ELEMENTA
Science of the Anthropocene

Zhai, S, et al. 2021. Dynamics and Characteristics in decomposing litter. *Elem Sci Anth*, 9: 1. DOI: https://doi.org/10.1525/elementa.2021.084

RESEARCH ARTICLE

Dynamics and characteristics of biogenic silica and macro- and microelements in decomposing litter in the Min River estuary, southeast China

Shuijing Zhai[1,2,3,*], Siting Qiu[2], Hui Gao[2], and Guanyun Hou[2]

Tidal marshes are important recycling areas for biogenic silica (BSi) and macro- and microelements at the land–sea interface and are key locations for examining the decomposition process of wetland plant litter. In this study, in situ decomposition experiments were conducted with *Phragmites australis*, *Cyperus malaccensis*, and *Spartina alterniflora* in the Min River estuary wetland. Litterbags of 0.2-mm mesh size were used to evaluate the litter decomposition process and residual values of BSi and macro- and microelements, including C, N, Cr, Cu, Cd, Zn, Pb, Al, Mn, and Fe over 520 days. The litter decomposition rate significantly differed among species in the following order: *C. malaccensis* (0.005 d^{-1}) > *S. alterniflora* (0.004 d^{-1}) > *P. australis* (0.003 d^{-1}) with BSi release rates of 98.64%, 96.75%, and 97.23%, respectively. Although there were net releases of BSi, C, and N from the three litter species, continuous decrease in the BSi/(C, N) ratio indicated that BSi was removed from the litter much faster than C and N. The accumulation index results showed that Cu, Pb, Al, and Fe were net-accumulated in the litter, whereas Cd, Mn, Cr, and Zn were predominantly released during litter decay. Pearson's correlation analysis results showed that the amounts of N, Cu, Cd, Pb, Al, and Fe in the litter restrained BSi release with a significant negative correlation. These findings in the Min River estuary have important implications for geochemical cycles within wetland systems and the transport processes of potential nutrients out of the system.

Keywords: Heavy metals, Litterbags, Carbon and nitrogen, Release, Tidal marsh

1. Introduction

Many studies regarding the decomposition dynamics of plant litter have been undertaken over previous decades (Prescott, 2005, 2010), with these found to be fundamental for nutrient cycling and play a significant role in ecosystem productivity and global climate change (Liski et al., 2003; Moretto and Distel, 2003; Tuomi et al., 2009; Schaller and Struyf, 2013; Ochoa-Hueso et al., 2019). Marsh ecosystems are characterized by fast plant growth and high biomass (Struyf and Conley, 2009), with a great majority (approximately 80%) of the annual plant production becoming litter (Bouchard and Lefeuvre, 2000; Struyf et al., 2007). The decomposition of plant litter in marshes, especially in tidal marshes, affects the accumulation of organic matter and the transfer of nutrients and chemical elements, as well as influences the potential export of materials toward ambient environments (White and Howes, 1994). Although both biotic and abiotic factors controlling litter decomposition rates have been widely researched, knowledge regarding the dynamics of elements during the process is limited to only a few nutrients: C, N, and P (Osono and Takeda, 2004; Gautam et al., 2016; Gautam et al., 2019). Compared with these macroelements, the dynamics of silicon (Si) and other microelements during litter decomposition have received limited attention (Berg and Laskowski, 2005).

Si is a "beneficial" element for plants, providing structural support (Epstein, 1994; Schoelynck et al., 2010) and alleviation against biotic and abiotic stressors (Ma, 2004; Massey et al., 2006; Pavlovic et al., 2013). Wetland plants take up dissolved Si (DSi) from the soil or water to increase their rigidity and survival chances, which accumulate as biogenic silica (BSi) in plant tissues (Schoelynck et al., 2010). After vegetation dieback and subsequent litter decay, BSi is gradually released back into the abiotic environment and is usually transferred to DSi in the water (Bartoli, 1983; Struyf et al., 2007; Struyf and Conley, 2009; Vandevenne et al., 2013). Tidal marshes have been shown to accelerate the cycling process at the land–sea interface (Struyf et al., 2006). For example, the decomposition of *Phragmites australis* (*PA*) in a small freshwater

[1] School of Geographical Sciences, Fujian Normal University, Fuzhou, PR China
[2] Key Laboratory of Humid Subtropical Eco-geographical Process (Fujian Normal University), Ministry of Education, Fuzhou, PR China
[3] Fujian Provincial Key Laboratory for Subtropical Resources and Environment, Fujian Normal University, Fuzhou, PR China
* Corresponding author:
Email: s2008shuijing@163.com.

tidal marsh resulted in more than 40% of DSi export in Belgium (Struyf et al., 2007). To date, the few studies that have examined BSi content in plants being linked to litter decomposition remain controversial on decomposition rate, dynamics, and factors (Eleuterius and Lanning, 1987; Cornelissen and Thompson, 1997; Struyf et al., 2007; Schaller and Struyf, 2013; Alfredsson et al., 2016; Emsens et al., 2016). Inconclusive interactions between litter BSi and other litter quality parameters are reasons for such debate (Schoelynck et al., 2010; Cooke and Leishman, 2012; Schaller et al., 2012). Schaller and Struyf (2013) concluded that Si availability in marshes dominated by *PA* plays an important role in C sequestration, nutrient cycling, and remobilisation during aquatic litter decay. In estuarine wetlands, release by plants might affect the rate of nutrient export, especially for Si, toward the open water and eventually the oceans, thereby affecting the global nutrient cycle (Ehrenfeld et al., 2005; Borrelli et al., 2012; Struyf and Conley, 2012; Liu et al., 2019; Ochoa-Hueso et al., 2019; Xia et al., 2020).

Heavy metal pollution is common globally, especially in estuarine marshes, where large quantities of heavy metals are imported (Duarte et al., 2010), threatening food security and creating extreme risks to human health (Liu et al., 2019; Wang et al., 2020). Although it has been well-documented that accumulated metals can retard litter decomposition processes (Tyler, 1976; Laskowski et al., 1994; Johnson and Hale, 2004), it is unknown whether such metals influence BSi release by marsh litter. Additionally, knowledge regarding the coupling relationship of BSi and other macro- and microelements (such as C, N, and some heavy metals) in the decomposing litter is relatively scarce. The effect of these elements on litter decomposition is critical in estuarine marshes, which is the land–ocean transition zone, because elements released from litter fundamentally influence the coastal wetland ecosystem and oceanic red tides.

In the Min River estuarine marsh, the native species include *PA* and *Cyperus malaccensis* (*CM*), with *Spartina alterniflora* (*SA*) starting to invade this marsh in 2002 and gradually becoming the dominant species. Originally, different plant communities were distributed zonally from the land to the sea (Tong et al., 2010). In recent years, there has been a high load of total pollutants (8.6–12.9 $\times$ 10^5 tonnes) from industrial and urban sewage and mining enterprises entering the Min River basin (Li et al., 2020). This marsh is now contaminated with multiple heavy metals, mainly caused by human interferences (Hou et al., 2009). Levels of Cu, Pb, Zn, and Cd in this marsh soil are higher than those in most estuarine wetlands in China (Cai, 2011).

The objective of the present study was to explore the dynamics of BSi, C, N, and heavy metals (Cu, Cr, Zn, Cd, Pb, Al, Mn, and Fe) in three litter types of *CM*, *PA*, and *SA* using the litterbag approach during in situ experiments. Additionally, the litter BSi amount was linked to C, N, and heavy metal amount to compare their release rates and explore their mutual correlation. Several key abiotic factors and the accompanying changes in BSi and other elements were addressed to determine the effects of litter decomposition on the Min River estuary wetland ecosystem.

2. Materials and methods

2.1. Study site

The sites chosen for the present study are located in an intertidal zone of the Shanyutan marsh (119°34′12"–119°40′40" E, 26°00′36"–26°03′42" N), the largest tidal marsh (approximately 893 hm^2) in the Min River estuary, southeast China (**Figure 1**; Liu et al., 2006). The mean annual temperature is 19.6 °C and the average annual precipitation is 1,350 mm (Zheng et al., 2006). The marsh sediment in this area is dominated by saline soil, and the main vegetation species include *PA*, *CM*, and *SA* from the seawall to the sea. More detailed descriptions of the sites are provided by Tong et al. (2011) and Gao et al. (2019).

2.2. Experimental setup and sampling

The litterbag technique was used to study the litter decomposition in the experimental plots (black symbols in **Figure 1**), which were placed at the center of each plant community, from October 2014 to March 2016. In September 2014, the standing withered aboveground litter was collected from three single communities, which were *PA*-, *CM*- and *SA*-dominated. The litter was washed with tap water and then deionized water, chopped into 10-cm pieces, and oven-dried at 80 °C until it reached a constant weight. Oven-dried litter (25 g) was placed in 20 $\times$ 30 cm nylon litterbags with a mesh size of 0.2 $\times$ 0.2 mm for in situ decomposition. The 0.2 $\times$ 0.2-mm mesh size allowed free entry of soil microorganisms that might fragment the litter and prevented any significant loss of fragmented litter by macroinvertebrates. Thus, the decomposition process was mainly mediated by microflora.

