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Vegetation alters the effects of salinity on greenhouse gas emissions and carbon sequestration in a newly created wetland

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ABSTRACT

Wetland creation or restoration in degraded areas has become a new type of disturbance worldwide. Coastal wetlands serve a vital role in global carbon cycles; thus, it is important to understand the impacts of wetland creation on carbon storage functions. Carbon emissions and accumulation in wetlands are reported to be highly site-specific depending on factors such as salinity, plant type and productivity, and water table. This study investigated the effects of different salinities (<2‰, ∼5‰ and >10‰) on greenhouse gas emissions and carbon sequestration of created wetlands in the Yangtze River estuary. CH_4 emissions significantly declined with increasing salinity, likely because of the higher sediment sulfate content at higher salinities. CO_2 emissions were highest at intermediate salinities (∼5‰). In unvegetated sites, the absolute CO_2 emission equivalent was 0.178 kg m^{-2} y^{-1} in the <2‰ salinity treatment, which was 8.09 times higher than the >10‰ salinity treatment. In vegetated sites, the <2‰ salinity treatment had the highest annual net flux of carbon. Thus, despite the high carbon emission of low salinity wetland, enhanced plant productivity resulted in a high carbon absorption rate. Overall, these results demonstrate that the presence of vegetation altered the effects of salinity on carbon equivalency in created wetlands. This study suggests that to conserve the wetland carbon sink function, landscape design for wetland restoration in estuarine regions should consider creating open water wetland in high salinity regions and restoring vegetation in low salinity regions to facilitate the growth of macrophytes such as *Phragmites australis*.

1. Introduction

Coastal wetlands, such as saltmarshes and mangroves, trap and store carbon more efficiently than terrestrial forests and grasslands and are termed as "blue carbon" (Chmura et al., 2003; Doody, 2007; Pendleton et al., 2012; Sjögersten et al., 2014). In recent decades, wetland creation in degraded areas has been common worldwide, becoming a new type of disturbance in coastal regions (Crooks et al., 2011; Konnerup et al., 2014). Coastal wetlands play a vital role in the global carbon cycle; thus, it is important to understand the impacts of wetland creation on their carbon storage and greenhouse gas emissions (Whiting and Chanton, 2001; Altor and Mitsch, 2008).

As human activities such as reclamation, oil mining, plant invasion and road construction, the ecosystems services of wetlands are damaged. For that, wetlands are frequently created to compensate for the loss of wetland area and ecosystems services with the wetland hydrology constructed and vegetation established for specific aims (Craft et al., 2003; Doody, 2007). For example, impounded mangroves were created for insect management throughout Florida, USA in the 20th Century (Rey et al., 2012). Wetlands in Louisiana were diked to restore wetland wildlife habitats with the aim of reducing hurricane and storm damage (Coastal Protection and Restoration Authority of Louisiana, 2012). Wetlands have also been created widely for wastewater treatment (Kivaisi, 2001; DiMuro et al., 2014).

Compared with natural tidal wetlands, permanently inundated artificial wetlands may have lower carbon uptake rates due to reduced plant biomass accumulation (Fennessy et al., 2008; Drexler et al., 2013; Calvo-Cubero et al., 2014). Water-flooding in constructed wetlands is generally believed to reduce CO_2 (Krauss et al.,

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2012) but stimulate CH_4 production (King and Reeburgh, 2002) compared to natural wetlands. However, carbon emissions and accumulation in wetlands are complicated by site-specific conditions such as salinity, plant type, productivity, and the water table (Weston et al., 2014; Couto et al., 2013; Drexler et al., 2013). Therefore, optimizing the construction of coastal wetlands to reduce carbon release and promote carbon absorption is essential in the context of global climate warming.

Salinity is an influential and highly variable environmental parameter in coastal regions, particularly in estuarine ecosystems. Salinity has been shown to affect marsh carbon cycling (Poffenbarger et al., 2011; Morrissey et al., 2014; Chen et al., 2014), where increasing salinity may increase (Marton et al., 2012) or decrease (Krauss et al., 2012) CO_2 emissions and promote (Chen et al., 2014; Weston et al., 2014) or reduce (Neubauer, 2013; Antonellini and Mollema, 2010) carbon sequestration. Thus the impacts of wetland creation or restoration activities on carbon cycles in estuarine areas are likely to vary spatially with changes in salinity.

Many studies also showed that plants can influence the quantity of greenhouse gases emitted to the atmosphere by affecting gas production, consumption, and transport (Andrews et al., 2013; Koelbener et al., 2010; Picek et al., 2007). However, salinity factor could adjust the effects of plants on greenhouse gas emissions (Munns and Tester, 2008; Tavakkoli et al., 2011; Feng et al., 2014). Salinity stress could suppress plant growth and consequently inhibit photosynthetic rates (Sperling et al., 2014). Sutter et al. (2014) observed that the photosynthetic rate and plant biomass decreased in a wetland as salinity increased from 0 to 6‰. Higher salinity can also reduce plant root activity and stem biomass (Guan et al., 2011). These would together influence the carbon fluxes. In the globe climate change background, a comprehensive evaluation of carbon absorption and emission is essential to properly assess the climate effects of wetland construction (Poffenbarger et al., 2011; Mozdzer and Megonigal, 2013). However, the effect of salinity on greenhouse gas emissions and carbon sequestration in constructed wetlands has not yet been investigated.

