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RESEARCH ARTICLE

Effects of sediment burial disturbance on macro and microelement dynamics in decomposing litter of *Phragmites australis* in the coastal marsh of the Yellow River estuary, China

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Abstract From April 2008 to November 2009, a field decomposition experiment was conducted to investigate the effects of sediment burial on macro (C, N) and microelement (Pb, Cr, Cu, Zn, Ni, and Mn) variations in decomposing litter of *Phragmites australis* in the coastal marsh of the Yellow River estuary. Three one-off sediment burial treatments [no sediment burial (0 mm year^{-1}, S_0), current sediment burial (100 mm year^{-1}, S_{10}), and strong sediment burial (200 mm year^{-1}, S_{20})] were laid in different decomposition sites. Results showed that sediment burials showed significant influence on the decomposition rate of *P. australis*, in the order of S_{10} (0.001990 day^{-1}) ≈ S_{20} (0.001710 day^{-1}) > S_0 (0.000768 day^{-1}) ($p<0.05$). The macro and microelement in decomposing litters of the three burial depths exhibited different temporal variations except for Cu, Zn, and Ni. No significant differences in C, N, Pb, Cr, Zn, and Mn concentrations were observed among the three burial treatments except for Cu and Ni ($p>0.05$). With increasing burial depth, N, Cr, Cu, Ni, and Mn concentrations generally increased, while C, Pb, and Zn concentrations varied insignificantly. Sediment burial was favorable for C and N release from *P. australis*, and, with increasing burial depth, the C release from litter significantly increased, and the N in litter shifted from accumulation to release. With a few exceptions, Pb, Cr, Zn, and Mn stocks in *P. australis* in the three treatments evidenced the export of metals from litter to environment, and, with increasing burial depth, the export amounts increased greatly. Stocks of Cu and Ni in *P. australis* in the S_{10} and S_{20} treatments were generally positive, evidencing incorporation of the two metals in most sampling times. Except for Ni, the variations of C, N, Pb, Cr, Cu, Zn, and Mn stocks in *P. australis* in the S_{10} and S_{20} treatments were approximated, indicating that the strong burial episodes (S_{20}) occurred in *P. australis* marsh in the future would have little influence on the stocks of these elements. With increasing burial depths, the *P. australis* was particularly efficient in binding Cu and Ni and releasing C, N, Pb, Cr, Zn, and Mn, implying that the potential eco-toxic risk of Pb, Cr, Zn, and Mn exposure might be very serious. This study emphasized the effects of different burials on nutrient and metal cycling and mass balance in the *P. australis* marsh of the Yellow River estuary.

Keywords Decomposition · C and N · Heavy metals · Coastal marsh · Yellow River estuary

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Introduction

Coastal marshes have long been recognized by their remarkable rates of primary productivity (Bouchard and Lefeuvre 2000), and a substantial part of the annual plant production becomes litter (Foote and Reynolds 1997). The decomposition rates of detrital material in coastal marshes not only affect the accumulation rates of organic matter and the transfer of

nutrients and chemical elements between trophic levels (Sun et al. 2012a) but also influence the potential export of material from coastal marshes and estuaries to coastal waters (Hodson et al. 1984; White and Howes 1994). A number of extreme conditions such as high potential evapotranspiration, frequent ebb and flow of tide, inundation by seawater, and burial in sediment exist in intertidal zone (Baldwin and Maun 1983; Maun 1998; Deng et al. 2008). Among them, sediment burial has been recognized as a major factor influencing the decomposition rates of detrital material in coastal marshes (Vargo et al. 1998). Different sediment disturbances can result in the litters being directly buried within the sediment to different extent, which may greatly affect the processes of material cycling (Sun et al. 2015).

Coastal marshes generally act as a geochemical trap for heavy metals bonded in the sediments (Williams et al. 1994), and various hydrological processes and sediment physicochemical properties significantly influence the biochemical processes of heavy metals in marshes (Bai et al. 2014). With the rapid industrialization and economic development in coastal zone, heavy metal has become one of the most serious pollutants in coastal regions due to their toxicity, persistence in natural conditions, and ability to be incorporated into food chains (Armitage et al. 2007; Sakan et al. 2009; Wang et al. 2013; Xiao et al. 2015). Most of heavy metals imported into coastal marshes are rapidly fixed onto the solid phase, where a number of physical and chemical properties will determine the strength of metal retention. Only a small proportion of the metals dissolves and becomes available for plant uptake (Du Laing et al. 2006). Plant uptake directly reduces the input of metals into adjacent waters (Chen et al. 2000; Vandecasteele et al. 2005). Plant detritus can provide a sink if, during decomposition, metals could be bound to the litter by passive sorption on organic surfaces or by physiological mechanisms of microbial colonizers. Plant detritus can also act as a metal source when microbial activity mobilizes metals or when it becomes available to deposit feeders (Gadd 1993; Ledin 2000; Weis and Weis 2004). Particularly, the consumption of plant detritus that contained metals could cause metal accumulation and toxic effects in higher trophic levels (Dorgelo et al. 1995; Weis et al. 2002; Du Laing et al. 2002; Zhang et al. 2010). Considerable efforts have been conducted in the past two decades to study the litter decomposition in different coastal marshes (Mendelssohn et al. 1999; Anesio et al. 2003; Du Laing et al. 2006; Menéndez and Sanmartí 2007; Quintino et al. 2009; Simões et al. 2011) and mangrove swamps (Robertson 1988; Tam et al. 1998; Dick and Osunkoya 2000; Nielsen and Andersen 2003; Ramos e Silva et al. 2007; Sánchez-Andrés et al. 2010; Keuskamp et al. 2015). Most of these studies focused on exploring the decomposition rates and nutrient dynamics of different litters and the roles of abiotic (temperature, nutrient enrichment, salinity gradient, and tidal flooding, duration, etc.) and biotic factors (fungi, meiofauna, free and attached microorganisms, etc.) on decomposition, whereas information on the effects of sediment burial on litter decomposition is still very limited. In addition, only few studies discussed the variation or accumulation of heavy metals in plant detritus in different coastal marshes (Larsen and Schierup 1981; Zawislanski et al. 2001; Weis and Weis 2004; Du Laing et al. 2006; Pereira et al. 2007), while information on the influences of sediment burial on heavy metal dynamics in decomposing litters remains scarce. In China, studies on litter decomposition of mangrove swamps in the tropical and subtropical regions have been carried out in the early 1990s (Zhuang and Lin 1993; Zhang and Lin 1998; Huang et al. 2001; Sheng 2009; Chen 2013; Li and Ye 2014). In contrast with that, the research in coastal marshes or estuarine marshes start quite late, and current studies also focus on the tropical and subtropical regions of China, such as the coastal marshes in the Hangzhou Bay (Shao et al. 2014), and the estuarine marshes in the Yangtze River estuary (Zhou et al. 2006; Chen 2008; Guan 2013) and the Min River estuary (Tong et al. 2011; Zhang et al. 2014), while information on the coastal marshes in the warm temperate regions of China (such as Liao River estuary and Yellow River estuary) remains scarce. Moreover, present studies mainly focus on litter decomposition characteristics and its related affecting factors, while information on heavy metal dynamics in decomposing litters is still very limited.