On October 18, 2014, three bamboo stakes of distances between 20 and 30 m were positioned in the experimental plots of the three plant communities. Litterbags were randomly tiled at the sediment surface in the different plant communities, with only one side of the litterbag contacting the ground. The litterbags were fixed to the bamboo stakes with approximately 3 m of ropes to prevent the litterbags from being carried away by the tide. Three replicate bags were removed from each stake on the 7th, 15th, 28th, 56th, 91st, 148th, 190th, 224th, 285th, 343rd, 400th, 460th, and 520th day and immediately brought back to the laboratory. Thus, we had 3 species $\times$ 3 replicates $\times$ 13-time samplings $=$ 117 samples in total. These collected litterbags were immediately transferred to the laboratory and cleaned with slow-running tap water to remove the adhering sediments and macroinvertebrates. The litter remains in the bags were carefully removed and cleaned by hand using deionized water and oven-dried at 80 °C until a constant weight. The oven-dried samples were successively weighed and finely ground for subsequent chemistry analysis.

2.3. Sample analysis

To extract BSi, 30 mg of litter powder was digested for 5 h with Na_2CO_3 (0.1 mol/L) at 85 °C (DeMaster, 1981; Struyf et al., 2005). Dissolved BSi in the extractions was

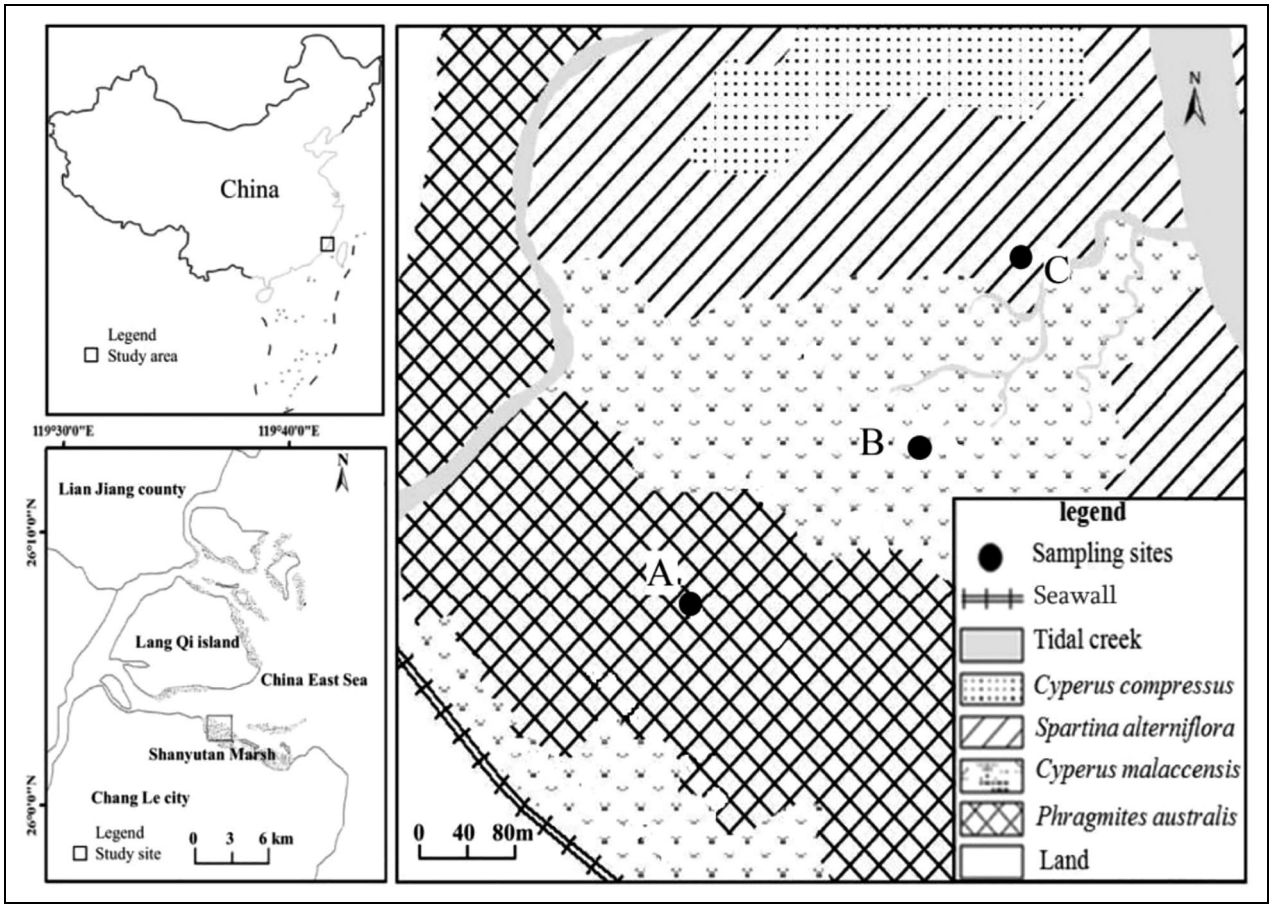

Figure 1. Location of study sites. DOI: https://doi.org/10.1525/elementa.2021.00084.f1

measured by the Molybdate blue spectrophotometric method (Saccone et al., 2006).

Total organic C and N content in the litter was measured with a Vario EL III Element Analyser (Elemental Scientific Instruments, Langenselbold, Germany).

A ground sample of 0.04 g was digested with 2 mL of HNO_3 (70%) and 2 mL of H_2O_2 (30%) at 180 °C for 10 h in closed 60-mL Teflon bottles (Savillex, Minnetonka, MN, USA). Digested samples were diluted with 1% HNO_3, and the concentration of heavy metals (Cr, Cu, Cd, Zn, Pb, Al, Mn, and Fe) in all samples was determined with inductively coupled plasma mass spectrometry (XSeriesII; Thermo Company, Waltham, MA, USA) analysis. Quality assurance and control were assessed with duplicates (three replications), method blanks, and certified reference material (GBW10020) from the National Research Center for Standards in China for each batch of samples (two blanks and one standard for every 30 samples). The temperature and conductivity of the topsoil (0–10 cm in depth) were measured with a portable instrument (Spectrum Technologies, Inc.; Chicago, IL, USA), whereas the pH of the topsoil was determined with a portable pH meter (HACH-sensION3; Loveland, CO, USA).

2.4. Calculations and statistical analysis

To correlate litter decomposition rates with measures of litter quality, litter decomposition rate (k, d^{-1}), litter release rate (P, %), and the accumulation index (AI, %) were calculated for each litter replicate using the following equations, respectively (Olson, 1963; Swift et al., 1979; Harmon et al., 1999):

$$\ln(X_t/X_0) = -k\tau$$

where X_t is the mass at time t, X_0 is the initial mass, and t is time (days).

$$P = [(X_0 \cdot C_0 - X_t \cdot C_t)/X_0 \cdot C_0] \times 100\%$$

$$\mathrm{AI}_j = (X_t \cdot C_t)/(X_0 \cdot C_0) \times 100\%$$

where C_t is the content or concentration of BSi or macro- or microelements in the litter at time t and C_0 is the initial content or concentration at the beginning of the experiment. P can reflect the loss rate of nutrients or heavy metals from the litter at the end of decomposition. AI can represent the accumulation or release status of elements in the litter. An AI value greater than 100% indicates net accumulation of BSi or macro- or microelements in the litter. A value less than 100% means net release. "j" stands for BSi, C, N, or heavy metals.

A one-way analysis of variance was performed on the original data to analyze the significant differences in decomposition rates and the amounts and AIs of BSi, C, N, and heavy metals among the three litter species. Regression analyses were used to determine the exponential relationships between the natural logarithm of the mass remains and the decomposition days. Pearson's correlation analysis was used to correlate the BSi content in

the litter with the chemical parameters (C, N, C/N ratios, and heavy metals) of the litter and the environmental indices.

3. Results

3.1. Litter mass loss

The mass loss rate of the three litter species increased during the early decomposition stage and was stable during the later stage, indicating that litter decomposition had almost stopped (**Figure 2**). After 520 days of in situ decomposition, only 14.71% *PA*, 13.63% *SA*, and 7.93% *CM* of the initial litter mass was left in the litterbags. Comparison of the decomposition process for each species revealed that *CM* exhibited the highest decomposition rate (0.005 d^{-1}), followed by *SA* (0.004 d^{-1}) and *PA* (0.003 d^{-1}) with no significant differences ($P > 0.05$; **Table 1**). Significantly higher $t_{0.95}$ (time needed for 95% of the litter dry mass to decompose) was observed for *PA* (442 days) than for *SA* (438 days) and *CM* (281 days).

