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The effects of litter quality and living plants on the home-field advantage of aquatic macrophyte decomposition in a eutrophic urban lake, China

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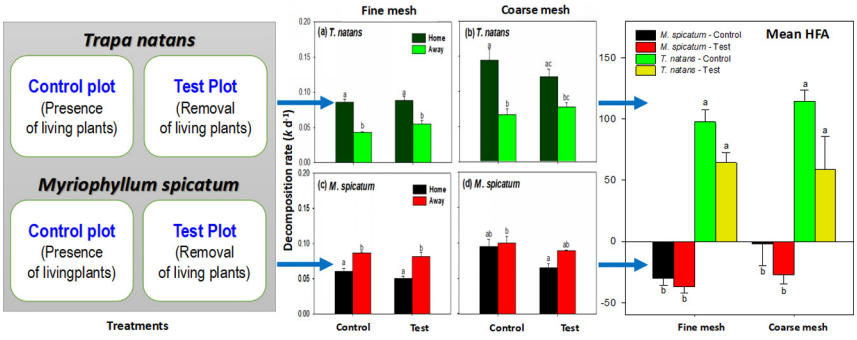
HIGHLIGHTS

- We investigated home-field advantage (HFA) effects on aquatic macrophyte decomposition in a eutrophic urban lake.
- *T. natans* showed a positive HFA effect but *M. spicatum* showed a negative one.
- HFA was strongly mediated by litter quality, habitat, and their interactions.
- The presence of living plants had inconsistent effects on decomposition rate and occurrence of HFA.

GRAPHICAL ABSTRACT



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ABSTRACT

The 'home-field advantage' (HFA) hypothesis states that litter decomposes faster in its 'home' habitat, i.e., in the same habitat as the plant species from which it was derived than it does 'away' from its home, i.e., in the habitat of a different plant species. However, studies pertaining to HFA in aquatic ecosystems are relatively few. One area not well-studied is whether the presence of living plants has an effect on the HFA of aquatic macrophyte decomposition in a eutrophic lake. Here, we conducted reciprocal litter transplanting experiments, coupled with removal of living plants, between a dominant submerged macrophyte (*Myriophyllum spicatum*) and a floating-leaved macrophyte (*Trapa natans*) in a eutrophic urban lake in China, for 50 days. Test plots were created at sites by removing the dominant macrophytes from their 'home' habitats to test the effect of living plants on decomposition rates and HFA effect. The water chemistry of the two sites was not significantly different. The initial litter qualities were significantly higher in *M. spicatum* than in *T. natans*. The decomposition rates of *T. natans* were significantly greater in both the control and test plots in its 'home' habitat, indicating a positive HFA effect, while the decomposition rates of *M. spicatum* were significantly greater in the 'away' habitat compared to its 'home' habitat in all treatments, indicating a home-field disadvantage effect. The removal of living plants had a noticeable effect on the abundance of associated-macroinvertebrates, but had an inconsistent effect on decomposition rates providing conflicting evidence for HFA. In total, 10 macroinvertebrate taxa from four functional feeding groups (FFGs) were collected during the experiment. Compared to macroinvertebrate communities, microbial activities showed less correlation with decomposition rates. Our results provide evidence to suggest that decomposition-based HFA is dependent upon litter quality, habitat, and their interactions in a eutrophic urban lake.

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1. Introduction

Decomposition of organic matter is a key process, facilitating the recycling of nutrients and chemical elements, sustaining important food chains, and primary production in aquatic ecosystems (Gessner et al., 2010). In aquatic ecosystems, plant breakdown and decomposition has been affected not only by internal factors such as chemical characteristics of their leaves (Canhoto and Graça, 1996) but also by external factors such as composition of the decomposer community (fungi, bacteria, and invertebrates) (Bardgett and van der Putten, 2014) and plant community structure (Hobbie, 1992; Ward et al., 2015). However, the effect of living plants on decomposition is less studied.

In eutrophic lake ecosystems, the enriched nutrient content can create favorable habitats for aquatic macrophyte growth (Rejmánková and Houdková, 2006). As a result, macrophytes usually spread rapidly in this kind of ecosystem. Macrophyte decomposition is important not only for the development of lakes but also for many other ecological processes involved in nutrient-enriched lake ecosystems (Debusk and Reddy, 2005). Despite widespread concerns regarding litter decomposition in freshwater ecosystems, few studies have explored the factors regulating decomposition of aquatic macrophytes in eutrophic lakes (Li et al., 2012). Therefore, it is important to understand the biological processes involved in the decomposition of aquatic macrophytes and nutrient cycling in shallow eutrophic lakes (Carvalho et al., 2015).

Previous studies have shown that litter decomposition occurs at a faster rate in the 'home' habitat of the plant species from which the litter is derived, than in a habitat 'away' from its 'home', which has been termed home-field advantage' (HFA) (Gholz et al., 2000). Decomposer communities generally prefer to decompose a continual input of similar litter quality, which could increase the specificity of decomposers for particular plant species and its associated habitat, regardless of chemical composition (Hunt et al., 1988; Gholz et al., 2000). Presently, there is no overwhelming evidence to suggest that HFA occurs commonly and consistently in terrestrial and aquatic decomposition. Some studies have provided evidence to support HFA effects in decomposer communities of terrestrial ecosystems (Ayres et al., 2009; Veen et al., 2015a; Li et al., 2017) and aquatic ecosystems (Jackrel and Wootton, 2014; Franzitta et al., 2015; Leroy et al., 2017). By contrast, several studies have found no support for this phenomenon either in terrestrial ecosystems (Gießelmann et al., 2011; Veen et al., 2015b) or aquatic ecosystems (Fenoy et al., 2016), or have reported variable results (Chomel et al., 2015; Jewell et al., 2015). Moreover, some studies have even provided evidence of a home-field disadvantage (St. John et al., 2011; Wang et al., 2015). Therefore, it is still a controversial phenomenon requiring clarification of its occurrence and underlying mechanisms.