The Yellow River is well known as a sediment-laden river. Every year, approximately 1.05×10^7 tons of sediment is carried to the estuary and deposited in the slow flowing landform, resulting in vast floodplain and special marsh landscape (Xu et al. 2002). Sediment deposition is an important process in the formation and development of coastal marsh in the Yellow River Delta. The deposition rate of sediment in the Yellow River not only affects the formation rate of coastal marsh but also influences the water or salinity gradients and the succession of plants from the land to the sea. *Phragmites australis* is a prevalent plant in the coastal marsh of the Yellow River estuary, which is often affected by the sediment of tide physical disturbance and river flooding. It was reported that the annual runoff of the Yellow River showed great interannual changes since the 1980s. The runoff reached the maximum value of 49.1 billion m^3 in 1983 and then decreased and fluctuated at 20.0 billion m^3 in the following several years. From 1997 to 2002, the annual runoff was below 10.0 billion m^3 (Cui et al. 2009). The low flows of the Yellow River led to a significant decrease in freshwater supply to the estuary, and the *P. australis* marshes near

the estuary exhibited seriously degraded status. In order to restore degraded marshes, the "flow-sediment regulation project" (FSRP) was initiated by the nation in 2002. The purpose of the FSRP was to increase the supply of freshwater and sediment for the Yellow River estuary by discharging the water in the Xiaolangdi Reservoir and scouring the sediment in the reservoir and riverbed (Cui et al. 2009). Although the FSRP increased the runoff and sediment for the Yellow River estuary, it could also produce some negative effects. Before the enforcement of FSRP, lower concentrations of As and heavy metals were reported in marsh soils of the Yellow River estuary (Rui et al. 2008). However, the concentrations of some heavy metals (such as As, Cr, Pb, and Cd) in the restored marsh were much higher than those in the degraded marsh, indicating that the heavy metal pollution in the restored marsh might become more serious after the regulation (Bai et al. 2012, 2015). In addition, during the FSRP (from June to July every year), the river water flooded the *P. australis* marshes near the estuary and resulted in the litters being directly buried within the sediment to considerable thickness (approximately 50~60 mm), which might significantly influence litter decomposition rate and element variations in decomposing litter (Mou 2010). However, little is known about the impacts of one-off burial on decomposition rate and heavy metal stocks of *P. australis* in the coastal marsh of the Yellow River estuary.

In this paper, the effects of one-off burial disturbance on macro (C, N) and microelement (Pb, Cr, Cu, Zn, Ni and Mn) variations of *P. australis* were investigated by litterbag technique. It is hypothesized that the decomposition rates and macro/microelement dynamics differ among different burial disturbances, which will have great influences on the functioning of *P. australis* marsh in the Yellow River estuary. The primary objectives of this study were (i) to examine whether sediment burials caused by one-off burial episodes would have great impacts on decomposition of *P. australis*, (ii) to determine the dynamics of C, N, and metals in *P. australis* during decomposition as affected by different sediment burials, and (iii) to determine the influences of sediment burials on C, N, and metal stocks in *P. australis* during decomposition.

Materials and methods

Study site

This study was conducted in the coastal marsh of the northern Yellow River estuary (Fig. 1), located in the Nature Reserve of Yellow River Delta (37° 35′ N~38° 12′ N, 118° 33′ E~119° 20′ E) in Dongying City, Shandong Province, China. The nature reserve is of typical continental monsoon climate with distinctive seasons. The average temperatures in spring, summer, autumn, and winter are 10.7, 27.3, 13.1, and −5.2 °C, respectively. Annual evaporation is 1962 mm, and annual precipitation is 551.6 mm, with about 70 % of precipitation occurring between June and August. The soils in the study area are dominated by intrazonal tide soil and salt soil, and the main vegetations include *P. australis*, *Suaeda salsa*, *Triarrhena sacchariflora*, *Myriophyllum spicatum*, and *Tamarix chinensis* (Sun et al. 2012b). Coastal marsh is the main marsh type, with an area of 964.8 km^2, accounting for 63.06 % of the total area of the Yellow River Delta (Cui et al. 2009). The tide in the intertidal zone of the Yellow River estuary is irregular semidiurnal tide, and the mean tidal range is 0.73~1.77 m (Li et al. 1991). The sequence of geomorphic units is complete in intertidal zone of the Yellow River estuary due to the protection of Nature Reserve, which generally comprises three areas in a seaward direction: high marsh (river bank), middle marsh, and low marsh. In the river bank, the most common species is *P. australis*, and to a less extent by *Calamagrostis pseudophragmites* and *Imperata cylindrica* (in similar proportions (from 5 to 10 % ground cover)). As a prevalent plant in high marsh, *P. australis* is often affected by the sediment deposition of tidal disturbance, bioturbation, and Yellow River flooding during FSRP. The sedimentary rate in *P. australis* marsh is about 90~100 mm $year^{-1}$, and approximately 60~70 mm occurs from June to July due to the significant effects of both tidally induced sediment and FSRP (Mou 2010). One experimental plot was laid in the abovementioned high marsh, and three subplots were laid in it along the river bank (Fig. 1).

Experimental design

Litter decomposition was studied by litterbag technique at the experimental plot from April 2008 to November 2009. On 20 March 2008, the litters were collected from *P. australis* community. In order to weaken the fragmentation impact of snowfalls and strong winds in winter, the standing litter was selected for use in this study. Each 20× 20 cm litterbag was made of nylon netting (0.5 mm mesh) and filled with 15 g litter (dried weight). In order to investigate the effects of one-off burial disturbance on decomposition and macro/microelement concentrations of *P. australis*, three one-off sediment burial treatments (no sediment burial (0 mm $year^{-1}$, S_0), current sediment burial (100 mm $year^{-1}$, S_{10}), and strong sediment burial (200 mm $year^{-1}$, S_{20})) were laid in different decomposition subplots. On 21 April 2008, the litterbags were artificially buried to depths of 0, 100, and 200 mm to simulate the S_0, S_{10}, and S_{20} treatments, respectively. In order