3.2. Amounts and AIs of BSi, C, and N

The initial BSi content and mean C/N ratio are shown in **Table 1**. As the decomposition process proceeded, BSi, C, and N amounts in the litter decreased for all species with a rapid decline during the initial period and a slow release during the later period. The release dynamics of BSi and C were relatively steady, whereas the release dynamics of N were irregular, with a fluctuating variation during the middle period (**Figure 3**). The release dynamics significantly differed among the species for N amount ($P < 0.05$) but not for BSi and C. BSi amount in *PA* was significantly different to those of *CM* and *SA* ($P < 0.05$); however, no statistical differences were observed between *CM* and *SA* ($P > 0.05$). After 520 days of decomposition in the three species, the BSi release rate was the highest, followed by that of C and N. The BSi release rates in *PA*, *SA*, and *CM* were 97.23%, 96.75%, and 98.64%, respectively, whereas those for C were 85.05%, 87.68% and 92.85%, and those for N were 85.45%, 81.45%, and 78.71%, respectively.

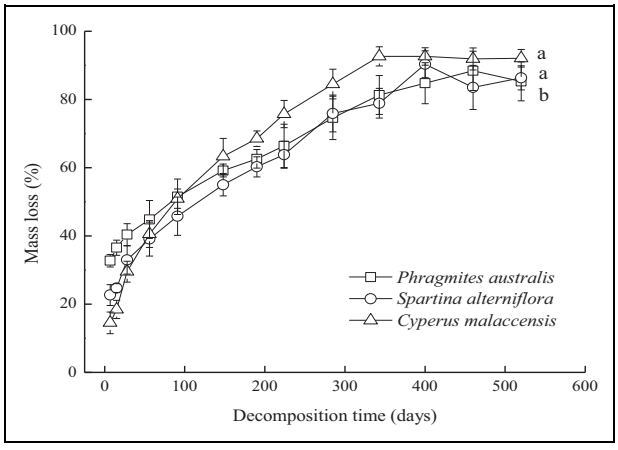

Figure 2. Mass loss rate in *Phragmites australis*, *Spartina alterniflora*, and *Cyperus malaccensis* during decomposition. Values with the same letters are not significantly different at $P > 0.05$. DOI: https://doi.org/10.1525/elementa.2021.00084.f2

The AIs of the three litter species for BSi, C, and N were all less than 100%, indicating net release throughout the 520 days of decomposition. The trend for AI_{BSi} resembled that of AI_C, with little variation during most of the experiment for the three litter species. In contrast, the trend for AI_N differed among the three litter species, with higher *CM* AIs showing that the N release rate for *CM* was lower than those of *PA* and *SA*. For C and N, toward the end of the experiment (after 400 days), AI had slightly increased except for N of *SA*, indicating that the three litter species completely released C and N during the decomposition process, but after a certain period, they began to accumulate C and N from the ambient environment. However, there was a net release of BSi from the decomposing litter in the Min River estuary marsh. For BSi and C, no significant correlations were found among the AIs of the three species ($P > 0.05$). For N, the AI of *CM* was significantly different from those of *PA* and *SA* ($P < 0.05$); however, no statistical differences were observed between *PA* and *SA* ($P > 0.05$).

3.3. Amounts and AIs of heavy metals

The changes in the amounts and AIs of heavy metals during decomposition for the three litter species showed that their dynamics were specific for each litter type (**Figures 4** and 5). The amount of most of the heavy metals first increased and then decreased, except for Mn and Cr that showed the opposite trend. After 520 days of decomposition, *PA*, *SA*, and *CM* litter had released 87.50%, 72.82%, and 76.48% of their initial Cr amount and 67.01%, 85.14%, and 84.20% of their initial Mn amount, respectively (**Table 2**). There were also some decreases in Cu (22.91%), Zn (76.44%), and Cd (87.91%) in *CM* litter and Zn (34.29%) in *PA* litter. Nevertheless, other metals in other species were characterized by an increase in metal amounts during the decomposition process (**Table 2**). For different species, *PA* litter had higher Cu, Cd, Pb, Al, and Fe amount. *SA* litter was enriched in most metals except for

Table 1. Initial biogenic silica (BSi) content, mean C/N ratio, and regression equations of mass remains ratio (W_t/W_o) with decomposition days (t) during the decomposition process. DOI: https://doi.org/10.1525/elementa.2021.00084.t1

Species	Regression Equations	k (d^{-1})	R^2	$t_{0.95}$ (Day)	Initial BSi Content (mg/g)	Mean C/N Ratio
Phragmites australis	$W_t/W_o = 0.6703e^{-0.003t}$	0.003a	0.9695	442	85.46	25.02 ± 0.68
Spartina alterniflora	$W_t/W_o = 0.7653e^{-0.004t}$	0.004a	0.9242	438	50.67	42.91 ± 1.77
Cyperus malaccensis	$W_t/W_o = 0.7975e^{-0.005t}$	0.005a	0.9414	281	45.91	25.48 ± 1.71

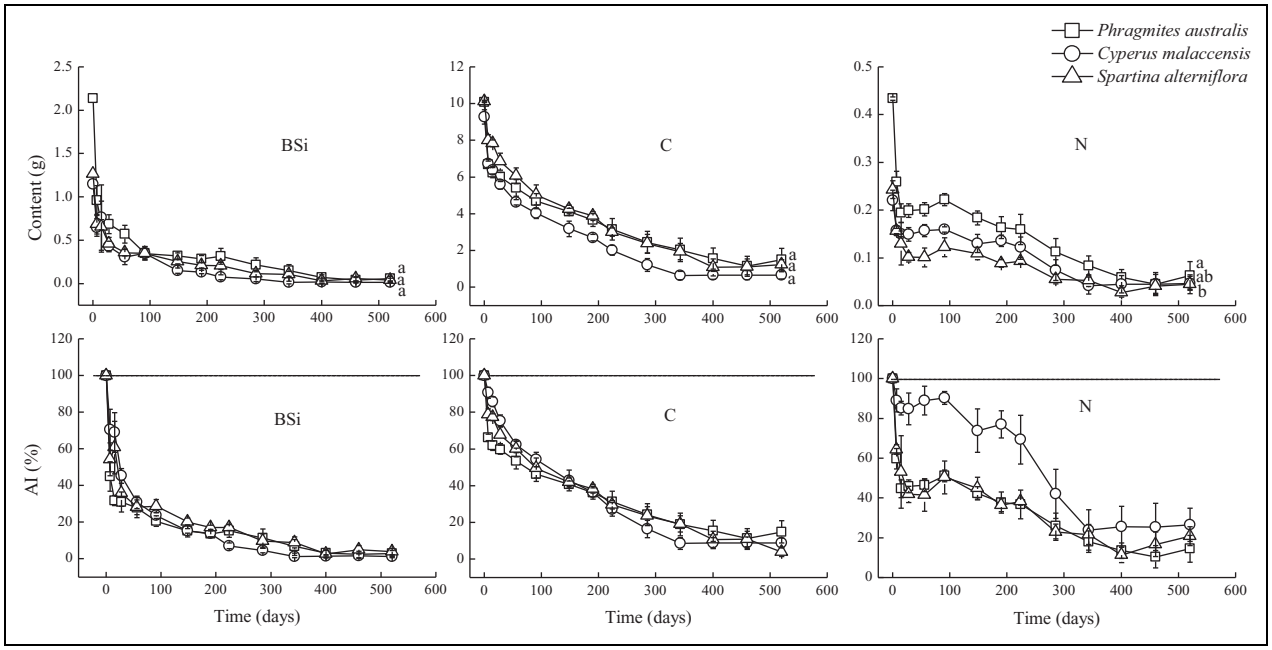

Figure 3. Nutrient amounts (per litterbag) and AIs of biogenic silica, C, and N in the decomposing litter over time for the three species. DOI: https://doi.org/10.1525/elementa.2021.00084.f3

Mn, whereas *CM* litter released most metals except for Pb, Al, and Fe.

The AIs of Cd were all lower than 100%, showing net release in the three litter species. The AI*s* of Mn were mostly lower than 100%; therefore, Mn was predominantly released by the three litter species during decomposition with some exceptions for *PA*. Higher content and AI of Cr and Zn during most early stages characterized their enriching, whereas both content and AI of Cr and Zn decreased from approximately the 250th day, indicating their releasing pattern during the later stages. Mn, Cr, and Zn were predominantly released during the decomposition of the three litter species, whereas there was net release of Cd. Thus, the AIs of the other heavy metals, including Cu, Pb, Al, and Fe, were higher than 100%, and their amounts increased, with the highest value found during the middle stage of decomposition, showing their enriching process.

4. Discussion

4.1. Decomposition rate in different litter species

Litter decomposition is an important pathway controlling nutrient cycling in wetlands (Jordan et al., 1989). Litter quality affects litter decomposition rates, and the C/N ratio is often used as a predictor of decomposition rates because it reflects the ratio of carbohydrate and lignin to protein in the litter, that is, a high ratio usually results in a low decomposition rate (Moretto and Distel, 2003). In the present decomposition experiment, the decomposition rates significantly differed among the species in the order of *CM* > *SA* > *PA*, which is consistent with Tong et al. (2011). *CM* showed the highest decomposition rate (0.005 d^{-1}) and required the shortest time (281 days) for 95% decomposition of 25 g of dry litter (**Figure 2** and **Table 1**). There were few differences between *PA* and *SA* for the decomposition rate and $t_{0.95}$. The mean C/N ratio of *CM* during decomposition was lower than that of *SA* (**Table 1**), which could account for the higher average decomposition rate of *CM*. Nevertheless, the relationship between the C/N ratio and decomposition rate is not absolute. *SA*, which had the highest C/N ratio, showed a lower decomposition rate than that of *CM* and a higher decomposition rate than that of *PA*. Inhibited decomposition is attributed to a high C/N ratio because decomposition is alleviated when N is added to the solution (Rosemond et al., 2010). High C/N ratios (greater than 30) could hamper plant decomposition because of microbes (Atkinson and Cairns, 2001). Hence, high C/N ratios (42.91 $\pm$ 1.77) and the 0.2-mm mesh size, which only allowed the entry of microorganisms, led to lower decomposition for *SA* in the present study.

