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Soil quality assessment of coastal wetlands in the Yellow River Delta of China based on the minimum data set

Guangliang Zhang[a,1], Junhong Bai[a,*,1], Min Xi[b], Qingqing Zhao[a], Qiongqiong Lu[a], Jia Jia[a]

[a] State Key Laboratory of Water Environment Simulation, Beijing Normal University, Beijing 100875, PR China
[b] College of Environmental Science and Engineering, Qingdao University, Qingdao 266071, PR China

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ABSTRACT

Little information is available to assess the dynamic changes in wetland soil quality in coastal regions, though it is essential for wetland conservation and management. Soil samples were collected in *Suaeda salsa* wetlands (SWs), *Tamarix chinensis* wetlands (TWs), *Suaeda salsa*–*Tamarix chinensis* wetlands (STWs), freshwater *Phragmites australis* wetlands (FPWs) and saltwater *Phragmites australis* wetlands (SPWs) in three sampling periods (i.e., summer and autumn of 2007 and spring of 2008). According to the flooding characteristics of these wetlands, the study area could be grouped into three sub-regions: short-term flooding region (STFR), seasonal flooding region (SFR) and tidal flooding region (TFR). Soil quality was evaluated using the soil quality index (SQI), which was calculated using the selected minimum data set (MDS) based on principal components analysis (PCA). Our results showed that soil salt content (SSC), total carbon (TC), magnesium (Mg), nitrate nitrogen (NO_3^--N) and total sulfur (TS) consisted of a MDS among 13 soil properties. The SQI values varied from 0.18 to 0.66 for all soil samples, of which the highest and lowest SQI values were observed in TFR. The average SQI values were significantly higher in summer (0.50 ± 0.13) than in spring (0.37 ± 0.13) and autumn (0.36 ± 0.11) in the whole study area ($p < 0.05$). The average SQI values followed the order STFR (0.44 ± 0.12) > TFR (0.41 ± 0.15) > SFR (0.35 ± 0.09) although no significant differences were observed among the three regions ($p > 0.05$). SPWs and SWs soils showed higher SQI values (0.50 ± 0.10 and 0.47 ± 0.15, respectively) than TWs (0.30 ± 0.08) soils ($p < 0.05$). The SSC was the dominant factor of soil quality with its proportion of 34.1% contributing to the SQI values, followed by TC (24.5%) and Mg (24.1%). Correlation analysis also showed that SQI values were significantly negatively correlated with SSC. SSC might be a characteristic indicator of wetland soil quality assessment in coastal regions. The findings of this study showed that the SQI based on MDS is a powerful tool for wetland soil quality assessment.

1. Introduction

The soil quality of wetlands is significantly influenced by many critical environmental processes including sediment deposition, freshwater–saltwater interaction, delta accretion and material-energy exchanges (Bai et al., 2012). Bai et al. (2012) demonstrated that flow-sediment regulation has increased sediment and chemical matter input to the Yellow River Delta during the period from late June to early July since 2002. Additionally, previous studies focused on the effects of different management types on farmland soil quality (Nakajima et al., 2015; Armenise et al., 2013; Nesbitt and Adl, 2014), soil quality assessment of forestland (Zornoza et al., 2007; Ngo-Mbogba et al., 2015) and grassland (Navas et al., 2011; Askari and Holden, 2014) under different land covers. However, little information is available on the comprehensive soil quality assessment in coastal wetlands with high hydrological fluctuations. Therefore, a better understanding of wetland soil quality and its spatio-temporal dynamic characteristics is needed for sustainable management and conservation of coastal wetlands.

After the concept of integrative indices was applied to soil ecosystems by Larson and Pierce (1991), many methods, such as soil quality cards and test kits (Ditzler and Tugel, 2002), soil quality index (SQI) methods (Doran and Parkin, 1994; Andrews et al., 2002), fuzzy association rules (Xue et al., 2010), dynamic soil quality models (Larson and Pierce, 1994) and the soil management assessment framework (Andrews et al., 2004; Karlen et al., 2008), have been established for soil quality assessment and soil

* Corresponding author. Tel.: +86 01058802029.
E-mail address: Baijh@126.com (J. Bai).
[1] These authors contributed equally to this work and should be considered co-first authors.

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management of farmlands. Comparatively, the SQI method has been widely applied (Andrews et al., 2002) due to its simplicity and quantitative flexibility (Chen et al., 2013). The SQI method involves three main steps (Andrews et al., 2002): (1) choosing appropriate indicators for a minimum data set (MDS); (2) transforming indicator scores; and (3) combining the indicator scores into the index. A large number of soil parameters (e.g., soil physical, chemical and biochemical properties) are needed to make the assessment result more accurate (Marzaioli et al., 2010; Bonanomi et al., 2011). However, there was no consensus on a definitive set of soil properties for soil quality assessment because soil quality is very complex (Liu et al., 2014a). Many researchers have agreed to establish a minimum data set (MDS) adequately representing the total data set (TDS) to reduce the cost of soil quality assessment (Glover et al., 2000; Rezaei et al., 2006; Qi et al., 2009).

The identification of the MDS can be conducted using methods such as linear and multiple regression analysis, factor analysis, discriminant analysis and scoring functions (Yao et al., 2013). Factor analysis is widely used to identify the MDS because it can reduce redundant information in the original data set (Yao et al., 2013). The process of establishing a MDS is not standardized as the indicators vary significantly, and it depends on practical research. Chen et al. (2013) selected sand, clay, cation exchange capacity, TP, AP, exchangeable Mg, available Fe and available boron as the MDS to assess soil quality in farmlands of Northeast China. Liu et al. (2014b) assessed paddy soil quality using an MDS including TN, pH, available Si, available Zn and microbial biomass carbon from twenty soil variables. Gong et al. (2015) selected total salt content, TN, pH and soil water content as the final indicators to calculate SQIs of farmlands, natural forestland, saline and alkaline land, desert and sand land. Armenise et al. (2013) assessed soil quality of croplands under different managements using the MDS including clay, SOM, exchangeable K^+, plant available water and AP. Therefore, MDS indicators and scoring functions might be associated with soil types and land-use types or even with time due to varying soil properties (Andrews and Carroll, 2001).