Litter quality is a determinant factor for decomposition rate. The original hypothesis of HFA, proposed by Hunt et al. (1988), focused mainly on its relation to microbial preference for 'home' litter and plant litter quality was not discussed (Palozzi and Lindo, 2017). A modified HFA hypothesis including reference to litter quality was suggested by Milcu and Manning (2011) and Veen et al. (2015b). They suggested that low-quality litters, which contain recalcitrant compounds, may facilitate HFA because only specialized decomposers can utilize these recalcitrant compounds. By contrast, high quality litter contains easily degradable compounds; therefore, most decomposers can exploit habitats with high quality litter. Thus, such habitats are expected to have a lower potential for HFA (Ayres et al., 2009; Austin et al., 2014; Chomel et al., 2015). To date, most of the studies on HFA pertaining to litter quality have been conducted in terrestrial ecosystems. In aquatic ecosystems, particularly in a eutrophic urban lake, the decomposition of aquatic macrophytes has long been studied with different parameters of decomposition rates. Nevertheless, investigations on HFA among aquatic macrophytes from different life-forms with different litter qualities are relatively few despite its importance.

Living plants can modify the decomposition process indirectly through changes in understory environmental conditions, soil biota community composition (Scherer-Lorenzen, 2008), and modification of the decomposition environment (Barantal et al., 2011). In aquatic ecosystems, living aquatic macrophytes serve as habitat, refuge, shelter, and feeding sites for invertebrates (Xiao et al., 2006; Mormul et al., 2010). Therefore, it is anticipated that the presence of living plants enhances the abundance and distribution of invertebrate communities and this would in turn affect the decomposition of aquatic macrophytes. Despite the general assumption that living plants influence decomposition rates, the effect it may have on decomposition of aquatic macrophytes is not well-studied (Eisenhauer et al., 2013).

Here, we explore the effects of litter quality and decomposer community composition on HFA regarding decomposition rates in habitats dominated by either submerged (*Myriophyllum spicatum* L.) or floating-leaved (*Trapa natans* L.) communities in a eutrophic urban lake in China. In addition to HFA measurement, we investigated how living plants affect the composition and activity of decomposer communities by removing living plants from their 'home' habitat and observing whether the resulting effect had any subsequent effect on HFA of aquatic macrophyte decomposition. Specifically, we hypothesized that (1) HFA would be observed for litter decomposition in *M. spicatum* and *T. natans* species; (2) HFA would be more pronounced in low quality litter with recalcitrant compounds; and (3) the decomposition rates and the magnitude of HFA would be higher at the plots with living plants than without living plants.

2. Materials and methods

2.1. Study area

The study was conducted in Donghu (East Lake) (30°33.118′N; 114°21.198′E), located in Wuhan city, Hubei province, China. The Lake is the largest 'City Lake' in China and covers an area of 88 km^2 (33 km^2 of water area). The lake was cut off from the Yangtze River in 1957. Water quality has deteriorated due to the discharge of 180,000 tons of wastewater (domestic and industrial sewage) into the lake per day. Each year, approximately 441 tons of nitrogen and 40 tons of phosphorus are input into the lake. Therefore, most parts of the lake have undergone eutrophication (total N 148.57 mmol m^{-3}, total P 9.03 mmol m^{-3}), and are unsuitable for drinking or recreation. The average depth of the lake basin is 2.21 m and the maximum is 4.75 m. The lake belongs to a warm, humid, northern subtropical climate. The average surface water temperature is 18.9 °C and average rainfall is about 1160.3 mm (Yang et al., 1999).

2.2. Experimental design

To test HFA effect of macrophyte decomposition, two experimental sites dominated by Eurasian water milfoil (*M. spicatum*; submerged macrophyte) and water caltrop (also known as water chestnut) (*T. natans*; floating-leaved macrophyte) were selected in Donghu Lake. The two experimental sites were at least 100 m apart from each other. In each site, a test plot was created by removing the dominant living macrophytes to investigate the effect of living plants on decomposition rate. Therefore, the study consisted of a control plot with living plants (hereafter 'control' plot) and a test plot with the removal of the living plants (hereafter 'test' plot); altogether four plots for the two sites (two 'control' plots and two 'test' plots) were established. The macrophytes were removed from each site during August 2017.

The collected plant material was cleaned with water to exclude any unwanted material; then oven-dried at 60 °C to a constant dry weight. Two kinds of 15 cm × 20 cm nylon litterbags were used: litterbags of 5 mm mesh size (coarse-mesh), to allow the colonization of macroinvertebrates, and litterbags of 0.5 mm mesh size (fine-mesh), for the exclusion of invertebrates >0.5 mm. Litterbags were filled with 10 g (±0.05) dry weight of material from each species, and three replicates were prepared for each type comprising of 240 litterbags in total

(two species × two mesh sizes × four treatments × five sampling times × three replicates). Each litterbag was coded with plastic labels to indicate plant species, site, treatment type, mesh size, and replicate number. Five pairs of litterbags (a pair from a single species with coarse and fine mesh litterbags) were tied on a nylon rope by using nylon thread about 1 m apart from each other. The nylon ropes were used to ease retrieval of litterbags. There were 12 nylon ropes with five pairs of litterbags for each species (altogether 24 ropes for two species). On September 1, 2017, the nylon ropes with pairs of litterbags were deployed reciprocally at the control and test plots of each site and the ropes were anchored on a stone at the shoreline. Weights were attached to each pair of litterbags so that they remained on the surface of the sediment.