PA, a herb with high lignin and cellulose content (Gessner, 2000), had the lowest decomposition rate (0.003 d^{-1}) and required the longest time (442 days) for the decomposition of 25 g dry litter in the present study (**Figure 2** and **Table 1**). Schaller and Struyf (2013) showed that reed leaf litter decay by heterotrophic microbes (as the central part of litter decay) was strongly influenced by Si availability during plant growth, with higher decay rates for the litter with higher Si content. However, this is inconsistent with the results of the present study, where plant litter with higher BSi content had lower decay rates for *PA*, *SA*, and *CM* in the Min River estuary. This can be explained by the discrepancy between the leaves and the entire plant. Additionally, litter decomposition rates among the three species had nonsignificant correlations with soil pH, conductivity, and temperature ($P > 0.05$), implying that these abiotic factors might have a negligible influence on litter decomposition rates for a long-term study in the Min River estuary.

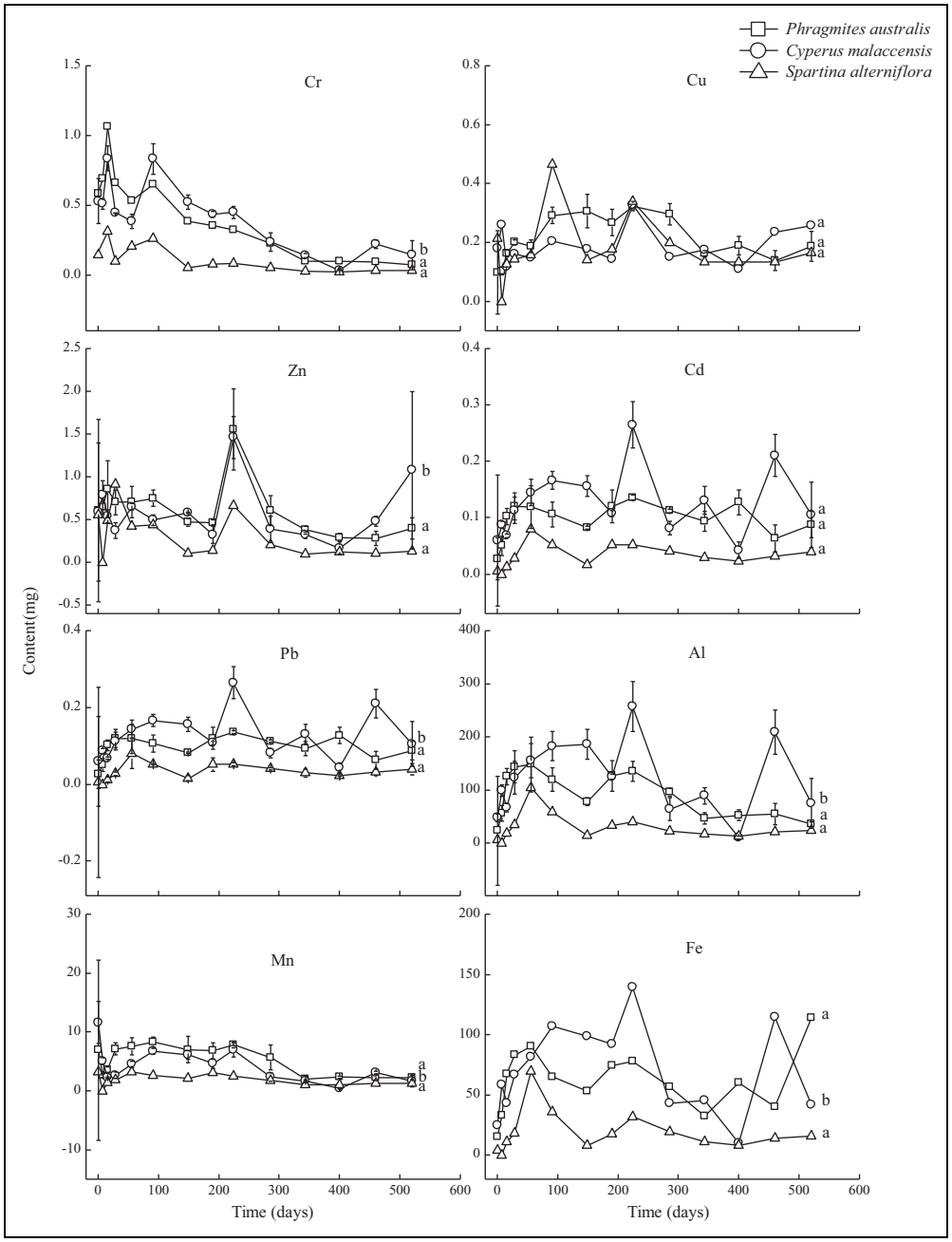

Figure 4. Amounts (per litterbag) of heavy metals in the decomposing litter over time for the three species. DOI: https://doi.org/10.1525/elementa.2021.00084.f4

4.2. Decomposition characteristics of BSi, C, and N in different litters species

Litter decomposition is one of the most important ecosystem processes that can modulate elemental biogeochemistry (Berg and Laskowski, 2005; Prescott, 2005). The release and retention of nutrients in the litter directly determine the nutritional status of the marsh ecosystem and ultimately affect the ecosystem productivity (Tong et al., 2011; Zhang et al., 2014). After vegetation dieback and subsequent litter decay, Si, C, and N are gradually released back into the ambient environment (Struyf and Conley, 2009; Vandevenne et al., 2013). However, the amount and AI of N in *PA*, *SA,* and *CM* showed an increasing trend during the early stages of in situ decomposition. N is a limiting factor for the growth of decomposer populations (Blair, 1988). Hence, many studies have stated that N dynamics in the decomposing litter are generally characterized by an initial immobilization phase followed by a release, indicating that insufficient N might limit decomposer activity during the early stages (Blair, 1988; Parton et al., 2007; Manzoni et al., 2010; Van Nevel et al., 2014).

We observed that the BSi/C and BSi/N patterns steadily decreased during decomposition, except for a few time points (**Figure 6**). Therefore, BSi was removed from the decomposing litter much faster than C and N, which supports the findings of Struyf et al. (2007). During the first 50 days, the dramatic dynamics of BSi/(C, N) might be due to interactions with abiotic and biotic factors. Thus, it is important to study plant decomposition during the initial

Table 2. Average amounts of nutrients (biogenic silica [BSi], C, and N; g) and metals (mg) in litterbags during 520 days of litter decomposition ($n = 14$); litter release rates (%) between brackets [release (–) enrichment (+)]; differences in averages between species were tested with analysis of variance. DOI: https://doi.org/10.1525/elementa.2021.00084.t2

	PA	SA	CM
	Average (Release Rate)	Average (Release Rate)	Average (Release Rate)
BSi	0.490 ± 0.146[a] (–97.23)	0.344 ± 0.091[b] (–96.75)	0.295 ± 0.092[b] (–98.64)
C	4.20 ± 0.674[a] (–85.05)	4.49 ± 0.779[a] (–87.68)	3.46 ± 0.736[b] (–92.85)
N	0.170 ± 0.027[a] (–85.45)	0.098 ± 0.015[b] (–81.45)	0.117 ± 0.015[c] (–78.71)
Cr	0.418 ± 0.078[a] (–87.50)	0.410 ± 0.064[b] (–72.82)	0.110 ± 0.025[a] (–76.48)
Cu	0.209 ± 0.020[a] (+90.48)	0.190 ± 0.016[a] (+43.54)	0.196 ± 0.026[a] (–22.91)
Zn	0.625 ± 0.085[a] (–34.29)	0.590 ± 0.089[b] (+81.60)	0.340 ± 0.071[a] (–76.44)
Cd	0.0014 ± 0.0001[a] (+608.06)	0.0013 ± 0.0001[a] (+34.61)	0.0015 ± 0.0004[a] (–87.91)
Pb	0.097 ± 0.008[a] (+219.49)	0.125 ± 0.016[b] (+76.04)	0.037 ± 0.005[c] (+166.38)
Al	89.28 ± 11.72[a] (+51.24)	121.23 ± 18.61[b] (+56.53)	32.60 ± 6.79[c] (+49.61)
Mn	5.21 ± 0.66[a] (–67.01)	4.32 ± 0.78[b] (–85.14)	2.46 ± 0.51[c] (–84.20)
Fe	61.86 ± 6.99[a] (+634.96)	69.28 ± 10.04[b] (+69.69)	20.91 ± 4.56[b] (+89.09)

Values with the same letter did not significantly differ between species ($P > 0.05$).

stage, in which large soluble nutrients are rapidly released. Schaller and Struyf (2013) found that the aquatic litter decomposition process is controlled in the tropics mainly by microbes. Microorganisms in litter can absorb nutrients from the surrounding environment to maintain their activity, which increases the decomposition of nutrients in the litter (Liao et al., 2008). In nutrient-rich environments, soil bacteria thrive and accelerate decomposition (Fierer et al., 2012). Between the 200th and 350th day, BSi/C values increased, which was attributed to more C released than BSi. For *CM*, BSi/(C, N) values were lower than those of *PA* and *SA*, thus implying *CM* released less BSi than those of the other two species.

