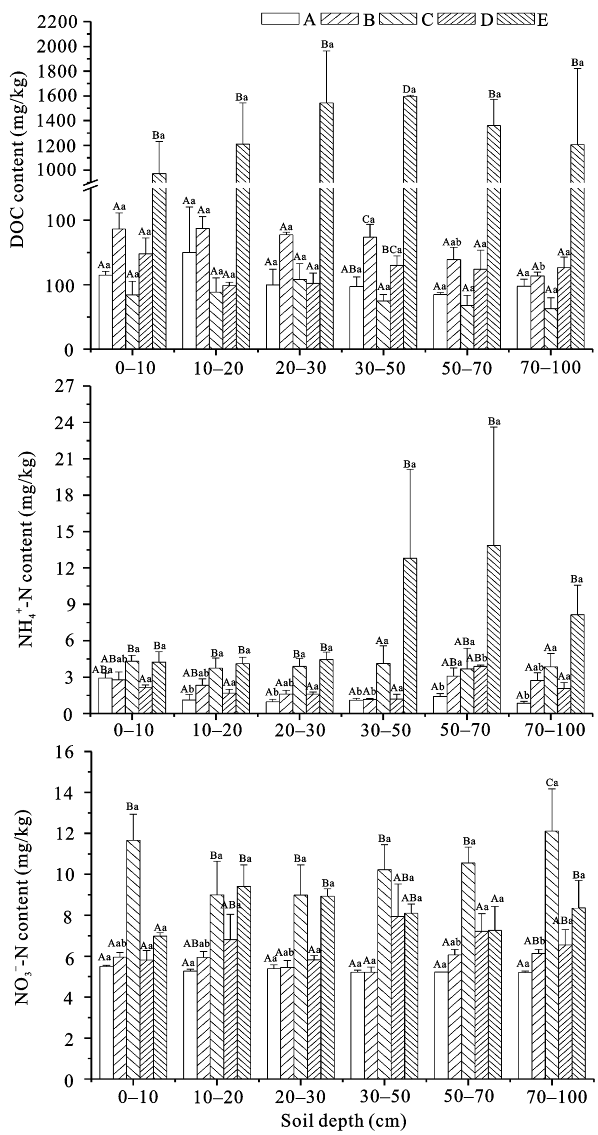

Fig. 3 Effects of wetland reclamation on dissolved organic carbon (DOC), ammonium nitrogen (NH_4^+-N) and nitrate nitrogen (NO_3^--N) in different soil layers

correlations between soil DOC and SOC, STN, C/N in paddy field (P < 0.01) (Table 2) and in forest land (P < 0.05). Soil EC had significant effects on soil DOC in paddy field (P < 0.01), corn field and forest land (P < 0.05).

The soil NH_4^+-N in the five plots followed the order oil-polluted wetland > corn field > paddy field > forest land > reed marsh, with mean values of 7.93 mg/kg, 3.93 mg/kg, 2.29 mg/kg, 2.10 mg/kg and 1.40 mg/kg, respectively. The soil NH_4^+-N values in oil-polluted wetland and corn field were significantly higher than those in reed marsh (P < 0.01), while NH_4^+-N values of paddy field and forest land did not differ from those of reed marsh (P > 0.05). In reed marsh and corn field, there were positive correlations between soil NH_4^+-N and SOC, STN (P < 0.01) and negative correlations between soil NH_4^+-N and pH, Eh (P < 0.01).

The soil NO_3^--N in the five plots followed the order corn field > oil-polluted wetland > forest land > paddy field > reed marsh, with mean values of 10.42 mg/kg, 8.17 mg/kg, 6.69 mg/kg, 5.79 mg/kg and 5.30 mg/kg, respectively. The soil NO_3^--N values in corn field, forest land and oil-polluted wetland were significantly higher than those in reed marsh (P < 0.01), while no significant difference in NO_3^--N was observed between paddy field and reed marsh (P > 0.05). Soil depth and interaction of reclamation type and soil depth had no significant effects on soil DOC, NH_4^+-N or NO_3^--N (P > 0.05) (Table 3).

4 Discussion

4.1 Effects of wetland reclamation on EC, pH and Eh of soils

All soils investigated in the present study were alkaline (pH 8.13–9.91). The soil pH values in paddy field, corn field, and forest land were slightly higher than those in reed marsh, and oil-polluted wetland soil had the highest pH values (9.82–10.08) in every layer. Generally, soil pH tends to increase temporarily during the early reclamation period because of the increase of HCO_3^- and Ca^{2+} and the reduction of Na^+ and Mg^{2+} (Chen et al., 2000). Oil exploitation will produce large amounts of alkaline waste drilling fluids, resulting in higher soil pH values, and these will increase with soil depth (Han et al., 2008). In each plot, the pH values of surface layers were slightly lower than those in deeper layers, which might be attributed to the release of organic acids from litter decomposition in the surface (Cui et al., 2000). Soil pH was negatively correlated with SOC and STN in reed marsh and corn field (P < 0.01) (Table 2). Moreover, soil pH greatly impacted the contents and spatial distributions of SOC and STN in wetland soils by influencing microbial activities (Bai et al., 2005). This is because microorganisms would be inhibited under alkaline conditions, and were most active in neutral conditions (Huang, 1994). Additionally, soil pH is sensitive to land management techniques such as fertilization (Kalbitz et al., 2013), and can be regulated to adapt to agricultural production.