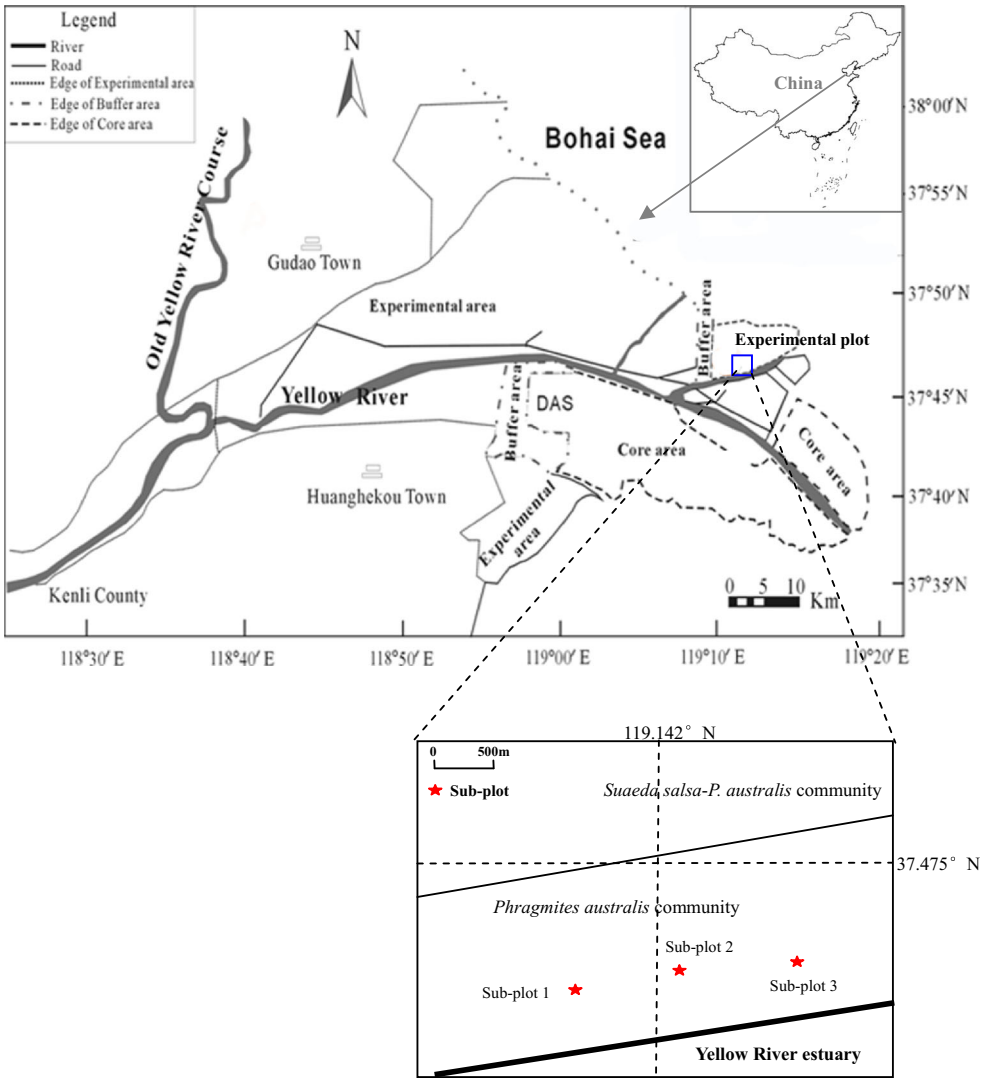

Fig. 1 Sketch of the Yellow River estuary and the experimental subplots (*red star*)

to prevent the litterbags from being affected by sediment burial disturbance during decomposition, each subplot was tightly enclosed by nylon netting (0.25 mm mesh, 1.5 m height). The bulk density of sediment used in this experiment is 1.28±0.08 g cm^{-3} and presents silty clay texture, with 7.83±2.52 % of clay, 76.84±2.39 % of silt, and 15.33±0.13 % of sand (fine sand). The sediment shows low organic matter and total nitrogen contents, with the values of 1.10±0.14 % and 0.63±0.03 mg g^{-1}, respectively. The pH and electrical conductivity of the sediment are 7.90±0.05 and 3.58±1.48 mS cm^{-1}, respectively. The concentrations of Pb, Cr, Cu, Zn, Ni, and Mn in sediment are 48.01±0.94, 47.17±10.80, 29.65±1.48, 66.43±3.01, 26.95±1.36, and 579.46±14.84 mg kg^{-1}, respectively.

The experiment included nine sampling times with different intervals (11 July 2008 (80 days), 09 August 2008 (109 days), 20 September 2008 (151 days), 20 October 2008 (181 days), 15 November 2008 (207 days), 26 April 2009 (371 days), 25

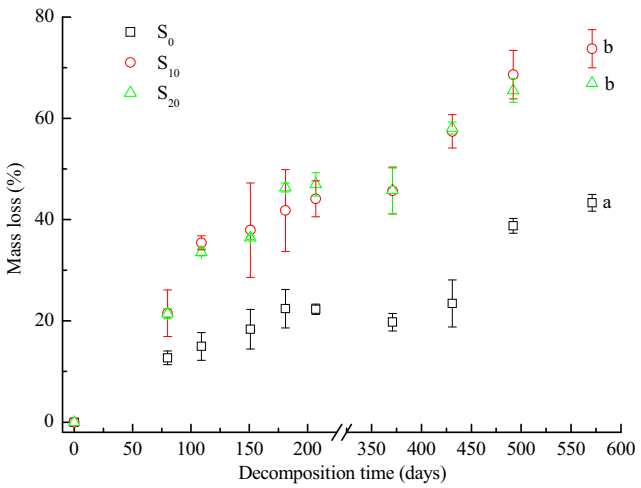

Fig. 2 Mass losses of *Phragmites australis* in different burial treatments during decomposition. Values with the same letters are not significantly different at $p<0.05$

Table 1 Regression equations and parameters relating natural logarithm of mass remaining (y) with decomposition days (t)

Burial treatments	Equations	k (day^{-1})	R	p value	$t_{0.95}$ (year)
S_0	$y=-0.05612-0.000768t$	0.000768	−0.898	<0.001	10.49
S_{10}	$y=-0.10819-0.001990t$	0.001990	−0.957	<0.001	3.98
S_{20}	$y=-0.15764-0.001710t$	0.001710	−0.948	<0.0001	4.55

k, decomposition rate, $t_{0.95}$ time needed for 95 % of dry mass decomposed (year)

June 2009 (431 days), 25 August 2009 (492 days), and 12 November 2009 (571 days)), and on each sampling date, three or four litterbags were retrieved from each subplot. After retrieval, these litterbags were immediately taken back to the laboratory, and the plant roots, lichen, sediment, and macroinvertebrates were removed from the remaining litter. All litterbags were further cleaned gently in deionized water and weighed after being dried to a constant weight.

In situ measurements

Sediment temperatures (0, 10, and 20 cm) were measured in the three subplots on sampling date. Sediment moisture and electrical conductivity (EC) in 0, 10, and 20 cm depths were determined in situ by high-precision moisture measuring instrument (AZS-2) and soil and solution EC meter (Field Scout), respectively. Sediment pH in 0, 10, and 20 cm depths was measured by portable pH meter (IQ150).