A pair of litterbags from each plot of each site was retrieved every 10 days over the 50-days study period without disturbing other samples. In total, 48 litterbags were collected at each sampling time and the litterbags were placed, individually, in labeled and sterilized polythene bags and brought back to the laboratory in cool boxes with ice-bags. The litterbags were refrigerated at 4 °C until processing. Any unwanted contaminants collected along with the litterbags, such as sediment soil and debris, were carefully removed. The remaining material was oven-dried at 65 °C to a constant dry weight for determination of the remaining dry mass (%), which was used to calculate decomposition rates.

2.3. Litter quality and water chemistry analyses

The oven-dried initial and remaining materials were ground and sieved through 0.25 mm mesh for subsequent chemical analysis. The carbon content (C%) in the material was determined by rapid wet oxidation-titration with $K_2Cr_2O_7$ digestion (Nelson and Sommers, 1982); nitrogen (N%) and phosphorous (P%) by using a $H_2SO_4 \cdot H_2O_2$ digestion method (Shi, 1994). The content of N% was determined by indigo blue colorimetry and P% by molybdenum antimony colorimetric assay; lignin, hemicellulose and fiber by proximate analysis via an acid detergent fiber (ADF) method (Rowland and Roberts, 1994); polyphenolic substances using Folin-Ciocalteu phenol reagent.

Water chemistry (water temperature, pH, DO, NH_4-N, NO_3-N, and TP) from each plot of the two sites was measured at each collection time during the experiment. However, the results were not considered for the determination of the decomposition process because there were no significant differences between the plots of each site.

2.4. Macroinvertebrate extraction and microbial activity determination

In the laboratory, the materials were taken out of the litterbags and the macroinvertebrates were carefully extracted with forceps, and preserved in 70% ethanol for further taxonomic identification, enumeration and biomass assessment. By using a stereo-microscope at a magnification of at least 8×, macroinvertebrates were identified to the lowest possible taxonomic level. Abundance was expressed as number of individuals per litterbag (no. ind.·bag^{-1}), taxa richness was expressed as number of taxa per litterbag (no. taxa·bag^{-1}), and biomass was expressed as grams per litterbag (g·bag^{-1}). All individuals were assigned to one of the following functional feeding groups (FFGs): shredders, scrapers, gathering-collectors, and predators according to Merritt et al. (2008). The material from each litterbag was weighted to ~5.0 g (wet weight) to test microbial respiration rate. The microbial respiration rate was measured after incubation (24 h) at 22.5 °C, using an alkali absorption in a closed chamber method (Wollum, 1982).

2.5. Data analyses

Mass remaining was expressed as a percentage of the total initial dry mass. The litter decomposition rate was determined by using the exponential equation as per Olson (1963), which is as follows:

$$X_t = \times_0 \times e^{-kt},$$

where X_t is the litter mass remaining (g) at time 't' (day), X_0 is the initial mass of the litter, and 'k' is the litter decomposition rate constant (day^{-1}).

The HFA index (hereafter HFAI) for litter reciprocal transplant decomposition in the habitat of submerged and floating-leaved plants was calculated in two steps based on relationships according to Ayres et al. (2009). First, the relative mass loss was calculated as

$$ARMLa = \frac{Aa}{Aa + Ba} \times 100$$

where $ARMLa$ represents the relative mass loss of the litter from species A at habitat a, and Aa and Ba represent the percentage (of initial) litter mass loss of plant species A and B decomposing at habitat a.

The HFAI was then calculated as

$$HFAI = \frac{ARMLa + BRMLb}{ARMLb + BRMLa} \times 100 - 100$$

HFAI stands for the additional decomposition at 'home' versus 'away' habitat and is a net value for both species (A and B) in the reciprocal transplant.

The mean HFA (% increase in k value at 'home' compared with the 'away' habitat) for each litter type was calculated based on the work of Austin et al. (2014):

$$\text{Mean HFA} = (k_{\text{home}} - k_{\text{away}})/k_{\text{away}} \times 100$$

where k_{home} and k_{away} are the decomposition constants of a given species at 'home' and in 'away' habitats, respectively.

All statistical tests were performed using SPSS (Version 20.0). Data were checked for deviations from normality and homogeneity of variance before analysis. ANOVAs and Tukey's tests were applied to assess significant differences between the various treatments. Multi-way ANOVAs were used to examine effects of litter species (*M. spicatum* and *T. natans*), habitat (home and away), sampling time (every 10 days over the 50-days study period) and treatment (control and test) on dry weight remaining percentage, macroinvertebrate abundance and richness, and microbial respiration rate. Independent sample t-tests were used to compare initial litter qualities of the species and the differences between decomposition rates (k day^{-1}) from control and test plots of 'home' and 'away' habitats. One-way ANOVAs were used to test the significant difference between HFAI and the water chemistry.