The soil EC varied significantly among different reclamation types. Ganjegunte et al. (2009) observed an increase in EC of the reclaimed areas compared to the undisturbed site. However, Ding et al. (2001) reported that the EC of reclamation areas was lower than that of natural coastal saline soil. The soil EC in paddy field and reed marsh was significantly higher than that in corn field and forest land, possibly because the application of irrigated water from adjacent ditches led to an increase in and accumulation of soil salinity in surface soils (Bai et al., 2013). The increase of EC in oil-polluted wetland may be caused by the drilling mud, because drilling mud is composed of various stable colloidal suspensions that contain clays, weighting materials, chemical treatment agents, sewage and detritus, and usually have a high degree of mineralization, ions and anions (Wang et al., 2009).

Oil-polluted wetland soils have had the highest Eh value in every soil layer, with a mean value of 164.17 mV, which was higher than those in other plots, while the values in other plots ranged from 64.39 to 76.39 mV. There is a mutual restriction between oil content and moisture content in oil-polluted wetland. Some studies indicated that the strong hydrophobicity of oil caused high hydrophobicity of oil-polluted soil, which led to decreased water holding ability of soil, and lower moisture content (Jia et al., 2009). Moreover, this hydrophobicity had indirect effects on soil Eh. There were negative correlations between Eh and SOC, STN, NH_4^+-N in reed marsh and corn field (P < 0.05) (Table 2). This is likely because soil aeration will affect the mineralization of SOC (Yang et al., 2009) and nitrification of NH_4^+-N (Liu et al., 2015), leading to loss of soil carbon and nitrogen. However, this correlation was weaker in paddy field and forest land, possibly because of frequent water control and field management.

4.2 Effects of wetland reclamation on SOC, STN and C/N of soils

In this study, we found that the SOC contents (3.35–15.84 g/kg) in the reed marsh were lower than those in other studies (Song et al., 2011; Lou et al., 2015). This was associated with the reed harvest in our study area, which led to decreased input of plant carbon. Below the surface (0–10 cm), the SOC levels in paddy field and corn field were 2–4 times greater than those in reed marsh and forest land, and also higher than some natural wetlands in this area (Lin et al., 2013). This may be attributed to the mature cultivation management and fertilization (Bai et al., 2013). More plant residues are returned to agricultural land, leading to higher soil SOC contents. Some studies have indicated that the amount of plant residues returned to paddy soils is positively correlated with SOM accumulation (Zhou et al., 2009). In China, the return of plant residues to soil is a common practice in paddy lands (Cui et al., 2012). Additionally, application of chemical fertilizers and organic manure promote crop biomass production and bring more organic materials into agroecosystems directly (Roth et al., 2011). Generally, the effects of soil properties (EC, pH and Eh) on SOC mainly influence plant growth, microbial activity and physical leaching, but those correlations may be weakened by human disturbance. For example, the increases in soil pH that occur during coastal reclamation normally have a negative effect on soil organic matter. This is because high soil pH usually limits plant growth, leading to a decrease of plant carbon source into soils (Zhang et al., 2016). In our study, significant negative correlations between SOC and pH were only observed in reed marsh and corn field ($P < 0.01$). This difference can be attributed to the plant residues return management. Moreover, no significant correlations were found between SOC and soil properties in forest land and oil-polluted wetland.

The mean SOC content was 52.17 g/kg in oil-polluted wetland, which was significantly higher than in the other four types of soils. Some studies indicated that the oil content in severely polluted soil of the Liaohe River oilfield has reached 10 g/kg, which is much higher than the critical value of 500 mg/kg (He et al., 1999). Oil is composed of hundreds of materials, but its main ingredients are C_{15}–C_{36} alkanes, PAHs/olefin and benzene (Liu et al., 2007). Carbon is the main component of petroleum hydrocarbons; therefore, oil pollution could lead to increased SOC. However, this kind of increase does not mean that there are more available nutrients for plants because the components of oil could poison plants, destroy the growth environment and cause biological dysfunction, as well as severely damage agricultural and forestry production (Liu et al., 2007). As a result, SOC cannot be used to indicate soil fertility in areas of oil pollution.