Sample analysis

The samples of decomposing material and sediment were ground (<0.25 mm) using a Wiley mill and analyzed for total carbon (TC) and total nitrogen (TN) concentrations by an element analyzer (Elementar Vario Micro, German). The organic matter and grain size for the sediment used in this experiment were determined by $K_2Cr_2O_7$ oxidation method (The Committee of Agro-chemistry of the Chinese Society of Soil Science 1983) and the Coulter Laser granulometer, respectively. A 0.1000-g homogenized sediment subsample was digested with 2 mL HNO_3, 1 mL $HClO_4$, and 5 mL HF at 160~190 °C for 16 h. The residue was dissolved in 2 mL of 4 mol L^{-1} HCl and then diluted to 10 mL with deionized water for heavy metal analysis. A 0.2000-g plant subsample was digested in a mixture of 65 % HNO_3 (2 mL) and 30 % H_2O_2 (1 mL). The residue was diluted with deionized water to 10 mL for analyzing heavy metal concentrations. The concentrations of heavy metals (Pb, Cr, Cu, Zn, Ni, and Mn) in all samples were determined by Agilent 7500 ICP-MS (Agilent Company, America). Quality assurance and quality control were assessed using duplicates, method blanks, and standard reference materials (GBW07401 and GBW08513) from the National Research Center for Standards in China with each batch of samples (one blank and one standard for each 20 samples).

Parameter calculation

Litter mass loss (R, %) and decomposition rate (day^{-1}) were calculated by the following equations (Olson 1963):

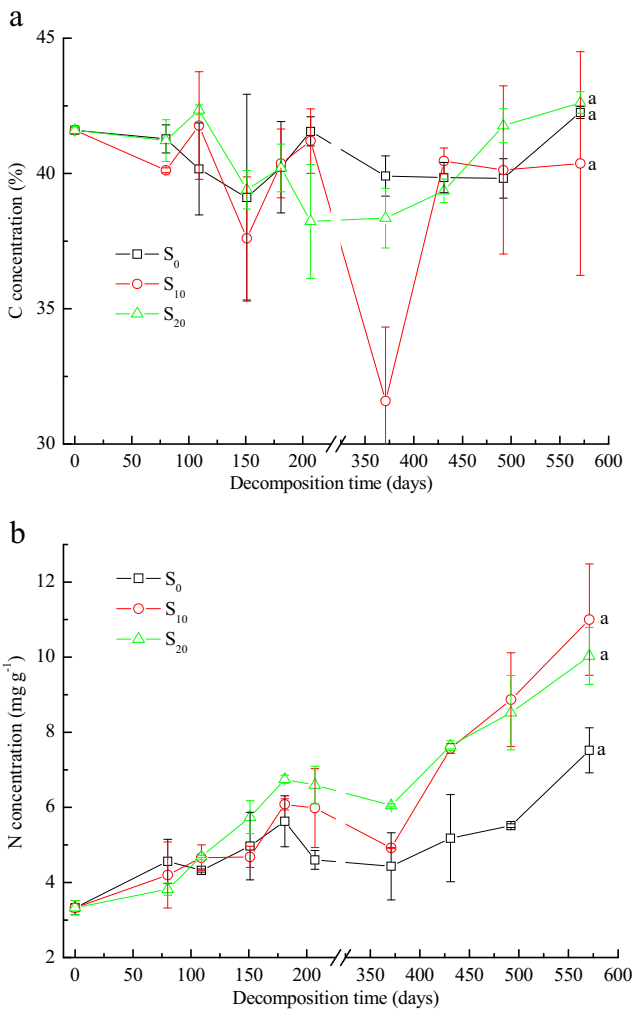

Fig. 3 Dynamics of C (**a**) and N (**b**) concentrations in *Phragmites australis* in different burial treatments during decomposition. Values with the same letters are not significantly different at $p<0.05$

$$R = \left[(W_t - W_0)/W_0\right] \times 100\%$$
$$ln\left(W_t/W_0\right) = -kt$$

where W_0 (g) is the original dry mass and W_t (g) is the dry mass at time "t"; k is the decay constant and t (day) is decomposition time in days.

The accumulation index of the "i" macro/microelement (C, N, Pb, Cr, Cu, Zn, Ni, and Mn) (AI_i) was used to express its accumulation or release status during litter decomposition, which could be calculated by the following equation (Sun et al. 2012a):

$$AI_i = \frac{M_j \cdot C_j}{M_0 \cdot C_0} \times 100\%$$

where M_0 is the original dry mass, C_0 is the original element concentration in litter, M_j is the dry mass at time "j", and C_j is the element concentration in litter at time "j". $AI > 100$ % indicated net element accumulation, whereas $AI < 100$ % indicated net element release.

Statistical analysis

The samples were presented as means over the replications, with standard deviation (SD). The analysis of variance (ANOVA) tests (SPSS for windows 11.0) was employed to determine if treatments differed significantly ($p < 0.05$). If ANOVA showed significant differences, multiple comparison of means was undertaken by Tukey's test with a significance level of $p = 0.05$.

Results

Mass loss and decomposition rate

The mass losses of *P. australis* in the three burial treatments generally increased and the values presented $S_{10} \approx S_{20} > S_0$ ($p < 0.05$) (Fig. 2). Over 571 days of decomposition, the percent of dry mass remaining in the S_0, S_{10}, and S_{20} treatments were 56.67, 26.25, and 33.08 %, respectively. Sediment burials showed great influence on the decomposition rate of *P. australis*, in the order of S_{10} ($0.001990\ day^{-1}$) $\approx S_{20}$ ($0.001710\ day^{-1}$) $> S_0$ ($0.000768\ day^{-1}$) ($p < 0.05$) (Table 1). Significantly higher $t_{0.95}$ (time needed for 95 % of dry mass decomposed) were observed for S_0 treatment (10.49 years) compared to the S_{10} (3.98 years) and S_{20} (4.55 years) treatments.

Macro and microelement concentrations

During decomposition, the C concentrations in *P. australis* fluctuated greatly and varied between 39.11 and 42.26 % for S_0, from 31.59 to 41.77 % for S_{10}, and from 38.23 to 42.62 % for S_{20}, respectively (Fig. 3a), while the N concentrations in the three treatments generally demonstrated increasing tendency (Fig. 3b). The trace metal concentrations in the three