To date, the effects of BSi content in litter on decomposition rates have remained unclear. Several researchers have suggested that high litter BSi content hampers decomposition (Cornelissen and Thompson, 1997); however, the opposite patterns have also been reported (Schaller and Struyf, 2013). In the present study, the BSi release rate and the initial BSi content were 97.23% and 85.46 mg/g in *PA*, 96.75% and 50.67 mg/g in *SA*, and 98.64% and 45.91 mg/g in *CM*, respectively (**Table 1**). Therefore, there was no relationship between the BSi decomposition rate and the initial content. Moreover, for the three species, the average release rates of C and N were 88.52% and 81.87%, respectively. BSi in *PA*, *SA*, and *CM* was released thoroughly into the Min River estuary; however, it was not the same for C and N. AIs of C, N, and BSi in the three litter species were lower than 100%; therefore, there were net releases of these three elements by *PA*, *SA*, and *CM* during the 520 days of decomposition. Litter decomposition is an important resource of BSi in wetland ecosystems, and it exports rich BSi to the estuary or sea, affecting the wetland and global nutrient cycling.

4.3. Dynamics of heavy metals in different litter species

Previous studies have indicated that the release or accumulation of metals during litter decomposition involves the following two major processes: the release of metals from the litter as they oxidize and the retention of metals on organic films of the litter (Pellenbarg, 1984; Du Laing et al., 2006). For the different heavy metals in the three marsh plants in the present study, Al and Fe were present at high levels, but other heavy metals (especially Pb and Cd) were present in minute quantities. The change in the content of heavy metals during decomposition showed that their dynamics were distinct for each litter type (**Figures 4** and **5**). The amounts of heavy metals increased in the beginning and decreased afterward, except for Mn that showed the opposite trend. An increase in the metal amount is 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). As litter decomposition proceeds, organic acids forming during the humification could create stable complexes with metals and, therefore, affect the metal solubility and speciation (Fox and Comerford, 1990). Subsequently, litter rich in humic substances might have oxygen-containing functional groups to bind heavy metals and form stable complexes (Laskowski et al., 1995). Thus, as the amount of humic substances increases during decay, presumably more metal ions can be bound to the decomposing litter (Gautam et al., 2019). In the present study, analogous results were found, in which most heavy metals began to enrich the litter during the later decomposition period, which was attributed to heavy pollution from multiple

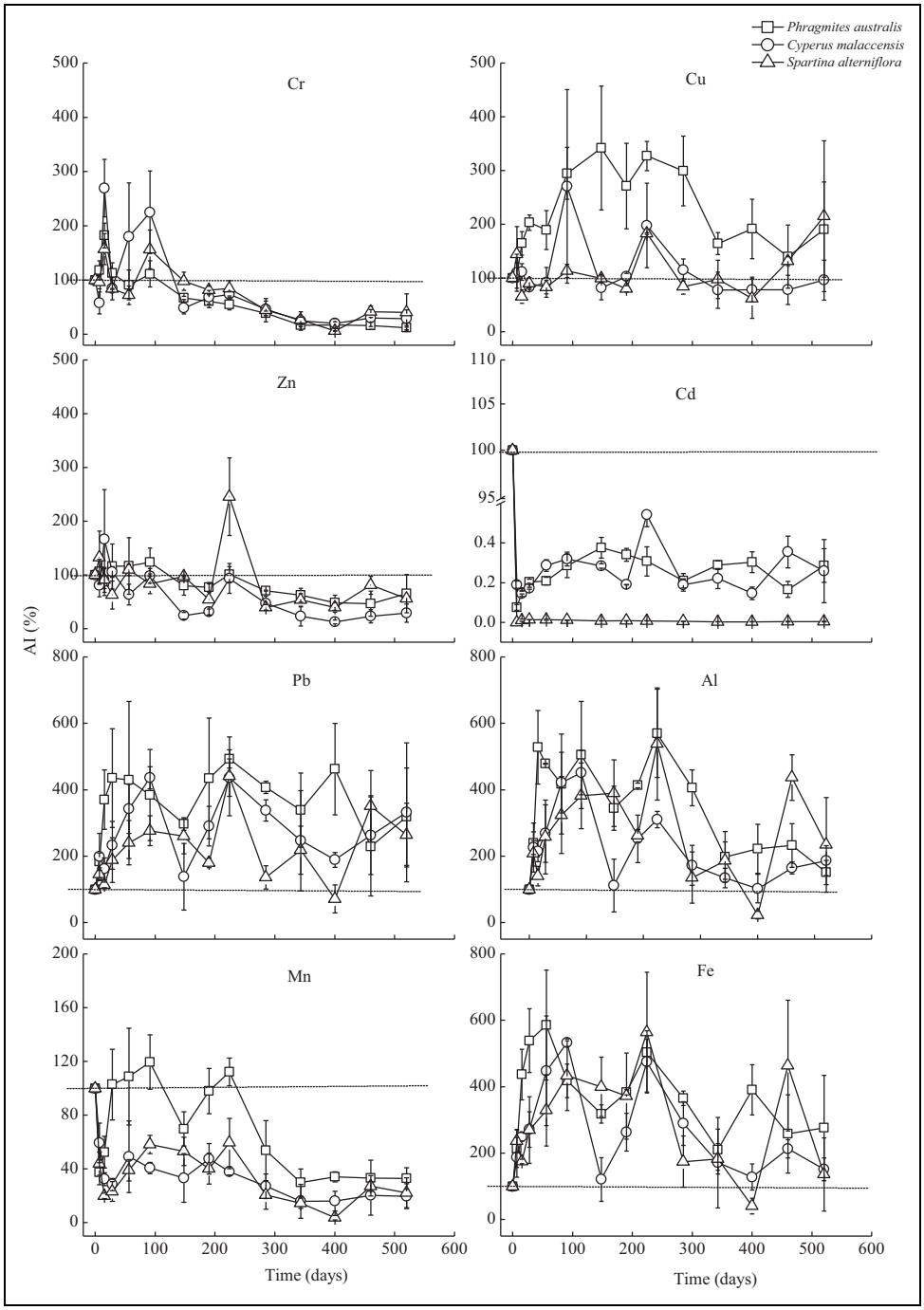

Figure 5. AIs of heavy metals in decomposing litters over time for the three litter species. DOI: https://doi.org/10.1525/elementa.2021.00084.f5

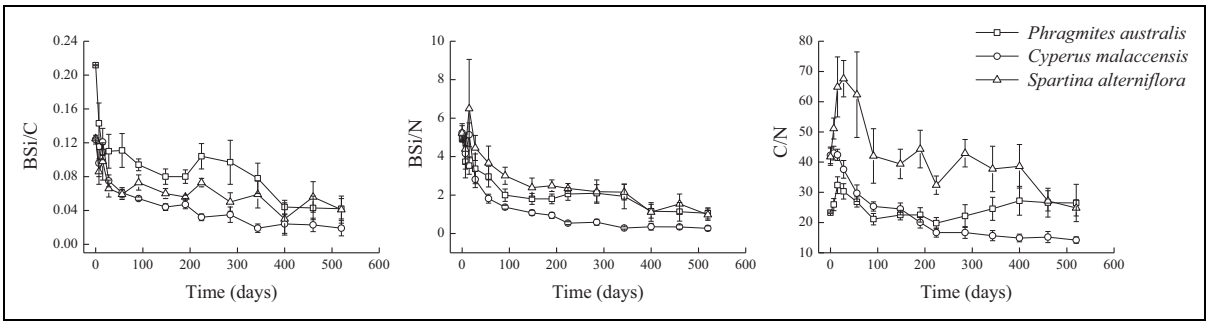

Figure 6. Ratios among biogenic silica, C, and N in the three litters during decomposition. DOI: https://doi.org/10.1525/elementa.2021.00084.f6

Table 3. Correlation between biogenic silica [BSi] and macro- and microelements of the three litter species. DOI: https://doi.org/10.1525/elementa.2021.00084.t3

Species	C	N	C/N	Cr	Cu	Zn	Cd	Pb	Al	Mn	Fe
Phragmites australis	–0.19*	–0.20*	0.43	0.54**	–0.67**	–0.16	–0.49**	–0.68**	–0.32*	–0.27	–0.62**
Spartina alterniflora	0.25**	–0.46**	0.46**	–0.29	–0.78**	–0.38*	–0.50**	–0.78**	–0.68**	–0.63**	–0.78**
Cyperus malaccensis	–0.45**	–0.78**	0.83**	0.21	–0.57**	–0.24	–0.42*	–0.56**	–0.34*	–0.22	–0.39*

Asterisks indicate significant differences between BSi and macro- and microelements.