The Yellow River Delta is one of the largest deltas in China, and the high sediment load has made this delta one of the richest sediment supplies in the world (Bai et al., 2015) despite the declining trend of sediment load from 1950 to 2007 (Peng et al., 2010). The amount of sand and soil carried and deposited per year is very high in the delta due to serious soil erosion in the Loess Plateau region (Zhao et al., 2004; Xu et al., 2002). Meanwhile, massive sediment transport brings a large amount of nutrients from the Loess Plateau and the upstream agricultural irrigation regions to the Yellow River Delta (Wang and Liang, 2000); thus, the nutrients in the transported sediments and developed wetland soils are essential in sustaining estuarine ecosystem health in this region (Peng et al., 2010; Zhao et al., 2016).

The primary objectives of this study were (1) to establish a minimum data set (MDS) of wetland soil quality in the Yellow River Delta; (2) to compare the soil qualities among different wetlands in three sampling seasons and analyze the spatio-temporal soil quality dynamics; and (3) to determine the dominant factors influencing soil quality of coastal wetlands in the Yellow River Delta.

2. Materials and methods

2.1. Study area

The study area is located in the Yellow River Delta (37°40′–37°50′ N and 118°57′–119°20′ E, Fig. 1), Shandong province, China. It has a warm-temperate and continental monsoon climate, with annual mean precipitation of 640 mm and annual mean evaporation of 1962 mm. The annual mean air temperature is 11.9 °C, with 196 frostless days (Bai et al., 2015). The soil type in this region is mainly coastal saline soil, derived from the sediment and the parent materials of loess soil (Huang et al., 2012). The dominant vegetation includes *Phragmites australis*, *Suaeda salsa* and *Tamarix chinensis*.

According to the flooding characteristics of the wetlands, the area could be grouped into three sub-regions: short-term flooding region (STFR), seasonal flooding region (SFR) and tidal flooding region (TFR) (Fig. 1). The flooding duration of the STFR lasted less than one month, beginning from the water and sediment regulation project in the summer. The flooding of the SFR lasted three or four months due to its low elevation. The TFR was obviously affected by tidal flooding twice a day. Each region could also be divided into three kinds of wetlands based on the dominant species. The STFR and SFR mainly include *Suaeda salsa*–*Tamarix chinensis* wetlands (STWs), *Suaeda salsa* wetlands (SWs) and freshwater *Phragmites australis* wetlands (FPWs). The TFR can be divided into *Suaeda salsa* wetlands (SWs), *Tamarix chinensis* wetlands (TWs) and saltwater *Phragmites australis* wetlands (SPWs). Overall, there were five types of wetlands (i.e., SWs, TWs, STWs, FPWs and SPWs) based on vegetation covers in this study area.

2.2. Soil collection and analysis

Surface soil (0–20 cm) samples were collected from three sampling regions including the STFR, SFR and TFR in summer (August) and autumn (November) of 2007 and spring (April) of 2008. STFR soils with 5–6 replicates and TFR soils with 13 replicates in three sampling seasons, and SFR soils with 4 replicates in autumn and spring were sampled. SFR soils were not collected in summer due to difficult sampling. In total, 63 surface soil samples were obtained and used for the determination of 13 soil physico-chemical properties. All soil samples were then placed in polyethylene bags, and transported to the laboratory. All soil samples were air dried at room temperature for three weeks and sieved through a 2-mm nylon sieve to remove coarse debris. All the air-dried soil samples were then ground with a pestle and mortar until all particles passed a 0.149-mm nylon sieve. Additionally, a single 4.8-cm diameter soil core was collected from each site for the determination of bulk density (BD) and soil water content (SWC).

Soil samples were digested using a mixture of $HClO_4$, HNO_3 and HF in Teflon tubes for analysis of total phosphorus (TP), total potassium (TK), total magnesium (Mg) and total sulfur (TS). The digested sample solutions were analyzed using inductively coupled plasma atomic emission spectrometry (ICP/AES). The total carbon (TC) and total nitrogen (TN) contents were measured using an Elemental Analyzer (CHOS Elemental Analyzer, Vario EL, Germany). Available phosphorus (AP) was determined using the Mo-Sb colorimetric method. Nitrate nitrogen (NO_3^--N) was measured using an Ion Chromatograph (Dionex-300). TK and Mg were determined using the flame photometer method.

Soil organic matter (SOM) was measured using the dichromate oxidation method (Nelson and Sommers, 1982). Soil pH was measured using a Hach pH meter (Hach Company, Loveland, CO, USA) (soil:water = 1:5). Soil salt content (SSC) was determined in the supernatant of 1:5 soil–water mixtures using a salinity meter (VWR Scientific, West Chester, PA, USA). The soils were oven dried at 105 °C for 24 h and weighed for the determination of BD and SWC.

2.3. Soil quality assessment method

The first step for soil quality assessment is to select soil quality indicators that can influence the capacity of a soil to perform and are sensitive to the final outcome (Nakajima et al., 2015). Those selected indicators compose the MDS. We employed Principal Component Analysis (PCA) as a data reduction tool in combining the norm values to establish the MDS for the study area. Secondly,