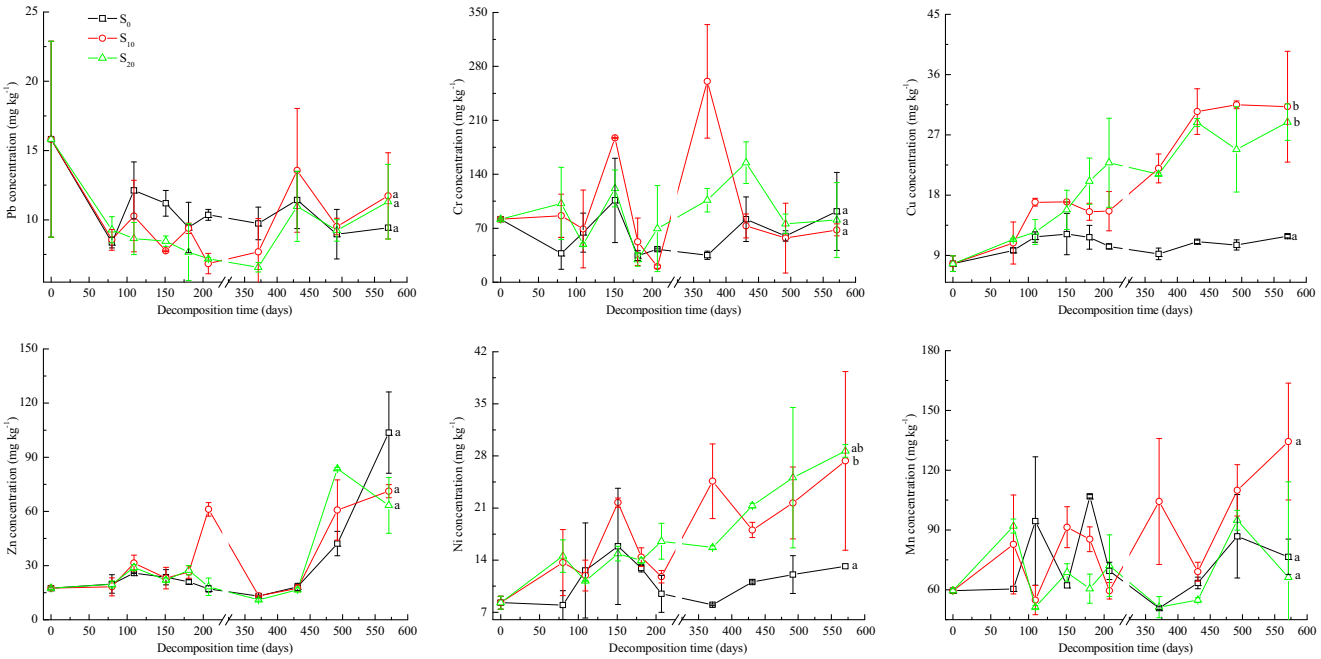

Fig. 4 Variations of metal concentrations (mg kg^{-1} dry mass) in *Phragmites australis* in different burial treatments during decomposition. Values with the same letters are not significantly different at $p < 0.05$

burial treatments exhibited different temporal variations except for Cu, Zn, and Ni. With a few exceptions, the Cu and Ni concentrations in the S_{10} and S_{20} treatments and the Zn concentrations in the three burials generally showed increasing tendency. Compared to the S_{10} and S_{20} treatments, the variations of Pb, Cr, Cu, and Ni concentrations in the S_0 treatment were less pronounced. With increasing burial depth, Cr, Cu, Ni, and Mn concentrations generally increased, while Pb and Zn concentrations varied insignificantly (Fig. 4). Except for Cu and Ni, no significant differences in C, N, Pb, Cr, Zn, and Mn concentrations were observed among the three burial treatments ($p>0.05$).

Macro and microelement stocks

The AI_c of *P. australis* in the three burial treatments presented $S_{10} \approx S_{20} > S_0$ ($p<0.05$), indicating that sediment burial was favorable for C release (Fig. 5a). The AI_N variations of *P. australis* in the three treatments were significantly different

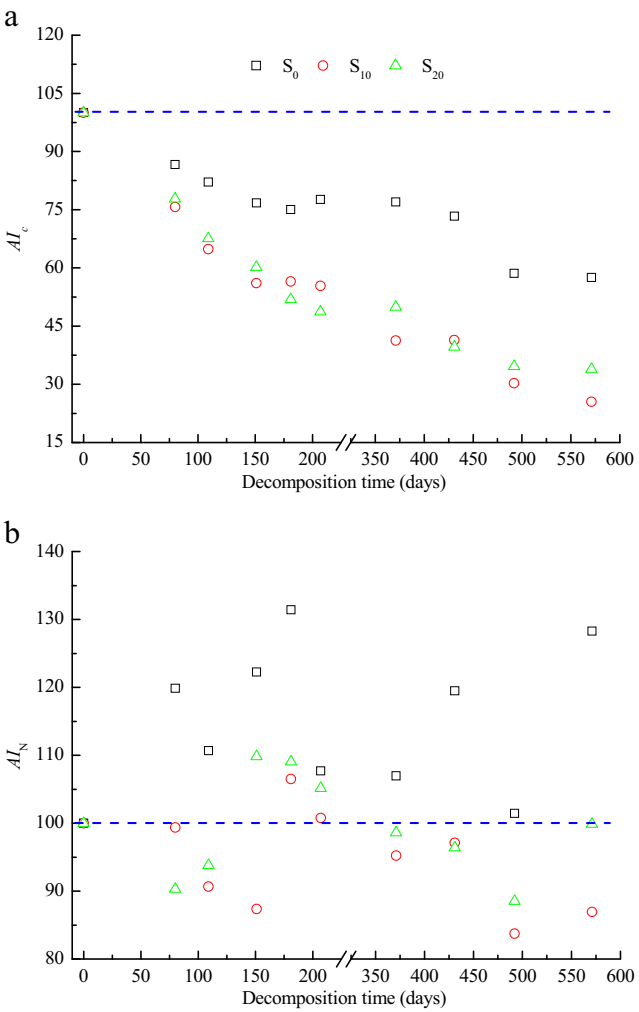

Fig. 5 Variations of C (**a**) and N (**b**) stocks in *Phragmites australis* in different burial treatments during decomposition

($p<0.05$), and, with increasing burial depth, the N in litters generally shifted from accumulation to release during decomposition, indicating that sediment burial was also favorable for N release (Fig. 5b). With a few exceptions, Pb, Cr, and Mn stocks in *P. australis* in the three treatments evidenced the export of metals from litters to environment and, with increasing burial depth, the export amounts increased greatly (Fig. 6). Stocks of Cu and Ni in *P. australis* in the S_{10} and S_{20} treatments were generally positive, evidencing incorporation of the two metals in most sampling times. With increasing burial depth (particularly in the S_{20} treatment), stocks of Zn in *P. australis* shifted from accumulation to release in most periods (Fig. 6).

Discussion

Effects of sediment burial on decomposition rate and macroelement dynamics

Previous studies have indicated that sediment burials significantly inhibited the decomposition of detrital materials via a series of direct and indirect mechanisms, such as compaction of detritus, reduction of gas (O_2 and CO_2) exchange between the detrital layer and the surrounding, and suppression of bacterial and fungal breakdown of detritus (Vargo et al. 1998; Nielsen and Andersen 2003). However, we have drawn a different result. Although there was little difference in decomposition rates of the S_{10} and S_{20} treatments, the values were much higher than that in the S_0 treatment, indicating that sediment burials stimulated the decomposition rate of *P. australis*. There were three possible reasons. Firstly, the difference in decomposition rates in the three treatments was closely correlated with litter quality. Present studies have indicated that litter quality clearly affected decomposition rate, and the C/N ratio was often used as predictors of decomposition rates since it reflected the ratio of carbohydrate and lignin to protein in litter; a high ratio generally induced a slow decomposition rate (Harmon et al. 1990; Hobbic 1996; Chen 1999; Alicia and Roberto 2003). In this study, the C/N ratios of *P. australis* in the S_{10} and S_{20} treatments during decomposition were generally lower than those in the S_0 treatment (Fig. 7), which, to some extent, could better explain the higher decomposition rates in burial treatments. Secondly, the difference in decomposition rates in the three treatments might be affected by the key environmental variables in different burial depths. Ming-Yi et al. (1993) indicated that the anoxic degradation was independence of temperature, and this was tested in our study. In the three treatments, although the average temperatures differed by up to 4.10 and 5.98 °C between 0 and 10 cm depths and between 10 and 20 cm depths,