It is generally acknowledged that SOC decreases exponentially with increasing depths (Bowman and Savory, 1992; Hobley et al., 2013). Similar results were observed in the present study, but the pattern was modified in different reclamation areas. In reed marsh and forest land, the SOC content of the surface layer (0–10 cm) was two to three times greater than that of deep layers. In paddy field, the SOC contents were higher at 0–20 cm, then decreased gradually with increasing depths. Moreover, the SOC content was higher at 20–50 cm in corn field. The vertical distribution of SOC was affected by many factors. First, vegetation provides soil with an important carbon source that directly affects the distribution of SOC (Gower et al., 2001; Zhang et al, 2008). Jobbágy and Jackson (2000) found that the roots of trees are distributed deeper and flourish more than herbs, and SOC decomposed more slowly in forest than in farmland and grassland, leading to a change in vertical distribution of SOC. Second, land management exerts a strong influence on SOC by changing the soil aeration condition. Paddy field submergence is considered a practice for the sequestration of soil carbon (Roth et al., 2011; Cui et al., 2012). Soils in drained agricultural lands have different redox conditions compared to waterlogged wetlands, which could lead to greater organic mineralization and decomposition under drying aerobic conditions (Sigua et al., 2009; Kögel-Knabner et al., 2010).

The content of STN showed a similar pattern to that of SOC, as did the correlations between STN and soil properties. Specifically, there were significantly negative correlations between STN and pH in reed marsh and corn field ($P < 0.01$), while significant correlations between STN and EC were found only in paddy field ($P < 0.01$), and no significant correlations were found between STN and soil properties in forest land and oil-polluted wetland ($P > 0.05$). In our study, the STN contents (390.90–1351.14 mg/kg) did not differ significantly relative to some natural wetlands (Mao et al., 2009; Mou et al., 2012). The STN values in paddy field and corn field were nearly double those in reed marsh and forest land, and decreased with depth (except for those in oil-polluted wetland). STN content is usually correlated with vegetation and land reclamation, similar to SOC (Bedard-Haughn et al., 2006). Moreover, factors such as soil management (Laudicina et al., 2009), fertilization (Powers, 2004) and reclamation duration (Cui et al., 2012) exert a strong influence on STN. Agricultural reclamation generally utilizes different crops, which leads to different species-specific effects on STN (Wang et al., 2010a). Some studies have indicated that plants adsorb about 50%–70% of N from the soil, although this ability varies among plants (Dewes, 1999). Long–term agricultural actions will increase the STN of topsoil and depth of the organic horizon (Cui et al., 2012). With the application of N-based fertilizer, STN increased in reclamation areas (Fu et al., 2014).

The STN content of oil-polluted soil varies with changes in regional and petroleum properties. Some studies have indicated that oil pollution causes decreased STN (Wang et al., 2009; 2010b). In the present study, STN was 1351.14 mg/kg in oil-polluted wetland, which was significantly higher than that in the other four reclamation types. This difference may have been because of the high content of nitrogen in petroleum. The amount of nitrogen in oil is usually less than a few percent, but the nitrogen content of crude oil was 0.06% in the nitrogen-rich Liaohe River oil field (Wang et al., 2010b). Additionally, STN values in oil-polluted wetland were higher in deeper layers. This might be attributed to accumulation of petroleum hydrocarbons, which are difficult to degrade (Soclo et al., 2000). Generally, the STN values were significantly correlated with SOC, but this correlation in oil-polluted wetland ($r^2 = 0.57$) was weaker than in the other four reclamation types ($r^2 > 0.9$) investigated in our study. Additionally, NO_3^--N levels in oil-polluted wetland were significantly higher than in other reclamation types (Fig. 3), indicating that the composition and morphology of STN was changed by oil pollution.

Soil C/N ratios were between 10 and 20, except in oil-polluted wetland (41.2), where the degree of humification of soil organic matter is higher and more conducive to microbial fermentation, so that organic nitrogen mineralizes easily (Thornton and Mcmanus, 1994; Springob and Kirchmann, 2003). Soil C/N can also indicate the soil organic matter source (oceanic or terrigenous) of an estuary (Milliman et al., 1984). In the present study, the mean values of soil C/N in all plots (11.26–41.18) were >8, indicating that the soil organic matter mainly comes from terrestrial in this study area. Generally, when the soil C/N reaches 12, it is beneficial to soil microbial degradation of petroleum contaminants (Płaza et al., 2005). The soil C/N value was 41.2 in oil-polluted wetlands, which was significantly higher than the ideal value. Oil pollution leads to increased soil carbon, and in the process of degradation of oil, soil microorganisms consume soil nutrients such as nitrogen and phosphorus (Atlas and Bartha, 1973), increasing the C/N. This imbalance of nutrients limits soil microbial reproduction and growth, thereby reducing the petroleum hydrocarbon degradation rate (Braddock et al., 1997).