*$P < 0.05$. **$P < 0.01$.

heavy metals due to gradually enhanced human interferences in the Min River estuary (Hou et al., 2009; Cai, 2011). The AIs of Al, Fe, Pb, and Cu were mostly greater than 100%; therefore, there was net accumulation of these metals during the 520 days of in situ decomposition. Cd, Mn, Cr, and Zn showed net release during the majority of the decomposition period. Therefore, as the plant litters decomposed, they acted differently to enrich or release heavy metals. Simultaneously, the heavy metal amount in the wetland soil or ambient surroundings affects the enrichment or release of these metals by the litters (Weis and Weis, 2004; Du Laing et al., 2006; Keuskamp et al., 2015; Li et al., 2020). Fungi can accumulate and transfer significant amounts of elements from organic and mineral soil to decomposing litters (Tyler, 2005; Van Nevel et al., 2014).

Pearson's correlation analysis results showed that the BSi amount was adversely correlated with N and some heavy metal amount in the three decomposing litter species, especially for Cu, Cd, Pb, Al, and Fe with significant correlation coefficients (**Table 3**), which persistently accumulated during the decay process except for Cd. Later in the decomposition process, the more the heavy metals such as Cu, Pb, Al, and Fe accumulated, the slower was the release of BSi that eventually stopped. Thus, we deduced that the N amount and these heavy metals in the litter restrained the BSi release. Such a finding requires further studies for confirmation. The BSi amount was also adversely correlated with the Zn and Mn amount with nonsignificant differences ($P > 0.05$). Significant negative correlation was found between the BSi and C amount for *PA* and *CM* but not for *SA*. Similarly, on comparing the start and end of experiments, it was found that a significantly lower C amount in the reed leaf litter correlated with a high BSi amount and significantly lower values for most other elements (such as N, Mn, Fe, and Cu; Schaller and Struyf, 2013). Further research is required to explore the positive correlation between the BSi and C amount for *SA*. A significant positive correlation between the BSi amount and C/N ratio for the three litter species was also observed; therefore, the high C/N ratio of the litter had a stimulative influence on BSi release by the litter.

5. Conclusions

In conclusion, the present study characterized the dynamics of BSi and other macro- and microelements, including C, N, and heavy metals, during 520 days of decomposition of *PA*, *CM*, and *SA* in the Min River estuary in southeast China. The decomposition rates significantly differed among the species in the order of *CM* (0.005 d^{-1}) > *SA* (0.004 d^{-1}) > *PA* (0.003 d^{-1}). The BSi amount generally decreased in the three litter species and significantly differed among the three species, and BSi was removed from the litter much faster than C and N. The present study clearly showed that there were net releases of BSi, C, and N by *PA*, *SA*, and *CM*, and Cd, Mn, Cr, and Zn were predominantly released, with a net accumulation of Cu, Pb, Al, and Fe during the decomposition period. Pearson's correlation analysis results showed that the BSi amount was adversely correlated with the N, Cu, Cd, Pb, Al, and Fe amount with a significant correlation. The present study revealed that the decomposition of marsh litter is an important source of BSi and has far-reaching effects on ecosystem functioning in marshes of the Min River estuary.

Data Accessibility Statement

The data generated for this study can be found at the following location: https://data.mendeley.com/datasets/h4v22c7ggv/1.

Financial Disclosure

We thank the Natural Science Foundation of China for awarding funding to SJ Zhai to run the fund-Controlling factors of spatiotemporal distributing characteristics of Silicon in Min river estuary and its mechanism (Grant No. 41401114). SJ Zhai acknowledges the funding of "Study on the characteristics and influencing factors of silicon, carbon and nitrogen during litter decomposition in Min River Estuarine Wetland" from Science and Technology Department of Fujian Province under grant agreement 2016J01184.

Competing interests

The authors have no competing interests to declare.

Author contributions

Contributed to conception and design: SJZ
Contributed to acquisition of data: GYH, HG
Contributed to analysis and interpretation of data: SJZ, STQ
Drafted and/or revised the article: SJZ, STQ
Approved the submitted version for publication: SJZ

References

Alfredsson, H, Clymans, W, Stadmark, J, Conley, D, Rousk, J. 2016. Bacterial and fungal colonization and decomposition of submerged plant litter: Consequences for biogenic silica dissolution. *FEMS Microbiology Ecology* **92**(3): 12. DOI: http://dx.doi.org/10.1093/femsec/fiw011.

Atkinson, RB, Cairns, J. 2001. Plant decomposition and litter accumulation in depressional wetlands: Functional performance of two wetland age classes that were created via excavation. *Wetlands* **21**(3): 354–362. DOI: http://dx.doi.org/10.1672/0277-5212(2001)021.

Bartoli, F. 1983. The biogeochemical cycle of silicon in two temperate forest ecosystems. *Ecological Bulletins* **35**: 469–476. DOI: http://dx.doi.org/10.2307/20112881.

Berg, B, Laskowski, R. 2005. *Litter decomposition, a guide to carbon and nutrient turnover*. New York, NY: Elsevier.

Blair, JM. 1988. Nutrient release from decomposing foliar litter of three tree species with special reference to calcium, magnesium and potassium dynamics. *Plant and Soil* **110**: 49–55. DOI:http://dx.doi.org/10.1007/bf02143538.

Borrelli, N, Osterrieth, M, Romanelli, A, Alvarez, MF, Cionchi, JL, Massone, H. 2012. Biogenic silica in wetlands and their relationship with soil and groundwater biogeochemistry in the Southeastern of Buenos Aires Province, Argentina. *Environmental Earth Sciences* **65**(2): 469–480. DOI: http://dx.doi.org/10.1007/s12665-011-1010-3.

Bouchard, V, Lefeuvre, JC. 2000. Primary production and macro-detritus dynamics in a European salt marsh: Carbon and nitrogen budgets. *Aquatic Botany* **67**(1): 23–42. DOI: http://dx.doi.org/10.1016/s0304-3770(99)00086-8.

Breteler, RJ, Teal, JM, Giblin, AE, Valiela, I. 1981. Trace element enrichments in decomposing litter of *Spartina alterniflora*. *Aquatic Botany* **11**: 111–120. DOI: http://dx.doi.org/10.1016/0304-3770(81)90052-8.

Cai, HY. 2011. Heavy metals in sediments of the Minjiang Estuary wetlands and assessment [Doctoral thesis]. Fujian Normal University. 35–41. DOI: http://dx.doi.org/10.7666/d.y1998939.

Cooke, J, Leishman, MR. 2012. Tradeoffs between foliar silicon and carbon-based defences: Evidence from vegetation communities of contrasting soil types. *Oikos* **121**(12): 2052–2060. DOI: http://dx.doi.org/10.1111/j.1600-0706.2012.20057.x.

Cornelissen, JHC, Thompson, K. 1997. Functional leaf attributes predict litter decomposition rate in herbaceous plants. *New Phytologist* **135**: 109–114. DOI: http://dx.doi.org/10.1046/ j.1469-8137.1997.00628.x.

DeMaster, DJ. 1981. The supply and accumulation of silica in the marine environment. *Geochimica et Cosmochimica Acta* **45**(10): 1715–1732. DOI: http://dx.doi.org/10.1016/ 0016-7037(81)90006-5.

Du Laing, G, Ryckegem, GV, Tack, FMG, Verloo, MG. 2006. Metal accumulation in intertidal litter through decomposing leaf blades, sheaths and stems of *Phragmites australis*. *Chemosphere* **63**(11): 1815–1823. DOI: http://dx.doi.org/10.1016/j.chemosphere.2005.10.034.

Duarte, B, Caetano, M, Almeida, PR, Vale, C, Caçador, I. 2010. Accumulation and biological cycling of heavy metal in four salt marsh species, from Tagus estuary (Portugal). *Environmental Pollution* **158**(5): 1661–1668. DOI: http://dx.doi.org/10.1016/j.envpol. 2009.12.004.

Ehrenfeld, JG, Ravit, B, Elgersma, K. 2005. Feedback in the plant soil system. *Annual Review of Environment and Resources* **30**(1): 75–115. DOI: http://dx.doi.org/10.1146/annurev.energy.30.050504.144212.

Eleuterius, LN, Lanning, FC. 1987. Silica in relation to leaf decomposition of *Juncus roemerianus*. *Journal of Coastal Research* **3**(4): 531–534.

Emsens, WJ, Schoelynck, J, Grootjans, AP, Struyf, E, Van Diggelen, R. 2016. Eutrophication alters Si cycling and litter decomposition in wetlands. *Biogeochemistry* **130**(3): 1–11. DOI: http://dx.doi.org/10.1007/s10533-016-0257-x.

Epstein, E. 1994. The anomaly of silicon in plant biology. *Proceedings of the National Academy of Sciences of the United States of America* **91**: 11–17. DOI: http://dx.doi.org/10.2307/2363729.