3. Results

3.1. Initial litter quality and changes during decomposition

There were significant differences between the initial litter quality characteristics of *M. spicatum* and *T. natans* (Table 1). The content of N % was significantly higher in *M. spicatum* than that of *T. natans*, while *T. natans* showed higher C%, P%, lignin%, lignin:N, total phenol %, cellulose%, and hemicellulose% than *M. spicatum*. Moreover, *M. spicatum* showed higher ratios of C:P and N:P than that of *T. natans*. The comparison of means of C:N, C:P, N:P, and lignin:N ratios were also significantly different between the two species (Table 1).

The removal of living plants from its 'home' habitat did not have significant effect on the remaining percentage of C and N ($p > 0.05$) (Figs. S1, S2). In contrast, it had a significant effect on the remaining P % in *M. spicatum* species from the 'away' habitat in fine-mesh litterbags ($t = 2.666$, $df = 28$ $p = 0.013$). For *T. natans*, the remaining P% decreased during the first 10 days of the experiment from both habitats and treatments. There were no differences on remaining P% between

Table 1
Initial litter quality characteristics of *M. spicatum* and *T. natans* from Independent sample *t*-test (mean ± SE, n = 3).

	M. spicatum	*T. natans*	df	*t*	*p*
C (%)	32.03 ± 1.04	41.02 ± 1.6	4	−6.053	**0.004**
N (%)	20.75 ± 0.44	2.85 ± 0.17	4	37.838	<**0.001**
P (%)	0.26 ± 0.004	2.63 ± 0.03	4	−74.698	<**0.001**
C:N	1.55 ± 0.06	15.94 ± 0.76	4	−18.873	<**0.001**
C:P	121.93 ± 5.52	15.58 ± 0.50	4	19.175	<**0.001**
N:P	78.94 ± 2.21	16.24 ± 0.96	4	35.262	<**0.001**
Lignin (%)	0.32 ± 0.003	4.73 ± 1.37	4	−4.374	**0.012**
Lignin:N	0.02 ± 0.0003	1.72 ± 0.46	4	−3.661	**0.022**
Total phenol (%)	2.41 ± 0.03	10.72 ± 0.11	4	−41.651	<**0.001**
Cellulose (%)	0.03 ± 0.002	6.37 ± 1.38	4	−15.768	<**0.001**
Hemicellulose (%)	0.03 ± 0.001	9.86 ± 1.03	4	−17.729	<**0.001**

Significant differences are indicated by bold-face *p* values.

the habitats and plots from the two mesh sizes; therefore, the curves became flatter till the end of the experiment (Fig. S3). Pearson's correlation analysis showed that the remaining percentage of C, N and P was significantly positively correlated with the decomposition rates of both species from all treatments (Table S1).

3.2. Dry weight remaining percentage, decomposition rates and HFA

The dry weight remaining percentage of both *M. spicatum* and *T. natans* decreased rapidly during the first 10 days of the experiment (>50%) which is possibly attributed to the leaching, and slowly declined throughout the rest of the whole experiment in all treatments from both habitats (Fig. 1). Dry weight remaining percentage was significantly affected by habitats ($F_{1,200} = 5.537$, $p = 0.020$), sampling time ($F_{4,200} = 114.952$, $p < 0.001$), and their interactions (Table 2).

In fine-mesh litterbags, the decomposition rates of *T. natans* from control plots were 0.086 day^{-1} at 'home' and 0.043 day^{-1} in 'away' habitat; 0.089 day^{-1} at 'home' and 0.055 day^{-1} in 'away' habitat in test plots. In coarse-mesh litterbags, 0.144 day^{-1} at 'home' and 0.067 day^{-1} in 'away' habitat from control plots; 0.120 day^{-1} at 'home' and 0.078 day^{-1} in 'away' habitat in test plots. In fine-mesh litterbags of *M. spicatum*, the decomposition rates from control plots were 0.060 day^{-1} at 'home' and 0.086 day^{-1} in 'away' habitat; 0.051 day^{-1} at 'home' habitat and 0.081 day^{-1} in 'away' habitat in test plots. In coarse-mesh litterbags, 0.095 day^{-1} at 'home' and 0.100 day^{-1} in 'away' habitat from control plots; 0.065 day^{-1} at 'home' and 0.090 day^{-1} in 'away' habitat in test plots.

The decomposition rates of *T. natans* were significantly faster at 'home' rather than in the 'away' habitats in both fine-mesh and coarse-mesh litterbags from both control and test plots (fine-mesh: $t = 8.267$, $df = 4$, $p = 0.001$ and $t = 4.266$, $df = 4$, $p = 0.013$; coarse-mesh: $t = 4.292$, $df = 4$, $p = 0.013$ and $t = 3.299$, $df = 4$, $p = 0.030$). For *M. spicatum*, however, the decomposition rates in fine-mesh litterbags were significantly faster in the 'away' habitat compared to 'home' from both control and test plots ($t = -5.077$, $df = 4$, $p = 0.007$ and $t = -4.647$, $df = 4$, $p = 0.010$) and in coarse-mesh litterbags from the test plot ($t = -4.013$, $df = 4$, $p = 0.016$), but not from the control plot ($t = -0.386$, $df = 4$, $p = 0.719$). The decomposition rate at the 'home' habitat of *T. natans* was 3.4% faster at the test plot in fine-mesh and 16.7% faster at the control plot in coarse-mesh litterbags. Meanwhile, for *M. spicatum*, the decomposition rate at the 'away' habitat was faster by 15% in fine-mesh and 31.6% in coarse-mesh litterbags that at the control plot with living plants.

