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Seawall construction alters soil carbon and nitrogen dynamics and soil microbial biomass in an invasive *Spartina alterniflora* salt marsh in eastern China

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ABSTRACT

Seawalls have increasingly been constructed to control the invasion of an exotic perennial grass, *Spartina alterniflora*, in coastal wetlands of eastern China. We investigated soil organic carbon (C) and nitrogen (N), available and microbial C and N, soil microbial community composition and biomass in a seawall-reclaimed *S. alterniflora* salt marsh compared with an adjacent natural *S. alterniflora* salt marsh. Seawall reclamation in *S. alterniflora* salt marsh significantly decreased soil salinity, moisture, litter and root biomass, and strongly decreased soil total organic C by 57% and total organic N by 59%, and also lowered soil available C and N in *S. alterniflora* salt marsh. Seawall reclamation significantly decreased soil microbial biomass C and the quantities of the total phospholipid fatty acids (PLFAs), and bacterial, fungal, gram-negative bacterial, gram-positive bacterial, saturated straight-chain, and monounsaturated PLFAs in deeper soil layers (10–30 cm). Our results suggested that seawall construction could greatly decrease soil C and N accumulation of *S. alterniflora* salt marsh by decreasing *S. alterniflora* residuals input into the soil and lowering soil salinity and moisture, which further decreased soil microbial biomass by lowering soil available C and N in *S. alterniflora* salt marsh. Changes in soil microbial community composition and biomass could in turn affect soil C and N accumulation in a seawall-reclaimed *S. alterniflora* salt marsh in eastern China.

1. Introduction

Coastal wetlands (e.g., salt marshes) have been recognized as important components of ‘blue carbon’ (C) sinks and play vital roles in alleviating global climate change (Laffoley and Grimsditch, 2009; Bu et al., 2015). It has been suggested that C burial rate in coastal salt marshes is approximately 55 times higher than that in tropical rainforests (Macreadie et al., 2013), and their global C sequestration (preliminary estimation up to 87.2 Tg C yr^{-1}) seems to surpass that in tropical rainforests (53 Tg C yr^{-1}) (McLeod et al., 2011; Macreadie et al., 2013). Although coastal wetlands provide various ecological services, such as biodiversity preservation, C sinks, retention of pollutants, nutrient filtration, shoreline erosion control, and flood peak reduction protection (Santín et al., 2009; Borsje et al., 2011; Bu et al., 2015), an increasing number of coastal wetlands have been degraded or lost by human activity in many countries (Santín et al., 2009; Ma et al., 2014). It is estimated that approximately 50% of salt marshes worldwide have been degraded or lost primarily due to land reclamation (Barbier et al., 2011). For instance, China’s coastal wetlands have been enclosed by thousands of kilometers of seawalls that cover 60% of the total length of coastline of mainland China (Ma et al., 2014). Additionally, half of coastal wetlands were lost until 2000 due to reclamation and coastal engineering (Bi et al., 2014), leading to a

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dramatic decline in biodiversity, ecological service function and changes in biogeochemical cycles (Ma et al., 2014).

In the reclamation process, coastal wetlands are usually embanked (i.e., seawall construction) to use as agricultural land or for urban purposes (Santín et al., 2009). The destruction or conversion of coastal wetlands can strongly modify the redox environment and hydrology conditions of coastal wetlands due to blocking tidal inundation after reclamation (Dick and Osunkoya, 2000). The changes in soil physico-chemical properties (Wang et al., 2014) and hydrology conditions after seawall reclamation potentially affect soil C and nitrogen (N) sinks in coastal wetlands (Bu et al., 2015). Previous studies have reported that coastal embankment significantly drives the succession of salt marshes and soil (Bozek and Burdick, 2005; Bi et al., 2014), promotes soil C pool loss and releases more CO_2 into the atmosphere from a coastal *Phragmites australis* salt marsh (Bu et al., 2015). The conversion of coastal wetlands to agricultural land has been reported to significantly decrease soil C and N accumulation during the early period of reclamation (i.e., initial 16 years), but then rapidly recover within 30 years and thereafter slow accumulated soil C and N following cultivation time until 500 years (Cui et al., 2012). However, Wang et al. (2014) have found that although organic and chemical fertilizers have been applied over a long-term period of time, 60-year agricultural reclamation significantly decreases soil C and N accumulation in the surface layer (0–30 cm), possibly because soil organic matter (SOM) output exceeds input after long-term agricultural reclamation. These contradictory findings may result from differences in reclamation history, hydrologic conditions and land use patterns after reclamation (Wang et al., 2014).

It is well known that soil microorganisms play key roles in mediating soil C and N pools, and ecosystem C and N cycling (Zeller et al., 2008; Hargreaves and Hofmockel, 2014). Coastal wetland reclamation has been shown to significantly modify soil physico-chemical properties (Li et al., 2014; Wang et al., 2014), and alter soil C and N levels (Cui et al., 2012; Bu et al., 2015). The changes in soil physico-chemical properties and available substrate induced by coastal wetland reclamation could greatly affect soil microbial community structure (Liu et al., 2013). Changes in the soil microbial community structure (e.g., the ratio of fungal:bacterial biomass) would in turn further impact soil C and N sequestration and turnover (Bailey et al., 2002). Thus, a focus on the dynamic changes in soil C and N, and the soil microbial community structure following the reclamation of coastal wetlands has important significance for predicting the effects of global climate change, especially because more intensive reclamation is expected in the foreseeable future (Ma et al., 2014; Bu et al., 2015).