Fierer, N, Lauber, CL, Ramirez, KS, Zaneveld, J, Bradford, MA, Knight, R. 2012. Comparative metagenomic, phylogenetic and physiological analyses of soil microbial communities across nitrogen gradients. *The ISME Journal* **6**: 1007–1017. DOI: http://dx.doi.org/10.1038/ismej.2011.159.

Fox, TR, Comerford, NB. 1990. Low-molecular-weight organic acids in selected forest soils of the southeastern USA. *Soil Science Society of America Journal* **54**: 1139–1144. DOI: http://dx.doi.org/10.2136/sssaj1990.03615995005400040051x.

Gadd, GM. 1993. Interactions of fungi with toxic metals. *New Phytologist* **124**: 25–60. DOI: http://dx.doi.org/10.1111/j.1469-8137.1993.tb03796.x.

Gao, H, Zhai, SJ, Sun, ZG, Liu, J, Tong, C. 2019. Differences in biomass and silica content in typical plant communities with ecotones in the Min River estuary of southeast China. *PeerJ* **7**. DOI: http://dx.doi.org/10.7717/peerj.7218.

Gautam, MK, Lee, KS, Berg, B, Song, B, Yeon, J. 2019. Trends of major, minor and rare earth elements in decomposing litter in a cool temperate ecosystem, South Korea. *Chemosphere* **222**: 214–226. DOI: http://dx.doi.org/10.1016/j.chemosphere.2019.01.114.

Gautam, MK, Lee, KS, Song, BY, Lee, D, Bong, YS. 2016. Early-stage changes in natural ^{13}C and ^{15}N abundance and nutrient dynamics during different litter decomposition. *Journal of Plant Research* **129**: 463–476. DOI: http://dx.doi.org/10.1007/s10265-016-0798-z.

Gessner, MO. 2000. Breakdown and nutrient dynamics of submerged Phragmites shoots in the littoral zone of a temperate hardwater lake. *Aquatic Botany* **66**(1): 9–20. DOI: http://dx.doi.org/10.1016/S0304-3770(99)00022-4.

Harmon, ME, Nadelhoffer, KJ, Blair, JM. 1999. Measuring decomposition, nutrient turnover, and stores in plant litter, in Robertson, GP, Coleman, DC, Bledsoe, CS, Sollins, P eds., *Standard soil methods for long-term ecological research*. Oxford, UK: Oxford University Press: 202–240.

Hou, XL, Huang, JG, Liu, AQ. 2009. Heavy metals pollution and it's assessment in the wetlands of Min River Estuary in Fujian Province. *Journal of Agro-Environment Science* **28**(11): 2302–2306.

Johnson, D, Hale, B. 2004. White birch (Betula papyrifera Marshall) foliar litter decomposition in relation to trace metal atmospheric inputs at metal-contaminated and uncontaminated sites near Sudbury, Ontario and Rouyn Noranda, Quebec, Canada. *Environmental Pollution* **127**: 65–72. DOI: http://dx.doi.org/10.1016/S0269-7491(03)00260-4.

Jordan, TE, Whigham, DF, Correll, DL. 1989. The role of litter in nutrient cycling in a brackish tidal marsh. *Ecology* **70**: 1906–1915. DOI: http://dx.doi.org/10.2307/1938121.

Keuskamp, JA, Hefting, MM, Dingemans, BJ, Verhoeven, JA, Feller, IC. 2015. Effects of nutrient enrichment on mangrove leaf litter decomposition. *Science of the Total Environment* **508**: 402–410. DOI: http://dx.doi.org/10.1016/j.scitotenv.2014.11.092.

Kovacova, S, Sturdik, E. 2002. Interactions between microorganisms and heavy metals including radionuclides. *Biologia* **57**(6): 651–663. DOI: http://dx.doi.org/10.1093/ bioinformatics/18.12.1705.

Laskowski, R, Maryanski, M, Niklinska, M. 1994. Effect of heavy metals and mineral nutrients on forest litter respiration rate. *Environmental Pollution* **84**: 97–102. DOI: http://dx.doi.org/10.1016/0269-7491(94)90076-0.

Laskowski, R, Niklinska, M, Maryanski, M. 1995. The dynamics of chemical elements in forest litter. *Ecology* **76**: 1393–1406. DOI: http://dx.doi.org/10.2307/1938143.

Li, X, Sun, ZG, Tian, LP, He, T, Li, J, Wang, J, Wang, H, Chen, BB. 2020. Effects of spatial expansion between *Phragmites australis* and *Cyperus malaccensis* on variations of arsenic and heavy metals in decomposing litters in a typical subtropical estuary (Min River), China. *Chemosphere* **240**. DOI: http://dx.doi.org/10.1016/j.chemosphere.2019.124965.

Liao, CZ, Luo, YQ, Fang, CM, Chen, JK, Li, B. 2008. Litter pool sizes, decomposition, and nitrogen dynamics in Spartina alterniflora-invaded and native coastal marshlands of the Yangtze estuary. *Oecologia* **156**(3): 589–600. DOI: http://dx.doi.org/10.1007/s00442-008- 1007-0.

Liski, J, Nissinen, A, Erhard, M, Taskinen, O. 2003. Climatic effects on litter decomposition from arctic tundra to tropical rainforest. *Global Change Biology* **9**(4): 575–584. DOI: http://dx.doi.org/10.1046/j.1365-2486.2003.00605.x.

Liu, J, Luo, X, Sun, Y, Tsang, D, Qi, J, Zhang, W, Li, N, Yin, M, Wang, J, Lippold, H, Chen, Y, Sheng, G. 2019. Thallium pollution in China and removal technologies for waters: A review. *Environment International* **126**: 771–790. DOI: http://dx.doi.org/10.1016/j.envint.2019.01.076.

Liu, J, Song, ZL, Wang, JB, Bouwman, AF, Li, ML, Liu, S, Cao, L, Zang, JY, Ran, XB. 2019. Biogenic Silica Composition and Storage in the Yellow River Delta Wetland with Implications for the Carbon Preservation. *Wetland*. DOI: http://dx.doi.org/10.1007/s13157-019-01233-z.

Liu, JQ, Zeng, CS, Chen, N. 2006. *Research of Minjiang River Estuary Wetland*. Beijing, China: Science Press.

Ma, JF. 2004. Role of silicon in enhancing the resistance of plants to biotic and abiotic stresses. *Soil Science and Plant Nutrition* **50**: 11–18. DOI: http://dx.doi.org/10.1080/00380768.2004.10408447.

Manzoni, S, Trofymow, JA, Jackson, RB, Porporato, A. 2010. Stoichiometric controls on carbon, nitrogen, and phosphorus dynamics in decomposing litter. *Ecological Monographs* **80**(1): 89–106. DOI: http://dx.doi.org/10.1890/09-0179.1.

Massey, FP, Ennos, AR, Hartley, SE. 2006. Silica in grasses as a defence against insect herbivores: Contrasting effects on folivores and a phloem feeder. *Journal of Animal Ecology* **75**: 595–603. DOI: http://dx.doi.org/10.1111/j.1365-2656.2006.01082.x.

Moretto, AS, Distel, RA. 2003. Decomposition of and nutrient dynamics in leaf litter and roots of *Poa ligularis* and *Stipa gyneriodes*. *Journal of Arid Environments* **55**(3): 503–514. DOI: http://dx.doi.org/10.1016/S0140-1963(02)00271-9.

Ochoa-Hueso, R, Delgado-Baquerizo, M, An King, PT, Benham, M, Arca, V, Power, SA. 2019. Ecosystem type and resource quality are more important than global change drivers in regulating early stages of litter decomposition. *Soil Biology and Biochemistry* **717**(18): 30384–30385. DOI: http://dx.doi.org/10.1016/j.soilbio.2018.11.009.

Olson, JS. 1963. Energy storage and the balance of producers and decomposers in ecological systems. *Ecology* **44**(2): 322–331. DOI: http://dx.doi.org/10.2307/1932179.

Osono, T, Takeda, H. 2004. Potassium, calcium, and magnesium dynamics during litter decomposition in a cool temperate forest. *Journal of Forest Research* **9**(1): 23–31. DOI: http://dx.doi.org/10.1007/s10310-003-0047-x.

Parton, W, Silver, WL, Burke, IC, Grassens, L, Harmon, ME, Currie, WS, King, JY, Adair, EC, Brandt, LA, Hart, SC, Fasth, B. 2007. Global-scale similarities in nitrogen release patterns during long-term decomposition. *Science* **315**: 361–364. DOI: http://dx.doi.org/10.1126/ science.1134853.

Pavlovic, J, Samardzic, J, Maksimovic, V, Timotijevic, G, Stevic, N, Laursen, KH, Hansen, TH, Husted, S,

Schjoerring, JK, Liang, Y, Nikolic, M. 2013. Silicon alleviates iron deficiency in cucumber by promoting mobilization of iron in the root apoplast. *New Phytologist* **198**: 1096–1107. DOI: http://dx.doi.org/10.1111/nph.12213.

Pellenbarg, RE. 1984. On Spartina alterniflora litter and the trace metal biogeochemistry of a salt marsh. Estuarine. *Estuarine Coastal and Shelf Science* **18**(3): 331–346. DOI: http://dx.doi.org/10.1016/0272-7714(84)90075-1.