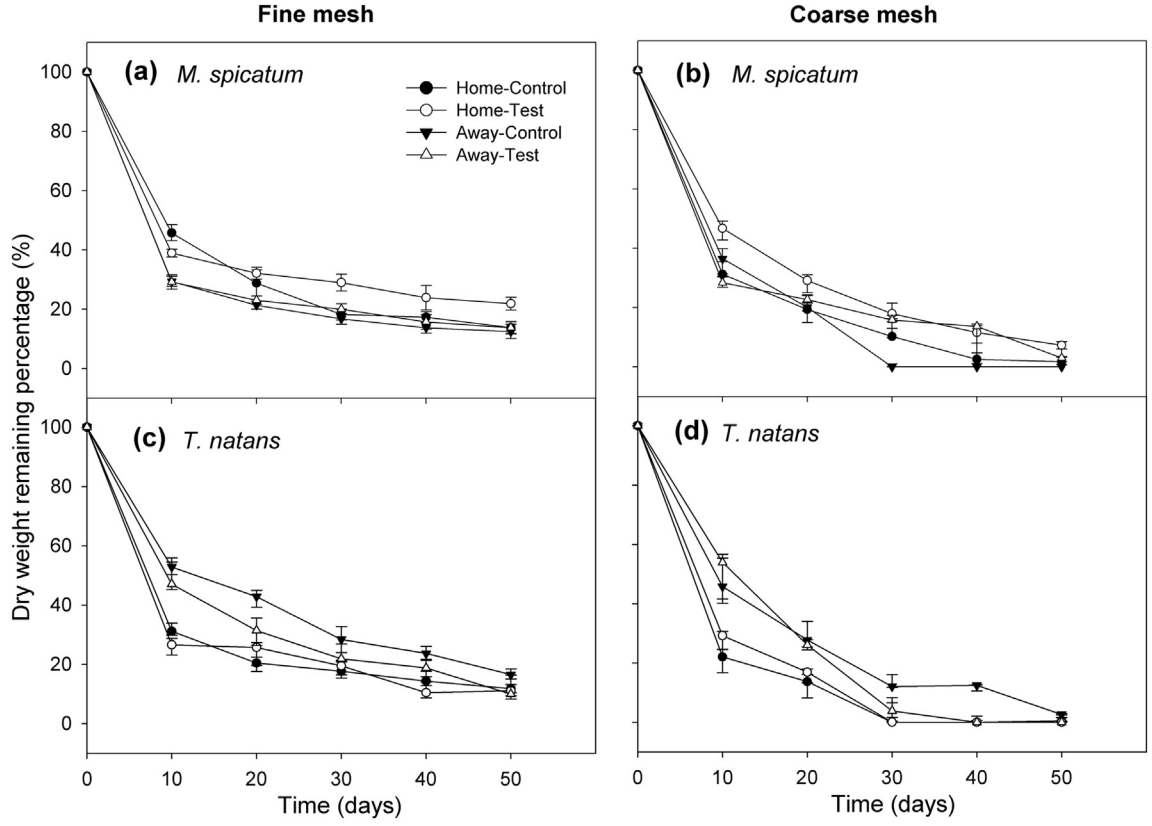

Fig. 1. Changes in dry weight remaining percentage (%) of *M. spicatum* and *T. natans* litter during 50 days of litter decomposition in fine-mesh and coarse-mesh litterbags at 'home' and 'away' habitats from control and test plots (removal of living plants treatments). Error bars indicate standard error (n = 3).

Table 2
Multi-way ANOVAs: effects of litter species, habitat, sampling time, treatment and their interactions on the dry weight remaining (%), invertebrate abundance and richness, and microbial respiration rate after 50 days of decomposition.

Source of variation		Dry weight remaining (%)		Macroinvertebrate abundance		Macroinvertebrate richness		Microbial respiration rate	
	df	F	*p*	F	*p*	F	*p*	F	*p*
Species (S)	1	0.403	0.526	0.819	0.367	1.439	0.232	0.829	0.364
Habitat (H)	1	5.537	**0.020**	0.040	0.841	0.058	0.811	0.643	0.423
Time (T)	4	114.952	<**0.001**	10.047	<**0.001**	7.122	<**0.001**	32.127	<**0.001**
Treatment (Tr)	1	1.115	0.292	11.348	<**0.001**	0.129	0.719	0.538	0.464
S × H	1	69.422	<**0.001**	0.228	0.633	2.072	0.152	3.405	0.066
S × T	4	0.901	0.464	2.064	0.087	0.737	0.567	0.897	0.467
S × Tr	1	14.096	<**0.001**	0.051	0.821	1.165	0.282	1.220	0.271
H × T	4	2.070	0.086	1.145	0.336	2.036	0.091	0.230	0.922
H × Tr	1	4.433	**0.036**	12.924	<**0.001**	1.165	0.282	0.495	0.482
T × Tr	4	0.124	0.974	1.472	0.212	0.237	0.917	0.697	0.595
S × H × T	4	6.324	<**0.001**	1.738	0.143	1.335	0.258	0.842	0.500
S × H × Tr	1	0.396	0.530	0.569	0.452	0.129	0.719	0.294	0.588
S × T × Tr	4	1.496	0.205	0.648	0.629	0.788	0.534	0.560	0.692
H × T × Tr	4	0.185	0.946	1.829	0.125	0.266	0.899	0.336	0.853
S × H × T × Tr	4	0.367	0.832	0.703	0.590	0.975	0.422	0.314	0.868

Significant differences are indicated by bold-face *p* values. 'df' values are the 'degree of freedom' for the factors.