4.3 Effects of wetland reclamation on DOC, NH_4^+-N and NO_3^--N of soils

Soil DOC mainly comes from plant litter, root secretions, hydrolysis of soil organic matter and microbial metabolites (Christ and David, 1996; Yang et al., 2006), which are all sensitive to soil properties. In the present study, a significant correlation was observed between DOC and EC in paddy field ($P < 0.01$), corn field and forest stand ($P < 0.05$), while no significant correlation was observed between DOC and pH, Eh in any plots. Mailapalli et al. (2010) found that wetland soil DOC content changed after being reclaimed because the drainage and farming changed the soil physical and chemical properties, and such factors will directly affect plant growth and microbial activities. Different irrigation patterns means different soil moisture, and it has a significant impact on soil DOC content (Han et al., 2010). In compared with the dry soil, alternate wetting and drying could improve the dissolution of SOC and scatter soil aggregates, thereby increasing the content of soil DOC (Xiao et al., 2015). The reclamation types and fertilization are also known to be important factors that influence soil DOC (Freeman et al., 2004; Tian et al., 2011). The content of DOC in paddy field and forest land was higher than in reed marsh and corn field, and there were positive correlations between DOC and SOC, STN, C/N in paddy field and forest land ($P < 0.05$) (Table 2). This may have been because of plant source and fertilization. Some studies have indicated that organic manure could increase soil DOC, but use of acid fertilizer will reduce the soil pH and microbial activity, leading to decreased soil DOC (Chantigny et al., 1999; Liu et al., 2006). In oil-polluted wetland, soil DOC increased with exogenous carbon input.

The soil NO_3^--N in corn field and forest stand were higher than in other plots, which could be caused by differences in soil aeration and nitrogen fertilization. In paddy field and reed marsh, soil was usually in an anaerobic state because of irrigation, which would inhibit nitrification, while soil under aerobic conditions was more conductive to the accumulation of NO_3^--N in corn field and forest (Haynes and Goh, 1978; Sun and Liu, 2008). In all plots, soil NO_3^--N decreased with depth (0–50 cm), then increased in deeper layers, which may have affected soil moisture and plant absorption capacity (Mou et al., 2010). NO_3^-–N is mineral nitrogen that can be absorbed by the plant, but is also easily leached (Wang et al., 2011). In the present study, the content of soil NH_4^+-N showed minor differences in agroecosystems, but in oil-polluted wetland, the content of soil NH_4^+-N in deep layers was significantly higher than in superficial layers and corresponding layers of other types. Because the transport and soil absorption ability of NH_4^+-N and NO_3^--N changed with soil properties, they were significantly influenced by oil pollution (Chang and He, 1998; Zhang et al., 2013).

5 Conclusions

In this study, we measured the content and distribution of SOC and STN in different land reclamation types. Overall, significant changes in SOC and STN with reclamation types were observed. The SOC and STN concentrations followed the same order, namely oil-polluted field > corn field > paddy field > forest land > reed marsh, and these differences were attributed to the different vegetation and land management among reclamation types. Moreover, the variations in soil properties exerted an indirect influence on SOC and STN. Higher SOC concentrations in paddy field and corn field highlighted the positive effects of mature land management on the accumulation of SOC. Interaction of reclamation type and soil depth had significant effects on SOC and STN, while soil depth had significant effects on SOC, but not STN. DOC, NH_4^+-N and NO_3^--N also changed with intensity of human activities. SOC and STN were significantly correlated, but this correlation was weaker in oil-polluted wetland, and the measured values in oil-polluted wetland were higher than those in other reclamation types. However, these types of increases were attributed to petroleum hydrocarbon pollution, and the contents of SOC and STN cannot be indicators of soil fertility and quality.

References

Andrews S S, Karlen D L, Cambardella C A, 2004. The soil management assessment framework: a quantitative soil quality evaluation method. *Soil Science Society of America Journal*, 68(6): 1945–1962.

Atlas R M, Bartha R, 1973. Stimulated biodegradation of oil slicks using oleophilic fertilizers. *Environmental Science & Technology*, 7(6): 538–541. doi: 10.1021/es60078a005

Bai J, Xiao R, Zhang K et al., 2013. Soil organic carbon as affected by land use in young and old reclaimed regions of a coastal estuary wetland, China. *Soil Use and Management*, 29(1): 57–64. doi: 10.1111/sum.12021

Bai J H, Ouyang H, Deng W et al., 2005. Spatial distribution characteristics of organic matter and total nitrogen of marsh soils in river marginal wetlands. *Geoderma*, 124(1–2): 181–192. doi: 10.1016/j.geoderma.2004.04.012

Bedard-Haughn A, Matson A L, Pennock D J, 2006. Land use effects on gross nitrogen mineralization, nitrification, and N_2O emissions in ephemeral wetlands. *Soil Biology and Biochemistry*, 38(12): 3398–3406. doi: 10.1016/j.soilbio.2006.05.010

Benoit L K, Askins R A, 2002. Relationship between habitat area and the distribution of tidal marsh birds. *The Wilson Bulletin*, 114(3): 314–323. doi: 10.1676/0043-5643(2002)114[0314:RBHAAT]2.0.CO;2

Boesch D F, 2002. Challenges and opportunities for science in reducing nutrient over-enrichment of coastal ecosystems. *Estuaries*, 25(4): 886–900. doi: 10.1007/BF02804914

Bowman R A, Savory D J, 1992. Phosphorus distribution in calcareous soil profiles of the Central Plains. *Soil Science Society of America Journal*, 56(2): 423–426. doi: 10.2136/sssaj1992.03615995005600020013x

Braddock J F, Ruth M L, Catterall P H et al., 1997. Enhancement and inhibition of microbial activity in hydrocarbon-contaminated arctic soils: implications for nutrient-amended bioremediation. *Environmental Science & Technology*, 31(7): 2078–2084. doi:

10.1021/es960904d

Chang Zhizhou, He Jiajun, 1998. Effects of crude oil contamination on the rates of nitrogen mineralization and nitrification in soil. *Rural Eco-Environment*, 14(1): 39–42. (in Chinese)

Chantigny M H, Angers D A, Prévost D et al., 1999. Dynamics of soluble organic C and C mineralization in cultivated soils with varying N fertilization. *Soil Biology and Biochemistry*, 31(4): 543–550. doi: 10.1016/S0038-0717(98)00139-4

Chen Wei, Chen Bangben, Shen Qirong, 2000. Studies on the changes of pH value and alkalization of heavily saline soil in seabeach during its desalting process. *Acta Pedologica Sinica*, 37(4): 521–528. (in Chinese)

Chen Yanzhen, Fang Guozhi, Ni Jin et al., 2010. Research on century's changes of coastlines of Liaohe Estuary. *Journal of Marine Sciences*, 28(2): 14–21. (in Chinese)

Christ M J, David M B, 1996. Dynamics of extractable organic carbon in Spodosol forest floors. *Soil Biology and Biochemistry*, 28(9): 1171–1179. doi: 10.1016/0038-0717(96)00128-9

Costanza R, d'Arge R, De Groot R et al., 1998. The value of the world's ecosystem services and natural capital. *Ecological Economics*, 25(1): 3–15. doi: 10.1016/S0921-8009(98)00020-2

Cui Guofa, Cai Tijiu, Yang Wenhua, 2000. Soil acidity of *Larix gmelini* plantation. *Journal of Beijing Forestry University*, 22(3): 33–36. (in Chinese)

Cui J, Liu C, Li Z L et al., 2012. Long-term changes in topsoil chemical properties under centuries of cultivation after reclamation of coastal wetlands in the Yangtze Estuary, China. *Soil and Tillage Research*, 123: 50–60. doi: 10.1016/j.still.2012.03.009

Deng X F, Zhan Y, Wang F et al., 2016. Soil organic carbon of an intensively reclaimed region in China: current status and carbon sequestration potential. *Science of the Total Environment*, 565: 539–546. doi: 10.1016/j.scitotenv.2016.05.042

Dewes T, 1999. Ammonia emissions during the initial phase of microbial degradation of solid and liquid cattle manure. *Bioresource Technology*, 70(3): 245–248. doi: 10.1016/S0960-8524(99)00046-2

Ding Nengfei, Li Renan, Dong Bingrong et al., 2001. Long-term observation and study on salinity and nutrients of coastal saline soils (sandy soil). *Chinese Journal of Soil Science*, 32(2): 57–59. (in Chinese)

Don A, Schumacher J, Freibauer A, 2011. Impact of tropical land-use change on soil organic carbon stocks–a meta-analysis. *Global Change Biology*, 17(4): 1658–1670. doi: 10.1111/j.1365-2486.2010.02336.x

Donato D C, Kauffman J B, Murdiyarso D et al., 2011. Mangroves among the most carbon-rich forests in the tropics. *Nature Geoscience*, 4(5): 293–297. doi: 10.1038/ngeo1123

Duke N C, Meynecke J O, Dittmann S et al., 2007. A world without mangroves? *Science*, 317(5834): 41–42. doi: 10.1126/science.317.5834.41b

Freeman C, Fenner N, Ostle N J et al., 2004. Export of dissolved organic carbon from peatlands under elevated carbon dioxide levels. *Nature*, 430(6996): 195–198. doi: 10.1038/nature02707

Fu Q L, Ding N F, Liu C et al., 2014. Soil development under different cropping systems in a reclaimed coastal soil chronosequence. *Geoderma*, 230–231: 50–57. doi: 10.1016/j.geoderma.2014.03.026

Ganjegunte G K, Wick A F, Stahl P D et al., 2009. Accumulation and composition of total organic carbon in reclaimed coal mine lands. *Land Degradation and Development*, 20(2): 156–175. doi: 10.1002/ldr.889

Gower S T, Krankina O, Olson R J et al., 2001. Net primary production and carbon allocation patterns of boreal forest ecosystems. *Ecological Applications*, 11(5): 1395–1411. doi: 10.2307/3060928

Han Dechang, Jiang Siwei, Wen Dazhong, 2008. Vertical distribution of TPHs in petroleum-contaminated soil of Liaohe oilfield. *Chinese Journal of Soil Science*, 39(4): 932–934. (in Chinese)

Han Lin, Zhang Yulong, Jin Shuo et al., 2010. Effect of different irrigation patterns on soil dissolved organic carbon and microbial biomass carbon in protected field. *Scientia Agricultura Sinica*, 43(8): 1625–1633. (in Chinese)

Haynes R J, Goh K M, 1978. Ammonium and nitrate nutrition of plants. *Biological Reviews*, 53(4): 465–510. doi: 10.1111/j.1469-185X.1978.tb00862.x

He Liangju, Wei Dezhou, Zhang Weiqing, 1999. Research of microbial treatment of petroleum contaminated soil. *Advances in Environmental Science*, 7(3): 110–115. (in Chinese)