Spartina alterniflora, a perennial grass that is native to North America, was introduced into China in 1979 for coastal erosion control and sediment stabilization (Yang et al., 2016a). *S. alterniflora* has many biological traits (e.g., fast growth, high leaf area index and net photosynthetic rate, well-developed root system, great clonal propagation, and high tolerance to salt and waterlogging) (Liao et al., 2007; Li et al., 2009; Yang et al., 2016a), making it a good 'ecosystem engineer' (Zhang et al., 2005; Li et al., 2009). For this reason, *S. alterniflora* has been widely introduced to the coastal wetlands of eastern China, and it has shown rampant expansion over the past 30 years, from Tianjin in the north to Beihai in the south, and covers a total area of more than 112,000 ha (An et al., 2007). Previous studies have showed that *S. alterniflora* invasion can greatly accelerate soil organic C and N sequestration by increasing *S. alterniflora* residuals input and lowering SOM decomposition in comparison with bare flat and native *Scirpus mariqueter*, *Suaeda salsa* and *P. australis* communities (Liao et al., 2007; Cheng et al., 2008; Yang et al., 2016a). Although *S. alterniflora* salt marsh strongly affects C and N sinks (Liao et al., 2007), many studies have reported that *S. alterniflora* invasion threatens native ecosystems, reduces biodiversity, alters the structure of trophic functional groups of nematode and macrobenthonic invertebrate communities, and changes the habitats of shorebirds (Li et al., 2009). The Chinese government has explored various methods, including cutting, burning and reaping young ramets, to eliminate *S. alterniflora*, and embankment has been proven to be an effective method to prevent *S. alterniflora* from expanding (An et al., 2007). Currently, coastal embankment has increasingly been used to control *S. alterniflora* invasion in coastal wetlands of eastern China (An et al., 2007).

A previous study have reported that coastal embankment significantly promotes C loss from previously sequestered soil C pool and accelerates soil respiration in a *P. australis* salt marsh (Bu et al., 2015). However, little is known about the effects of seawall construction in invasive *S. alterniflora* salt marsh on soil C and N dynamics, as well as the soil microbial community–the primary mediator of soil C and N cycling. We hypothesized that seawall construction in invasive *S. alterniflora* salt marsh would significantly decrease soil C and N accumulation by lowering plant residuals entering the soil and changing the soil physico-chemical properties, which further decrease soil microbial biomass by lowering soil available substrates. To test this hypothesis, we examined the concentrations of soil organic carbon (SOC) and organic nitrogen (SON), water-soluble organic carbon (WSOC), nitrate nitrogen (NO_3-N), ammonium nitrogen (NH_4-N), nitrite nitrogen (NO_2-N), microbial biomass carbon (MBC) and nitrogen (MBN), and examined the soil microbial community composition and biomass through phospholipid fatty acids (PLFAs) analysis in a seawall-reclaimed *S. alterniflora* salt marsh by comparing the results with an adjacent natural *S. alterniflora* salt marsh. We also identified the effects of biotic (e.g., litter and root biomass) and abiotic (e.g., soil moisture, bulk density, pH and salinity) factors on soil C and N dynamics, and soil microbial community composition and biomass in natural and seawall-reclaimed *S. alterniflora* salt marshes in a coastal wetland of eastern China.

2. Materials and methods

2.1. Study area

The study was conducted in the third core region of the Dafeng Milu National Nature Reserve (DMNNR), Jiangsu Province, China (32°59′–33°03′ N, and 120°47′–120°53′ E; Fig. 1). The DMNNR is located next to the Yellow Sea (Fig. 1). The climate in this area belongs to the typical monsoon climate transition belt from a subtropical zone to a warm temperate zone with a mean annual temperature of 14.1 °C; the mean annual precipitation exceeds 1000 mm (Liu et al., 2011). The DMNNR was established in 1986 and contains the world's largest population of wild *Elaphurus davidianus*, which have returned to nature (Liu et al., 2011). The DMNNR was recorded as part of the "Man and Biosphere Protection Network" in 1995, is a designated Ramsar Site and was listed in the directory of Wetlands of International Importance in 2002 (Wang and Wall, 2010).

S. alterniflora was intentionally introduced into the Dongtai River estuary of the intertidal zone, Dongtai city, Jiangsu province, in 1988, and it rapidly expanded northward over a distance of 100 km and occupied the coastal beach in Dafeng city, Jiangsu Province (Ding, 2009). At present, the dominant plant community of the third core area of the DMNNR is an invasive *S. alterniflora* community that occupies an area of 14 km^2, i.e., approximately 70% of the whole third core area (Ji et al., 2011). Native C_3 *S. salsa* and *P. australis* communities occupy approximately 30% of the third core area in the DMNNR. For the purposes of controlling *S. alterniflora* expansion and recovering the native salt marshes, a seawall was constructed in the third core area of the DMNNR in 2011 (Fig. 1);