Table 2 Environmental variables in different sediment burial treatments during decomposition

Burial treatments	Sediment temperature (°C)	Sediment moisture (cm^3 cm^{-3})	EC (mS cm^{-1})	pH
S_0	25.11±10.51a	0.228±0.044a	4.33±2.13a	7.89±0.17a
S_{10}	21.01±9.16a	0.401±0.010b	4.67±1.16a	8.00±0.07a
S_{20}	19.13±8.20a	0.418±0.005b	4.79±1.71a	8.32±0.12a

Different letters within the same column indicate significant differences at $p<0.05$

respectively, the difference in sediment temperatures among the three treatments were not large enough to contribute to the observed difference in decomposition rates as evidenced by the absence of significant correlations between temperatures and decomposition rates ($p>0.05$). Moreover, no significant differences in sediment pH and EC were observed among the three burial depths (Table 2), indicating that these variables might be less important in inducing the difference of decomposition rates. In contrast with them, sediment moisture was significantly different among the three burial treatments ($p<0.05$), and, with increasing burial depth, the moisture significantly increased (Table 2). Previous studies have indicated that, with increasing of moisture, the O_2 in litter would be depleted rapidly, and the metabolism of decomposition microbes would be restrained (Cai 2000; Laiho et al. 2004). As the devoid of O_2, the enzymes such as phenol oxidase, which required O_2 for their activity, were rarely active, and thus inhibited the decomposition of organic matter (Freeman et al. 2004). However, Zhao et al. (2015) found that, in the long-flooding coastal wetland, the mass loss of *P. australis* was the highest. Webster and Benfield (1986), reviewing decomposition in aquatic environments, noted that the effects of anaerobiosis on decomposition rate were not always inhibitory. Particularly, after the organic matter was buried in sediment, the anaerobic conditions formed in different burial depths might have either positive (Ming-Yi et al. 1993), negative (Benner et al. 1984), or even no effect at all (Kristensen and Blackburn 1987) on the decomposition of organic matter. In this study, the positive effect of sediment burials (S_{10} and S_{20} treatments) on decomposition was probably ascribed to the maintenance of adequate sediment moisture for microbial/fungal colonization and activity (Neckles and Neill 1994; Mendelssohn et al. 1999), and this might be verified by Wang et al. (2009) who found that, in the *P. australis* marsh of the Yellow River estuary, although the amounts of bacteria, fungi, and actinomycete in sediment of 10 and 20 cm depths were slightly lower than the surface sediment, the activities of microorganisms in 5~15 cm depth were still very high (Chen and Shi 2010). The high activities of microbes in

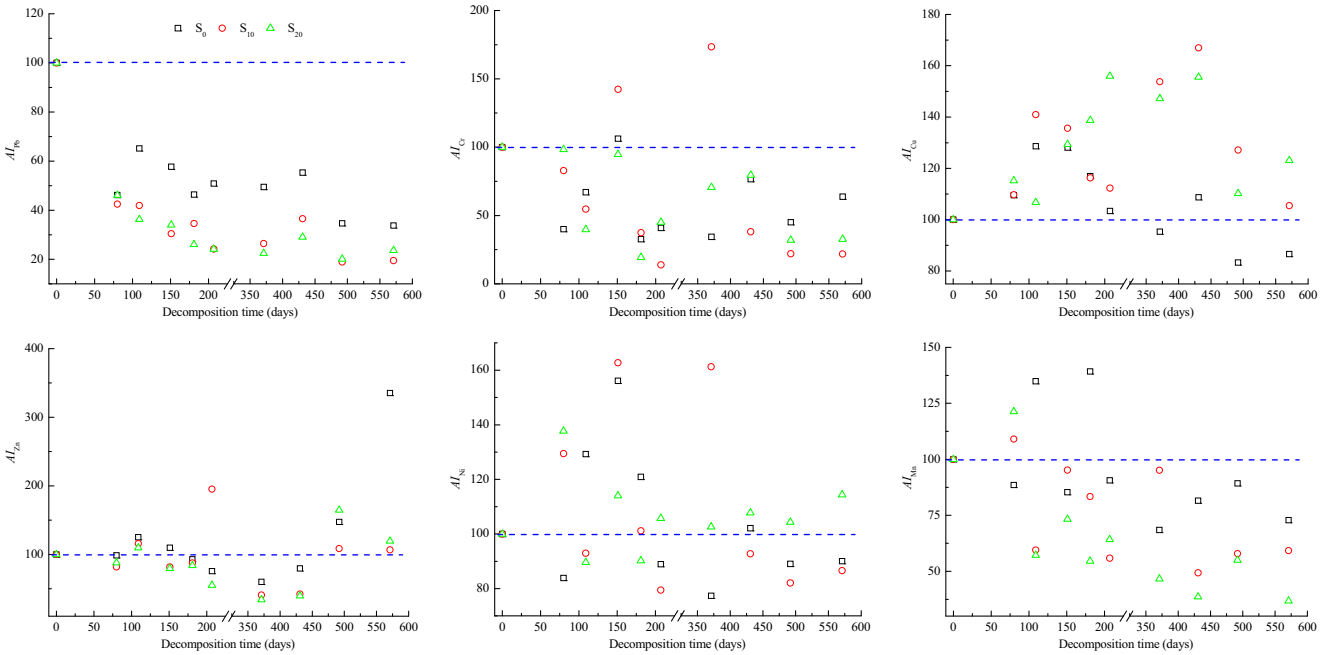

Fig. 6 Variations of Pb, Cr, Cu, Zn, Ni, and Mn stocks in *Phragmites australis* in different burial treatments during decomposition

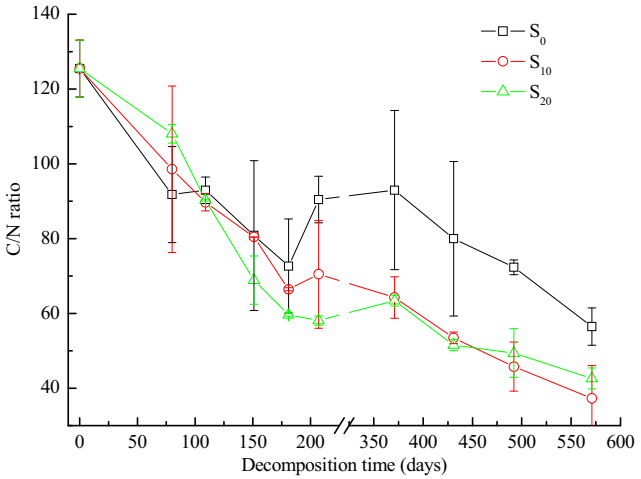

Fig. 7 Variations of C/N ratios in *Phragmites australis* in different burial treatments during decomposition