Decomposition rates were significantly different between habitats, mesh sizes and species as well as the interaction between habitat and mesh sizes, habitat and species, habitat and treatment, and mesh size and treatment (Table 2). Overall, the decomposition rates of the two macrophytes were greater in the coarse-mesh litterbags than in the fine-mesh litterbags at the end of the experiment (Tukey test, $p < 0.001$).

The HFAI of litter decomposition for the two species in both the control and test plots were positive in both mesh sizes except for fine-mesh litterbags from the control plot (Fig. 2). HFAI was not significantly different between plots and mesh sizes ($F_{3,56} = 1.192$, $p = 0.321$); however, the highest HFAI (30.20) was observed in the coarse-mesh litterbags from the control plot. The mean HFA of *M. spicatum* litter in the fine- and coarse-mesh litterbags from both treatments was negative (Fig. 3). Conversely, *T. natans* litter showed a positive HFA effect on decomposition at both mesh sizes and treatments. Furthermore, *T. natans* showed the highest value of mean HFA (114.49) in coarse-mesh litterbags at the control plot compared to its 'home' habitat (Fig. 3).

3.3. Macroinvertebrates and microbial association

Altogether 1407 macroinvertebrates from 10 taxa and four FFGs were collected (Table S2, Fig. 4). The macroinvertebrate abundance was significantly affected by sampling time ($F_{4,200} = 10.047$, $p < 0.001$), treatments ($F_{1,200} = 11.348$, $p < 0.001$) and the interaction between habitats and treatments ($F_{1,200} = 12.924$, $p < 0.001$) (Table 2). Taxa richness was significantly affected only by sampling time ($F_{4,200} = 7.122$, $p < 0.001$). Pearson's correlation analysis indicated that the decomposition rates of both *M. spicatum* and *T. natans* were significantly negatively correlated with the abundance of macroinvertebrates in their 'home' habitats from the fine-mesh litterbags of the test plots (Table S3). In comparison with fine-mesh, the decomposition rate of coarse-mesh litterbags from the test plot of *T. natans* at its 'home' habitat was significantly negatively correlated with abundance of macroinvertebrates ($r = -0.598$, $p = 0.019$) (Table S3).

Microbial respiration rate was significantly different only between mesh sizes ($F_{4,200} = 32.127$, $p < 0.001$) (Table 2). Pearson's correlation analysis showed that microbial respiration rate was significantly

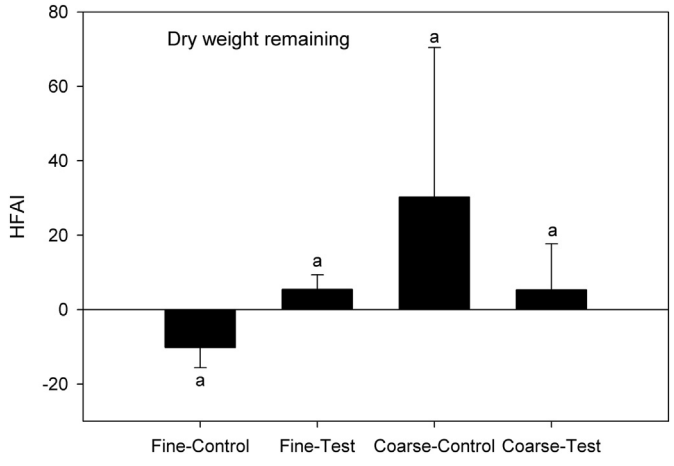

Fig. 2. Parameter estimates (mean ± SE, n = 3) calculated for HFA index (HFAI). Bars with different letters are significantly different at $p < 0.05$. Error bars indicate standard error. (Fine: fine-mesh litterbag; coarse: coarse-mesh litterbag; control: plot with living plants; test: plot with removal of living plants.)

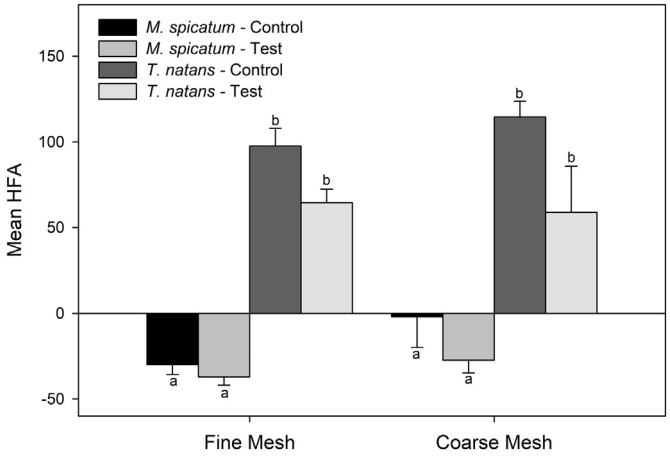

Fig. 3. The mean home-field advantage (mean HFA) for *M. spicatum* and *T. natans* litter mass loss in fine- and coarse-mesh litterbags in control and test plots. Error bars indicate standard error (n = 3). Bars with the different letter are significantly different at $p < 0.05$. (Control: plot with living plants; test: plot with removal of living plants.)