Prescott, CE. 2005. Decomposition and mineralization of nutrients from litter and humus, in Hormoz, B ed., *Nutrient acquisition by plants*. Berlin, Germany: Springer; 15–41.

Prescott, CE. 2010. Litter decomposition: What controls it and how can we alter it to sequester more carbon in forest soils? *Biogeochemistry* **101**(1–3): 133–149. DOI: http://dx.doi.org/10.1007/s10533-010-9439-0.

Rosemond, AD, Swan, CM, Kominoski, JS, Dye, SE. 2010. Non-additive effects of litter mixing are suppressed in a nutrient-enriched stream. *Oikos* **119**: 326–336. DOI: http://dx.doi.org/10.1111/j.1600-0706.2009.17904.x.

Saccone, L, Conley, DJ, Sauer, D. 2006. Methodologies for amorphous silica analysis. *Journal of Geochemical Exploration* **88**(1): 235–238. DOI: http://dx.doi.org/10.1016/j.gexplo.2005.08.045.

Schaller, J, Brackhage, C, Gessner, MO, Bauker, E, Dudel, EG. 2012. Silicon supply modifies C: N: P stoichiometry and growth of *Phragmites australis*. *Plant Biology* **14**(2): 392–396. DOI: http://dx.doi.org/10.1111/j.1438-8677.2011.00537.x.

Schaller, J, Struyf, E. 2013. Silicon controls microbial decay and nutrient release of grass litter during aquatic decomposition. *Hydrobiologia* **709**(1): 201–212. DOI: http://dx.doi.org/10.1007/ s10750-013-1449-1.

Schoelynck, J, Bal, K, Backx, H, Okruszko, T, Meire, P, Struyf, E. 2010. Silica uptake in aquatic and wetland macrophytes: A strategic choice between silica, lignin and cellulose? *New Phytologist* **186**: 385–391. DOI: http://dx.doi.org/10.1111/j.1469-8137.2009.03176.x.

Struyf, E, Conley, DJ. 2009. Silica: An essential nutrient in wetland biogeochemistry. *Frontiers in Ecology and the Environment* **7**(2): 88–94. DOI: http://dx.doi.org/10.1890/070126.

Struyf, E, Conley, DJ. 2012. Emerging understanding of the ecosystem silica filter. *Biogeochemistry* **107**(1–3): 9–18. DOI: http://dx.doi.org/10.1007/s10533-011-9590-2.

Struyf, E, Dausse, A, Van Damme, S, Bal, K, Gribsholt, B, Boschker, HTS, Middelburg, JJ, Meire, P. 2006. Tidal marshes and biogenic silica recycling at the land–sea interface. *Limnology and Oceanography* **51**(2): 838–846. DOI: http://dx.doi.org/10.1016/j.aquabot.2007.05.002.

Struyf, E, Van Damme, S, Gribsholt, B, Bal, K, Beauchard, O, Middelburg, JJ, Meire, P. 2007. *Phragmites australis* and silica cycling in tidal wetlands. *Aquatic Botany* **87**(2): 134–140. DOI: http://dx.doi.org/10.1016/j.aquabot.2007.05.002.

Struyf, E, Van Damme, S, Gribsholt, B, Middelbulg, JJ, Meire, P. 2005. Biogenic silica in tidal freshwater marsh sediments and vegetation (Schelde estuary, Belgium). *Marine Ecology Progress Series* **303**(1): 51–60. DOI: http://dx.doi.org/10.3354/meps303051.

Swift, MJ, Heal, OW, Anderson, JM. 1979. *Decomposition in terrestrial ecosystems*. Berkeley, CA: University of California Press.

Tong, C, Jia, RX, Wang, WQ, Zeng, CS. 2010. Spatial variations of carbon, nitrogen and phosphorous in tidal salt marsh soils of the Minjiang river estuary. *Geographical Research* **29**(7): 1203–1213. DOI: http://dx.doi.org/10.3724/SP.J.1084.2010.00199.

Tong, C, Zhang, L, Wang, W, Gauci, V, Marrs, R, Liu, B, Jia, R, Zeng, C. 2011. Contrasting nutrient stocks and litter decomposition in stands of native and invasive species in a sub-tropical estuarine marsh. *Environmental Research* **111**(7): 909–916. DOI: http://dx.doi.org/10.1016/j.envres.2011.05.023.

Tuomi, M, Thum, T, Järvinen, H, Fronzek, S, Berg, B, Harmon, M, Trofymow, JA, Sevanto, S, Liski, J. 2009. Leaf litter decomposition-estimates of global variability based on Yasso07 model. *Ecological Modelling* **220**(23): 3362–3371. DOI: http://dx.doi.org/10.1016/ j.ecolmodel.2009.05.016.

Tyler, G. 1976. Heavy metal pollution, phosphatase activity, and mineralization of organic phosphorus in forest soils. *Soil Biology and Biochemistry* **8**: 327–332. DOI: http://dx.doi.org/10.1016/0038-0717(76)90065-1.

Tyler, G. 2005. Changes in the concentrations of major, minor and rare-earth elements during leaf senescence and decomposition in a Fagus sylvatica forest. *Forest Ecology and Management* **206**(1–3): 167–177. DOI: http://dx.doi.org/10.1016/j.foreco.2004.10.065.

Vandevenne, FI, Barão, AL, Schoelynck, J, Smis, A, Ryken, N, Van Damme, S, Meire, P, Struyf, E. 2013. Grazers: Biocatalysts of terrestrial silica cycling. *Proceedings of the Royal Society B Biological Sciences* **280**(1772). DOI: http://dx.doi.org/10.1098/rspb.2013.2083.

Van Nevel, L, Mertens, J, Demey, A, Schrijver, AD, Neve, SD, Tack, FMG, Verheyen, K. 2014. Metal and nutrient dynamics in decomposing tree litter on a metal contaminated site. *Environmental Pollution* **189**: 54–62. DOI: http://dx.doi.org/10.1016/j.envpol.2014.02.009.

Wang, J, Zhou Y, Dong, X, Yin, M, Tsang, D, Sun, J, Liu, J, Song, G, Liu, Y. 2020. Temporal sedimentary record of thallium pollution in an urban lake: An emerging thallium pollution source from copper metallurgy. *Chemosphere* **242**. DOI: http://dx.doi.org/10.1016/j.chemosphere.2019.125172.

Weis, JS, Weis, P. 2004. Metal uptake, transport and release by wetland plants: Implications for phytoremediation and restoration. *Environment*

International **30**(5): 685–700. DOI: http://dx.doi.org/10.1016/j.envint.2003.11.002.

White, DS, Howes, BL. 1994. Nitrogen incorporation into decomposing litter of *Spartina alterniflora*. *Limnology and Oceanography* **39**(1): 133–140. DOI: http://dx.doi.org/10.2307/2838369.

Xia, SP, Song, ZL, Van Zwieten, L, Guo, LD, Yu, CX, Hartley, IP, Wang, HL. 2020. Silicon accumulation controls carbon cycle in wetlands through modifying nutrients stoichiometry and lignin synthesis of *Phragmites australis*. *Environmental and Experimental Botany* **175**: 1–11. DOI: http://dx.doi.org/10.1016/j.envexpbot.2020.104058.

Zawislanski, PT, Chau, S, Mountford, H, Wong, H, Sears, TC. 2001. Accumulation of Selenium and trace mtals on plant litter in a tidal marsh. *Estuarine Coastal and Shelf Science* **52**(5): 589–603. DOI: http://dx.doi.org/10.1006/ecss.2001.0772.

Zhang, LH, Tong, C, Marrs, R, Wang, TE, Zhang, WJ, Zeng, CS. 2014. Comparing litter dynamics of *Phragmites australis* and *Spartina alterniflora* in a sub-tropical Chinese estuary: Contrasts in early and late decomposition. *Aquatic Botany* **117**(5): 1–11. DOI: http://dx.doi.org/10.1016/j.aquabot.2014.03.003.

Zheng, CH, Zeng, CS, Chen, ZQ. 2006. A study on the changes of landscape pattern of estuary wetlands of the Minjiang River. *Wetland Science* **4**(1): 29–35. DOI: http://dx.doi.org/CNKI:SUN:KXSD.0.2006-01-005.

How to cite this article: Zhai, S, Qiu, S, Gao, H, Hou, G. 2020. Dynamics and characteristics of biogenic silica and macro- and microelements in decomposing litter in the Min River estuary, southeast China. *Elementa Science of the Anthropocene* 9(1). DOI: https://doi.org/10.1525/elementa.2021.00084

Domain Editor-in-Chief: Steven Allison, University of California, Irvine, CA, USA

Guest Editor: Juan Liu, Guangzhou University, Guangzhou, China

Knowledge Domain: Ecology and Earth Systems

Part of an Elementa Special Feature: Pan-Pacific Anthropocene

Published: February 08, 2021 **Accepted:** November 29, 2020 **Submitted:** June 30, 2020

ELEMENTA Science of the Anthropocene

Elem Sci Anth is a peer-reviewed open access journal published by University of California Press.

OPEN ACCESS