This study aims to investigate the effects of salinity on greenhouse gas emissions and calculated the global warming potentials for vegetated and unvegetated sites. We set up experimental ponds of differing salinities in a newly constructed wetland where invasive *Spatina alterniflora* has been eradicated. This wetland covered 1 km^2 and located at the Dongtan wetland of the Yangtze River estuary (121°50′–122°05′ E, 31°25′–31°38′ N), China. *S. alterniflora* is an invasive plant which has caused dramatic environmental problems along the eastern coastline of China (Zuo et al., 2012; Wan et al., 2009). In the Dongtan wetland, invasive *Spartina* has spread over half of the marsh area and has led to remarkable loss of shorebird habitat (Ma et al., 2003; Li et al., 2009). To eliminate the negative impacts of *Spartina* on native biodiversity in the Dongtan wetlands, the local government has decided to launch a large (24.2 km^2) restoration project to eradicate the invasive plant and create suitable wetland habitat for birds. The project design involves dividing the area into dozens of plots with differing types of wetlands (mainly vegetated and unvegetated) to meet the habitat demands of various bird groups. Because wetlands in the Yangtze River estuary range widely in salinity (<0.5‰ to >25‰) (Guo et al., 2014), the arrangement of created wetlands of different salinities and different vegetation coverage may influence spatial patterns of carbon uptake and release, thereby affecting conservation of the carbon sink function in the marsh. This study lays a foundation for future spatial design of created wetlands in the Dongtan restoration project. It may also extend our knowledge of integrating carbon dynamics into wetland creation and restoration projects in other estuarine ecosystems.

2. Materials and methods

2.1. Study site and experimental design

The created wetland where the experiments were conducted covers 1 km^2 in the northern part of Dongtan (Fig. 1). The area was dominated by the invasive plant *S. alterniflora* prior to 2008, when the area was dammed to eradicate *S. alterniflora* and restore shorebird habitat. The wetland was maintained constantly flooded by 50–60 cm water during the study period, and the native plant *Phragmites australis* was planted in specific areas for landscaping and to provide shelter for birds.

The constructed wetland was divided into ponds A, B, and C (Fig. 1), with areas of 0.15, 0.42, and 0.30 km^2, respectively. There were sluices between the study pond and tidal creeks, and between the study pond and a freshwater river beside the pond. The salinities in the different ponds were adjusted by mixing water from the tidal creek and the freshwater river, and were maintained relatively constant across seasons. Water salinity was measured using a YSI556 conductivity meter (YSI Incorporated, Yellow Springs, OH, USA) and adjusted manually by using of 1 mS/cm calibrating liquid (Sutter et al., 2014). Salinity was adjusted automatically according to water conductivity and temperature using the calculation of the Practical Salinity Scale 1978 (PSS-78) (UNESCO, 1981) in this device. Based on measurements from June 2011 to September 2012, salinity averaged 1.5‰ (range: 1.2–1.9‰) in Pond A, and 4.9‰ (range: 2.3–8.2‰) in Pond B. Measurements from June 2012 to September 2012 gave an average salinity of 10.5‰ (range: 10.3–11.0‰) in Pond C. For simplification, we hereafter refer to the salinity of these three ponds as <2‰, ~5‰, and >10‰.

2.2. Sampling and measurements

Three sampling sites were established in unvegetated (invasive *S. alterniflora* was eradicated) and three in vegetated areas (invasive *S. alterniflora* was eradicated and replaced with native *P. australis*) in each pond (Fig. 1). Adjacent sampling sites were always >5 m apart.

2.2.1. Greenhouse gas emissions

The static chamber method following Gulledge and Schimel (2000) was used to estimate wetland greenhouse gas (CO_2, CH_4 and N_2O) emissions from June 25–28 to September 16–19 of 2012. One chamber was set up at each sampling site. Chambers consisted of two sections made of stainless steel: a bottom section (50 cm × 50 cm, height: 20 cm) and an upper section (50 cm × 50 cm, height: 50 cm). The upper section was covered with foam insulation to reduce heat transmission, and the two sections were able to be connected and sealed with a water seal ring. The day before sampling, the bottom section with four 70 cm iron braces was inserted into the sediment until the upper fringe of bottom section was only about 3 cm above the water surface. At the sampling days, the upper section was placed on the bottom section and sealed by water-filled ring – gas samples were taken 0, 10, 20 and 30 min after the two chamber sections were connected together.

Gases from each chamber were sampled between 10:00 and 14:00 on each sampling date. Measuring gas flux between 10:00 and 14:00 were generally suggested, because the weather condition during this time period is relatively stable, and the measured values would approximate to the peak values of greenhouse gas emissions of the day (Ding et al., 2004a; Wang and Wang, 2008). Atmospheric pressure was measured with an aneroid barometer (HA29DYM3, Heng Odd Instrument Co., Ltd., Beijing, China). Air temperature in each chamber was measured with an electronic thermometer (JM624, JinMing Instrument Co., Ltd., Tianjin, China). After sampling, CO_2, CH_4, and N_2O concentrations were measured