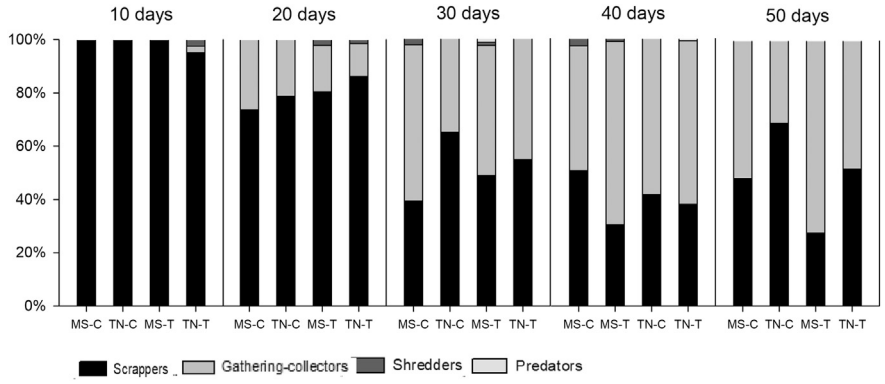

Fig. 4. Abundance of macroinvertebrate functional feeding groups present in litterbags from control and test plots of *M. spicatum* and *T. natans* habitats during 50 days of litter decomposition. MS: *M. spicatum* habitat; TN: *T. natans* habitat; C: control plot with living plants; and T: test plot with removal of living plants.

positively correlated with decomposition rates from coarse-mesh litterbags of *M. spicatum* from the control plot ($r = 0.525$, $p = 0.044$) and that of *T. natans* from the test plot ($r = 0.606$, $p = 0.017$) at their 'home' habitats, fine-mesh litterbags at the control plots ($r = 0.517$, $p = 0.048$) and coarse-mesh litterbags of *M. spicatum* at the test plots from the 'away' habitat ($r = 0.533$, $p = 0.041$) (Table S3).

4. Discussion

4.1. Decomposition rate and HFA

Although our experiment was carried out under the same abiotic conditions (water chemistry of the two sites), we found evidence for HFA only for the decomposition rate of *T. natans*, which is partially in line with our first hypothesis. The decomposition rate of *T. natans* was twice as fast at its 'home' compared to its 'away' habitat in all treatments, while that of *M. spicatum* was significantly higher in its 'away' habitat compared to its 'home' habitat in all treatments, indicating a home-field disadvantage. Among a handful of studies on HFA in aquatic ecosystem, there were contentious results on its occurrence and magnitude. For instance, when comparing the decomposition rates of *Phragmites australis* and *Fucus vesiculosus*, only the macrophyte *P. australis* decomposed significantly faster in the place where they occur naturally, providing evidence for an HFA effect for *P. australis* (Lopes et al., 2013). Franzitta et al. (2015) demonstrated HFA for detritus decomposition along estuarine gradients. The decomposition rates of *Quercus* in 'freshwater' and *Fucus* in 'high salinity' conditions showed that more rapid decomposition rates occurred at their 'home' habitats, which corroborated the occurrence of HFA in aquatic ecosystems. Jackrel and Wootton (2014) showed that the decomposition rate of locally derived red alder leaf litter was 24% faster than red alder leaf litter introduced from other riparian zones in four different rivers. By contrast, Fenoy et al. (2016) provided limited support for HFA in lowland and mountain streams between *Phragmites* and *Alnus* species because the decomposition rates of both species were faster in lowland than in mountain streams. Considering these results, the occurrence of HFA between and among aquatic macrophytes may be species-specific and mediated by habitat circumstances.

4.2. Litter quality effect on HFA

Litter quality is one of the most important factors of decomposition rates of aquatic macrophytes. In general, high-quality (low C:N and lignin:N) litter is more easily decomposed, and so most decomposers can break down these compounds regardless of the habitat. By contrast, low-quality, recalcitrant (high C:N and lignin:N) litter is difficult to decompose and therefore, only a few specialized species can decompose these compounds (Strickland et al., 2009). In this study, the litter quality of *T. natans* was significantly lower, with recalcitrant compounds, than that of *M. spicatum*. Our results showed that low quality *T. natans* litter decomposed significantly faster in its 'home' habitat compared to its 'away' habitat, indicating the presence of a HFA effect for *T. natans* which agrees with our second hypothesis. Most previous studies have suggested that there is a stronger HFA effect in more recalcitrant litter (Ayres et al., 2009; Milcu and Manning, 2011), which is in line with our findings. According to the HFA hypothesis (Hunt et al., 1988; Gholz et al., 2000), it may be a possible consequence that the decomposer communities from the habitat of *T. natans* are specialized in decomposing the litter that they always encounter, which results in faster decomposition rate for *T. natans* litter in its 'home' habitat than 'away'. Veen et al. (2015a) showed in their meta-analysis that the magnitude of HFA became stronger (regardless of the direction) when the quality of 'home' and 'away' litter became more dissimilar. Furthermore, home-field effects were determined by the degree of difference between the types of dominant plant species in the 'home' versus 'away' communities. In this study, we observed similar results in that there was evidence for a pronounced HFA effect for *T. natans* because the litter quality of *T. natans* was significantly lower than that of *M. spicatum*, and the two species were different life-form macrophytes displaying different physical characteristics (floating-leaved and submerged plants).