Hobley E, Willgoose G R, Frisia S et al., 2013. Environmental and site factors controlling the vertical distribution and radiocarbon ages of organic carbon in a sandy soil. *Biology and Fertility of Soils*, 49(8): 1015–1026. doi: 10.1007/s00374-013-0800-z

Huang Ruinong, 1994. *Environment Pedology*. Beijing: Higher Education Press, 145–146. (in Chinese)

Huang Guilin, Zhang Jianjun, Li Yuxiang, 2000. Analysis of classification and environment situation of wetlands in Liaohe River Delta. *Forest Resources Management*, (4): 51–56. (in Chinese)

Huang J Y, Song C C, 2010. Effects of land use on soil water soluble organic C and microbial biomass C concentrations in the Sanjiang Plain in Northeast China. *Acta Agriculturae Scandinavica, Section B–Soil & Plant Science*, 60(2): 182–188. doi: 10.1080/09064710802680387

Iost S, Landgraf D, Makeschin F, 2007. Chemical soil properties of reclaimed marsh soil from Zhejiang Province P.R. China. *Geoderma*, 142(3–4): 245–250. doi: 10.1016/j.geoderma.2007.08.001

Jia Jianli, Liu Ying, Li Guanghe et al., 2009. Contamination characteristics and its relationship with physicochemical properties of oil polluted soils in oilfields of China. *CIESC Journal*, 60(3): 726–732. (in Chinese)

Jin Wenhui, Yang Jinsong, Wang Xiangping, 2013. Spatial distribution of organic carbon in coastal saline soil and its correlation with reclamation age. *Transactions of the Chinese Society of Agricultural Engineering*, 29(5): 89–94. (in Chinese)

Jobbágy E G, Jackson R B, 2000. The vertical distribution of soil organic carbon and its relation to climate and vegetation. *Ecological Applications*, 10(2): 423–436. doi: 10.2307/2641104

Kalbitz K, Kaiser K, Fiedler S et al., 2013. The carbon count of 2000 years of rice cultivation. *Global Change Biology*, 19(4): 1107–1113. doi: 10.1111/gcb.12080

Kögel-Knabner I, Amelung W, Cao Z H et al., 2010. Biogeochemistry of paddy soils. *Geoderma*, 157(1–2): 1–14. doi: 10.1016/j.geoderma.2010.03.009

Lal R, 2004. Soil carbon sequestration impacts on global climate change and food security. *Science*, 304(5677): 1623–1627. doi: 10.1126/science.1097396

Laudicina V A, Hurtado M D, Badalucco L et al., 2009. Soil chemical and biochemical properties of a salt–marsh alluvial Spanish area after long-term reclamation. *Biology and Fertility of Soils*, 45(7): 691–700. doi: 10.1007/s00374-009-0380-0

Leng Yanhui, Guo Shuhai, Nie Yuanbin et al., 2006. Effects of petroleum exploitation on ecological environment of reed fields in Liaohe River Delta. *Journal of Agro-Environment Science*, 25(2): 432–435. (in Chinese)

Li X Z, Sun Y G, Mander Ü et al., 2013. Effects of land use intensity on soil nutrient distribution after reclamation in an estuary landscape. *Landscape Ecology*, 28(4): 699–707. doi: 10.1007/s10980-012-9796-2

Lin T, Ye S Y, Ma C L et al., 2013. Sources and preservation of organic matter in soils of the wetlands in the Liaohe (Liao River) Delta, North China. *Marine Pollution Bulletin*, 2013, 71(1–2): 276–285. doi: 10.1016/j.marpolbul.2013.01.036

Liu Min, Yu Wantai, Jiang Zishao et al., 2006. A research review on soil active organic carbon. *Chinese Journal of Ecology*, 25(11): 1412–1417. (in Chinese)

Liu Ruoxuan, Zhang Limei, Bai Ren et al., 2015. Response of nitrification and nitrifier to change in soil moisture content under simulated conditions. *Acta Pedologica Sinica*, 52(2): 415–422. (in Chinese)

Liu Wuxing, Luo Yongming, Teng Ying et al., 2007. A survey of petroleum contamination in several Chinese oilfield soils. *Soils*, 39(2): 247–251. (in Chinese)

Liu Z P, Shao M A, Wang Y Q, 2011. Effect of environmental factors on regional soil organic carbon stocks across the Loess Plateau region, China. *Agriculture*, *Ecosystems & Environment*, 142(3–4): 184–194. doi: 10.1016/j.agee.2011.05.002

Lu Rukun, 1999. *Soil Agrochemistry Analysis Method.* Beijing: China Agriculture Science Press, 106–150. (in Chinese)

Luo Xianxiang, Jia Hongli, Yang Jianqiang et al., 2015. A comparison of soil organic carbon pools in two typical estuary reed wetlands in northern China. *Periodical of Ocean University of China*, 45(3): 99–106. (in Chinese)

Ma Z J, Melville D S, Liu J G et al., 2014. Rethinking China's new great wall. *Science*, 346(6212): 912–914. doi: 10.1126/science.1257258