10~20 cm depth also could be tested by the lower C/N ratios of *P. australis* in the S_{10} and S_{20} treatments compared to the S_0 treatment (Fig. 7). Finally, the activities of macrobenthos such as *Macrophthalmus japonicus* and *Helice wuan* in *P. australis* marsh might also affect the difference in decomposition rates of *P. australis* in the three burial treatments. It was reported that the vertical depth of macroinvertebrates (crabs) caves in *P. australis* marsh (high marsh) varied from 10 to 20 cm, and in these depths, the horizontal or slop habitats were generally established (Mou 2010, Sun et al. 2015). Li (2011) also indicated that the biomass and habitat density of macrobenthos in high marsh of the Yellow River Delta were very high, and the values reached 92.58, 192.37 g m^{-2}, and 168.66, 690.26 ind m^{-2} during spring and autumn. Thus, the great disturbances of macrobenthos in *P. australis* marsh, to some extent, might improve the gas exchange between the detritus in 10~20 cm depth and the surrounding, which probably stimulated the decomposition of *P. australis* in the S_{10} and S_{20} burial treatments. This paper also found that there was little difference in decomposition rates between S_{10} and S_{20} treatments, and the reason might be related to the insignificant differences in sediment temperature, moisture, EC, and pH between them (Table 2). Compared to the S_{10} treatment, the decomposition rate in the S_{20} treatment was slightly lower, which might be dependent on the weak inhibitory on decomposition caused by the relative deficiency of O_2 in 20 cm depth. Moreover, the amounts and activities of microorganisms in sediment of the S_{20} treatment were slightly lower than those of the S_{10} treatment (Wang et al. 2009), which could better explain the lower decomposition rate.

It was anticipated that the dynamics of macroelement (C and N) in decomposing litter differed among the three burial treatments, and this was also tested in this study. Previous studies have indicated that the *P. australis* marsh in the Yellow River estuary was very limited by N (Sun 2015; Cao et al. 2015). Thus, over all sampling periods, the increase of N concentration in *P. australis* in the three burial treatments might be attributed to the N immobilization by microbes from sediment or river-water/seawater in decomposition environments (Sun et al. 2015). The N concentrations in *P. australis* in the S_{10} and S_{20} treatments were generally higher than those in the S_0 treatment (Fig. 3b), and there were three possible reasons. Firstly, although the litterbags in the S_0 treatment were placed in close contact with sediment layer, the chance of infiltration of sediment particles into the litterbags were much lower compared to the S_{10} and S_{20} treatments. Secondly, as mentioned above, although the amounts of microorganisms in sediments of the S_{10} and S_{20} treatments were slightly lower than those in the S_0 treatment, the activities of microorganisms in the two burial depths were still very high. Thirdly, significantly higher sediment moisture was observed in the S_{10} and S_{20} treatments compared to the S_0 treatment during decomposition (Table 2), which might increase the chance of N in sediment water for immobilization by microbes. Similarly, Gessner (2000) found that the N concentrations tended to increase in the leaf, culm, and sheath of *P. australis*, and the reasons were mainly related to the external biological immobilization from decomposition environment (lake water). Sun et al. (2012a) also indicated that the increase of N concentrations in *Calamagrostis angustifolia* during decomposition was dependent on the biological immobilization from sediment and marsh water. Another study by Gessner (2001) indicated that microbial immobilization was a very important process controlling the nutrient dynamics in litter, which was mainly regulated by the C/N ratios in litter and the N availability in decomposition environment (Berg 1986; Köchy and Wilson 1997). Pearson correlation analyses indicated that significantly negative correlations were observed between C/N ratios and N concentrations in the S_0 ($r=-0.934$), S_{10} ($r=-0.882$), and S_{20} ($r=-0.921$) treatments ($p<0.01$), but no significant correlations occurred between C/N ratios and C concentrations ($p>0.05$), indicating that C/N ratios might control the N dynamics in the three burial treatments, while the C variations in litters might be more subjected to the sediment moisture and the activities of microorganisms and macrobenthos as mentioned previously.

This study indicated that sediment burial was favorable for C release from *P. australis*. With increasing burial depth, the release amounts after 571 days of decomposition enhanced 23.67~32.09 % (Fig. 5a), which could be better explained by both appropriate sediment moisture and high activities of microbes and macrobenthos in the S_{10} and S_{20} treatments as

mentioned above. This paper also found that, with increasing burial depth, the N in *P. australis* generally shifted from accumulation to release (Fig. 5b). The difference in N release patterns of the three burial treatments was mainly dependent on the variations of C/N ratios during decomposition. As mentioned previously, the C/N ratios of *P. australis* in the S_{10} and S_{20} treatments were generally lower than those in the S_0 treatments (Fig. 7), indicating that, compared to the S_0 treatment, the N in current (S_{10}) and strong (S_{20}) burial treatments might not be very limited for microorganism during decomposition. Thus, the superfluous N in *P. australis* could be greatly released to the decomposition environments (Fig. 5b). Similar with decomposition rate, the C and N release from *P. australis* in the S_{10} and S_{20} treatments were approximated; implying that the strong one-off burial episodes (S_{20}) occurred in *P. australis* marsh in the future would have little influence on the C and N release from litters.

Effects of sediment burial on microelement concentrations and stocks

This study found that, with a few exceptions, the Cu and Ni concentrations in the S_{10} and S_{20} treatments and the Zn concentrations in the three burial treatments generally showed increasing tendency (Fig. 4). Similar results were reported by other studies. Du Laing et al. (2006) indicated increasing metal concentrations (Cu, Cr, Ni, Pb, and Zn) in leaf blades, sheaths, and stems of *P. australis* during decomposition. Windham et al. (2004) found increasing Cr, Cu, Pb, and Zn concentrations in decomposing leaves and stems of plants in reed marshes. Increases of the metal concentrations could be attributed to different factors, such as contamination by sediment particles, passive sorption onto recalcitrant organic fractions, and active accumulation by microbial colonizers (Breteler et al. 1981; Gadd 1993; Zawislanski et al. 2001; Kovacova and Sturdik 2002; Du Laing et al. 2006). Although the marsh sediment in the S_0 treatment could be easily resuspended by river flooding or tidal wave action, the risk of infiltration of fine particles into the litterbags were much lower compared to the S_{10} and S_{20} burial treatments. Accompanying with river flooding or tidal wave action, the metals in river water or seawater also increased the chance of sorption by *P. australis* in different burial treatments. It was reported that C/N ratio was an effective index in representing decomposition rate and microbial activity since it reflected the ratio of carbohydrate and recalcitrant organic fractions (lignin, cellulose, and hemicellulose etc.) to protein and available nitrogen in litter (Harmon et al. 1990; Hobbic 1996; Cai 2000; Sun et al. 2012a). Pearson correlation analyses showed that significantly negative correlations were observed between Cu concentrations and C/N ratios in the three burial treatments ($p<0.01$). Significantly negative correlations also occurred between Zn concentrations and C/N ratios in the S_0 treatment ($p<0.05$) and between Ni concentrations and C/N ratios in the S_{10} and S_{20} treatments ($p<0.01$) (Table 3). In this study, the C/N ratios of *P. australis* in the three treatments during decomposition generally decreased (Fig. 7), indicating that the activities of microbes in litters might be greatly enhanced, and this might significantly increase the active accumulation of Cu, Zn, and Ni by microorganisms. This conclusion was tested by some related studies. Windham et al. (2004) found that adsorption and accumulation of fine sediment could not be the major cause of increasing metal concentrations in litter, and microbial action was likely one of the major mechanisms responsible for the metal enrichment. Du Laing et al. (2006) also indicated that fungal biomass showed significantly positive correlations with metal concentrations in stem tissue, suggesting an involvement of fungal activity in metal accumulation. Moreover, with the process of decomposition, the proportions of recalcitrant organic fractions significantly increased, which might enhance the chance of physicochemical sorption of Cu, Zn, and Ni onto the remaining recalcitrant organic fractions.