Mailapalli D R, Wallender W W, Burger M et al., 2010. Effects of field length and management practices on dissolved organic carbon export in furrow irrigation. *Agricultural Water Management*, 98(1): 29–37. doi: 10.1016/j.agwat.2010.07.009

Mao Zhigang, Wang Guoxiang, Liu Jin'e et al., 2009. Influence of salt marsh vegetation on spatial distribution of soil carbon and nitrogen in Yancheng coastal wetland. *Chinese Journal of Applied Ecology*, 20(2): 293–297. (in Chinese)

Milliman J D, Xie Q C, Yang Z S, 1984. Transfer of particulate organic carbon and nitrogen from the Yangtze River to the ocean. *American Journal of Science*, 284(7): 824–834. doi: 10.2475/ajs.284.7.824

Mou Xiaojie, Sun Zhigao, Wang Lingling, 2010. Physical transport of nitrate nitrogen and ammonium nitrogen in mire wetland soils–a review. *Transactions of Oceanology and Limnology*, (1): 146–152. (in Chinese)

Mou Xiaojie, Sun Zhigao, Liu Xingtu, 2012. Spatial distribution patterns of carbon and nitrogen in the tidal marsh soil of the Yellow River Estuary. *Scientia Geographica Sinica*, 32(12): 1521–1529. (in Chinese)

Płaza G, Nałęcz-Jawecki G, Ulfig K et al., 2005. The application of bioassays as indicators of petroleum-contaminated soil remediation. *Chemosphere*, 59(2): 289–296. doi: 10.1016/j.chemosphere.2004.11.049

Poster W M, Emanuel W R, Zinke P J et al., 1982. Soil carbon pools and world life zones. *Nature*, 298(5870): 156–159. doi: 10.1038/298156a0

Powers J S, 2004. Changes in soil carbon and nitrogen after contrasting land-use transitions in northeastern Costa Rica. *Ecosystems*, 7(2): 134–146. doi: 10.1007/s10021-003-0123-2

Roth P J, Lehndorff E, Cao Z H et al., 2011. Accumulation of nitrogen and microbial residues during 2000 years of rice paddy and non-paddy soil development in the Yangtze River Delta, China. *Global Change Biology*, 17(11): 3405–3417. doi: 10.1111/j.1365-2486.2011.02500.x

Santín C, Otero X L, Fernández S et al., 2007. Variations of organic carbon stock in reclaimed estuarine soils (Villaviciosa estuary, NW Spain). *Science of the Total Environment*, 378(1–2): 138–142. doi: 10.1016/j.scitotenv.2007.01.033

Schindler D W, 2006. Recent advances in the understanding and management of Eutrophication. *Limnology and Oceanography*, 51(1): 356–363. doi: 10.4319/lo.2006.51.1_part_2.0356

Shao Hua, Shi Qinghua, Zhao Xiaomin et al., 2010. Impacts of various agricultural land use patterns on the content of organic carbon in soil in Jiangxi Province. *China Land Science*, 24(10): 13–17. (in Chinese)

Sigua G C, Coleman S W, Albano J, 2009. Beef cattle pasture to wetland reconversion: impact on soil organic carbon and phosphorus dynamics. *Ecological Engineering*, 35(8): 1231–1236. doi: 10.1016/j.ecoleng.2009.05.004

Soclo H H, Garrigues P, Ewald M, 2000. Origin of polycyclic aromatic hydrocarbons (PAHs) in coastal marine sediments: case studies in Cotonou (Benin) and Aquitaine (France) Areas. *Marine Pollution Bulletin*, 40(5): 387–396. doi: 10.1016/S0025-326X(99)00200-3

Song Xiaolin, Lü Xianguo, Zhang Zhongsheng et al., 2011. Study on nutrient and salinity in soil covered with different vegetations in Shuangtaizi estuarine wetlands. *Environmental Science*, 32(9): 2632–2638. (in Chinese)

Springob G, Kirchmann H, 2003. Bulk soil C to N ratio as a simple measure of net N mineralization from stabilized soil organic matter in sandy arable soils. *Soil Biology and Biochemistry*, 35(4): 629–632. doi: 10.1016/S0038-0717(03)00052-X

Sun Yongguang, Li Xiuzhen, Mander Ülo et al., 2011. Effect of reclamation time and land use on soil properties in Changjiang River Estuary, China. *Chinese Geographical Science*, 21(4): 403–416. doi: 10.1007/s11769-011-0482-0

Sun Zhigao, Liu Jingshuang, 2008. Nitrification-denitrification and its affecting factors in wetland soil–a review. *Chinese Journal of Soil Science*, 39(6): 1462–1467. (in Chinese)

Suo Anning, Zhao Dongzhi, Zhang Fengshou, 2010. Carbon storage and fixation by wetland vegetation at the estuaries in northern China: a case of Panjin area, Liaohe Delta. *Journal of Marine Sciences*, 28(3): 67–71. (in Chinese)