Except for Cu, Zn, and Ni, Pb, Cr, and Mn concentrations in the three burial treatments exhibited different temporal variations during decomposition. On the one hand, the fluctuations of metals in *P. australis* in different burials might be dependent on the complex interactions of the abovementioned factors such as infiltration of fine particles into litters, passive sorption onto recalcitrant organic fractions, and active

Table 3 Correlation coefficients between metal concentrations and carbon/metal (C/M) ratios or carbon/nitrogen (C/N) ratios

Burial treatments	Ratios	Pb	Cr	Cu	Zn	Ni	Mn
S_0	C/M	−0.956[b]	−0.965[b]	−0.978[b]	−0.842[b]	−0.981[b]	−0.980[b]
	C/N	0.738[a]	−0.105	−0.792[b]	−0.647[a]	−0.619	−0.428
S_{10}	C/M	−0.908[b]	−0.676[a]	−0.880[b]	−0.951[b]	−0.943[b]	−0.941[b]
	C/N	0.252	0.045	−0.905[b]	−0.574	−0.768[b]	−0.670[a]
S_{20}	C/M	−0.971[b]	−0.873[b]	−0.922[b]	−0.863[b]	−0.889[b]	−0.958[b]
	C/N	0.487	−0.079	−0.938[b]	−0.448	−0.789[b]	−0.002

[a] Correlation is significant at the 0.05 level

[b] Correlation is significant at the 0.01 level

accumulation by microorganisms. On the other hand, the metal variations in the three burial treatments might be rested with carbon/metal (C/M) and C/N ratios. Once carbon becomes the major constituent of litter, metal concentration could be normalized to carbon content to better interpret the variation of metal concentrations as organic matter decomposed (Pereira et al. 2007). In this study, significantly negative correlations were observed between metal concentrations (Pb, Cr, Cu, Zn, Ni, and Mn) and C/M ratios in the three burial treatments ($p<0.01$ or $p<0.05$) (Table 3), indicating that C/M ratios, to a great extent, might control the metal dynamics in *P. australis* in different burial treatments. Except for Cu, Zn, and Ni, significantly negative correlations also occurred between Mn concentrations and C/N ratios in the S_{10} treatment ($p<0.05$), implying that the Mn variation might be greatly influenced by both active accumulation by microorganisms and physicochemical sorption of dissolved metals onto recalcitrant organic fractions. Although the correlations between Pb concentrations and C/N ratios in the three burial treatments were positive, significant correlation only occurred in the S_0 treatment ($p<0.05$) (Table 3). Previous studies have indicated that the Pb behavior was greatly influenced by iron cycling and organic matter degradation. The oxidation of organic matter might lead to the use of Fe oxide as electron acceptor, and the reduced Fe form might leach from litter (Sundby et al. 2005). Particularly, Pb could be included in formed Fe sulfides as a tracer, and as Fe oxides were reduced, the Pb mobilization occurred (Pereira et al. 2007). In the Yellow River estuary, the Fe concentrations in sediments of coastal marsh were very high, and the values ranged from 16.49 to 33.11 g kg^{-1} (Sun et al. 2013), implying that the Pb mobilization might be enhanced as the Fe oxides were substantially reduced. This paper also found that Cr, Cu, Ni, and Mn concentrations generally increased with increasing burial depth (Fig. 4). There were three probable reasons. Firstly, as mentioned above, the chance of infiltration of fine particles into the litters in the S_{10} and S_{20} burial treatments was much higher than that in the S_0 treatment, which might greatly increase metal concentrations in *P. australis*. Secondly, significantly higher sediment moisture were observed in the S_{10} and S_{20} treatments compared to the S_0 treatment (Table 2), which might increase the chance of metals in sediment water for immobilization by microbes. Finally, the C/N ratios of *P. australis* in the S_{10} and S_{20} treatments were generally lower than those in the S_0 treatment (Fig. 7), implying that the activities of microbes in litters might be greatly enhanced, and this might significantly increase the active accumulation of metals by microorganisms.

This paper showed that, in most periods, Pb, Cr, Zn, and Mn stocks in *P. australis* in the three burial treatments were always lower than the initial ones, indicating that release during the 571-day experiment always exceeded incorporation (Fig. 6). However, the variations of metal stocks between sampling times meant that export was not uniform. Stocks of Cu and Ni in *P. australis* in the S_{10} and S_{20} treatments were generally positive, evidencing incorporation of the two metals in most sampling times. It was hypothesized that the release of Cu and Ni from *P. australis* in the S_{10} and S_{20} treatments was not counterbalanced by sorption due to the strong reducing conditions in sediments (Pereira et al., 2007), as proved by the high concentrations of acid volatile sulfides (AVS) in sediments of the Yellow River estuary (Wu et al. 2007). Particularly, the incorporation of Cu occurred in current (S_{10}) and strong (S_{20}) burial treatments at all times (Fig. 6). Similar results were reported by Windham et al. (2004) and Pereira et al. (2007) who also found Cu enrichment in litters as decomposition proceeded. Therefore, the *P. australis* in the three burial treatments acted as cation exchanger absorbing Cu from sediments or sediment water, and the strong affinity of Cu to organic matter might promote this sorption (Stumm and Morgan 1996). Except for Ni, the variations of Pb, Cr, Cu, Zn, and Mn stocks in *P. australis* in the S_{10} and S_{20} treatments were approximated, indicating that the strong one-off burial episodes (S_{20}) occurred in *P. australis* marsh in the future would have little influence on the stocks of the five metals. With increasing burial depths, the *P. australis* was particularly efficient in binding Cu and Ni and releasing C, N, Pb, Cr, Zn, and Mn, implying that the potential ecotoxic risk of Pb, Cr, Zn, and Mn exposure might be very serious. This study emphasized the effects of different one-off burial episodes on nutrient and metal cycling and mass balance in the *P. australis* marsh of the Yellow River estuary.

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