Syakila A, Kroeze C, 2011. The global nitrous oxide budget revisited. *Greenhouse Gas Measurement and Management*, 1(1): 17–26. doi: 10.3763/ghgmm.2010.0007

Thornton S F, McManus J, 1994. Application of organic carbon and nitrogen stable isotope and C/N ratios as source indicators of organic matter provenance in estuarine systems: evidence from the Tay Estuary, Scotland. *Estuarine*, *Coastal and Shelf Science*, 38(3): 219–233. doi: 10.1006/ecss.1994.1015

Tian Jing, Guo Jingheng, Chen Haiqing et al., 2011. Effect of land use on composition of soil dissolved organic carbon. *Acta Pedologica Sinica*, 48(2): 338–346. (in Chinese)

Utset A, Borroto M, 2001. A modeling-GIS approach for assessing irrigation effects on soil salinisation under global warming conditions. *Agricultural Water Management*, 50(1): 53–63. doi: 10.1016/S0378-3774(01)00090-7

Wang Chuanyuan, Yang Cuiyun, Sun Zhigao et al., 2010b. Contamination characteristics and its relationship with physico-chemical properties of oil polluted soils in the Yellow River Delta swamp. *Journal of Soil and Water Conservation*, 24(2): 214–217. (in Chinese)

Wang Lingling, Sun Zhigao, Mou Xiaojie et al., 2011. Spatial distribution of nitrogen in different wetlands' soils in intertidal flat of Yellow River Estuary. *Chinese Journal of Soil Science*, 42(6): 1439–1445. (in Chinese)

Wang Q K, Wang S L, Xu G B et al., 2010a. Conversion of secondary broadleaved forest into Chinese fir plantation alters litter production and potential nutrient returns. *Plant Ecology*, 209(2): 269–278. doi: 10.1007/s11258-010-9719-8

Wang Xiaoyu, Feng Jiang, Wang Jing, 2009. Petroleum hydrocarbon contamination and impact on soil characteristics from oilfield Momoge Wetland. *Environmental Science*, 30(8): 2394–2401.(in Chinese)

Wiesmeier M, Munro S, Barthold F et al., 2015. Carbon storage capacity of semi-arid grassland soils and sequestration potentials in northern China. *Global Change Biology*, 21(10): 3836–3845. doi: 10.1111/gcb.12957

Xiao Ye, Huang Zhigang, Wu Haitao et al., 2015. Compositions and contents of active organic carbon in different wetland soils in Sanjiang Plain, Northeast China. *Acta Ecologica Sinica*, 35(23): 7625–7633. (in Chinese)

Yang Gairen, Tong Chengli, Xiao Heai et al., 2009. Effects of water content on redox potential and carbon mineralization of wetland sediments. *Environmental Science*, 30(8): 2381–2386. (in Chinese)

Yang Jisong, Liu Jingshuang, Yu Junbao et al., 2006. Dynamics of dissolved organic carbon (DOC) leaching in meadow marsh soil and related affecting factors. *Chinese Journal of Applied Ecology*, 17(1): 113–117. (in Chinese)

Yin A J, Zhang M, Gao C et al., 2016. Salinity evolution of coastal soils following reclamation and intensive usage, eastern China. *Environmental Earth Sciences*, 75(18): 1281. doi: 10.1007/s12665-016-6095-2

Zhang Changkuan, Chen Jun, Lin Kang et al., 2011. Spatial layout of reclamation of coastal tidal flats in Jiangsu Province. *Journal of Hohai University* (*Natural Sciences*), 39(2): 206–212. (in Chinese)

Zhang H, Wu P B, Yin A J et al., 2016. Organic carbon and total nitrogen dynamics of reclaimed soils following intensive agricultural use in eastern China. *Agriculture*, *Ecosystems & Environment*, 235: 193–203. doi: 10.1016/j.agee.2016.10.017

Zhang W J, Xiao H A, Tong C L et al., 2008. Estimating organic carbon storage in temperate wetland profiles in Northeast China. *Geoderma*, 146(1–2): 311–316. doi: 10.1016/j. geoderma.2008.06.006

Zhang Xiaoyang, Li Kairong, Zhang Linjun, 2013. Effect of petroleum contamination on physical and chemical properties of soils in oilfield of northern Shaanxi. *Research of Soil and Water Conservation*, 20(3): 32–38. (in Chinese)

Zhang Xuliang, Zhang Zhaohui, Gu Dongqi et al., 2009. Research on evolution of coastal wetlands in Liaohe River Delta. *Ecology and Environmental Sciences*, 18(3): 1002–1009. (in Chinese)

Zhang Y, Xu H L, Chen H et al., 2014. Diversity of wetland plants used traditionally in China: a literature review. *Journal of Ethnobiology and Ethnomedicine*, 10: 72. doi: 10.1186/1746-4269-10-72

Zhou Ping, Pan Genxing, Li Lianqing et al., 2009. SOC enhancement in major types of paddy soils in a long-term agro-ecosystem experiment in South China. V. relationship between carbon input and soil carbon sequestration. *Scientia Agricultura Sinica*, 42(12): 4260–4268. (in Chinese)