Our results showed that the decomposition rate of *M. spicatum* was significantly higher in 'away' habitat compared to its 'home' habitat in all treatments, indicating a home-field disadvantage. Many previous studies have reported the absence of HFA in terrestrial ecosystems (Gießelmann et al., 2011; St. John et al., 2011; Veen et al., 2015b) and some studies have also reported the occurrence of home-field disadvantage (Wang et al., 2015). On the other hand, Leroy et al. (2017) showed that soft and less recalcitrant Melastomataceae leaves decomposed much faster in their 'home' (open area), providing evidence for the occurrence of HFA in high quality litter in aquatic ecosystem using an experiment conducted in water-filled tanks with bromeliads from open and forested areas. Similar findings have also been reported in other studies (Freschet et al., 2012). In contrast to these findings, our results showed that the decomposition rates of high-quality *M. spicatum* were significantly higher in low-quality *T. natans* habitats than its 'home'. This unexpected phenomenon is probably attributed to the effect of decomposer communities that associate with *T. natans* species. Functional breadth (FB) hypothesis (Strickland et al., 2009) stated that decomposer communities from the habitat of recalcitrant litter have wider functional abilities than those originating from nutrient rich environments. These results, alongside our own results, suggest that the functional breadth hypothesis may help explain the observed HFA results revealing that high-quality *M. spicatum* litter could be decomposed more efficiently by the decomposer communities from habitats of low-quality *T. natans* than by those from high quality *M. spicatum*. Therefore, the decomposition rate of *M. spicatum* was significantly higher in the 'away'

habitat compared to its 'home' habitat. Overall, we found a clear and significant pattern regarding litter quality for the control of HFA in a eutrophic urban lake.

4.3. Living plants, decomposition rates and HFA effect

In contrast with our third hypothesis, living plants seem to have an inconsistent effect on the decomposition rate of the two species investigated and HFA did not differ significantly between the treatments. We found that the decomposition rate of *M. spicatum* was slightly faster in the control plots with living plants than the test plots while that of *T. natans* was faster in the test plots without living plants. Therefore, the presence of living plants partially enhanced only the decomposition rate of *M. spicatum*. Our results for *T. natans* are consistent with the previous findings of Ward et al. (2015), which showed a higher decomposition rate of litter by the removal of shrubs from the plant community. Meanwhile, the results of *M. spicatum* in our study are in line with the findings of van der Krift et al. (2002), who showed a significant stimulation of the decomposition rate of dead root litter by the presence of the living roots. These findings demonstrated that in addition to litter quality, changes in vegetation composition may play a partial role in regulating decomposition rates.

There is clear evidence that living plants serve as habitat, refuge, shelter, and feeding sites for aquatic invertebrates in aquatic ecosystems (Xiao et al., 2006; Mormul et al., 2010). In our study, we also found that the higher abundance of macroinvertebrates occurred in the litterbags from the test plots than the control plots, suggesting that the removal of living plants may lead to the loss of habitat and feeding sites. As a consequence, macroinvertebrates readily rely on the litterbags they encountered for their feeding source and habitat. Meanwhile, the correlations of microbial respiration rates with litter decomposition rates were more significant in the control plots than the test plots (Table S3). This phenomenon might be possible while the removal of living plants accelerated the accumulation of macroinvertebrates in the litterbags from test plots, the microorganisms are still abundant at their original habitat without disturbance (at the control plots).

Microbial communities are fundamentally important in the decomposition process in aquatic ecosystems (Gulis and Suberkropp, 2003). Additionally, the influence of invertebrates in the decomposition rates of aquatic macrophytes should also be linked to microorganisms (Lopes et al., 2013). In addition, it has been suggested that microbial decomposers specialize in the decomposition of certain substrates (Gießelmann et al., 2011) and that the specificity and diversity of a microbial decomposer community between 'home' and 'away' habitats should have led to the presence of HFA (Prescott and Grayston, 2013). However, our findings contradicted these suggestions that decomposer communities might not be the main drivers of decomposition rates and HFA in a eutrophic urban lake. Taken together, although the presence of living plants significantly affected the associated-macroinvertebrate abundance, it did not strongly influence the decomposition rate of plant litter, which indicated a limited role for living plants as a driver of aquatic macrophyte decomposition rates and the occurrence of HFA in a eutrophic urban lake.

5. Conclusion

Our study showed that HFA does occur in aquatic ecosystems in certain circumstances, in that we found a positive HFA effect (twice as fast) of *T. natans* in all treatments but a negative HFA effect for *M. spicatum*. Litter quality and habitat, as well as their interactions have significant effect on the decomposition rate and the occurrence of HFA with a stronger influence of habitat effect compared to litter quality. Litter quality, in conjunction with habitat and mesh-size might be a determinant factor for controlling macrophyte decomposition in an urban lake. The removal of living plants from their 'home' habitat greatly influenced the associated-macroinvertebrate abundance but did not significantly affect decomposition rates and HFA. Overall, the present study sheds light on the HFA hypothesis. Further studies are needed to investigate litter origins and qualities, aquatic invertebrate colonization potentials and the interaction mechanisms underlying HFA in freshwater ecosystems.

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References

Austin, A.T., Vivanco, L., González-Arzac, A., Pérez, L.I., 2014. There's no place like 'home'? An exploration of the mechanisms behind plant litter-decomposer affinity in terrestrial ecosystems. New Phytol. 204, 307–314.

Ayres, E., Steltzer, H., Simmons, B.L., Simpson, R.T., Steinweg, J.M., Wallenstein, M.D., Mellor, N., Parton, W.J., Moore, J.C., Wall, D.H., 2009. Home-field advantage accelerates leaf litter decomposition in forests. Soil Biol. Biochem. 41, 606–610.

Barantal, S., Roy, J., Fromin, N., Schimann, H., Hättenschwiler, S., 2011. Long-term presence of tree species but not chemical diversity affect litter mixture effects on decomposition in a neotropical rainforest. Oecologia 167, 241–252.

Bardgett, R.D., van der Putten, W.H., 2014. Belowground biodiversity and ecosystem functioning. Nature 515, 505–511.

Canhoto, C., Graça, M.A.S., 1996. Decomposition of *Eucalyptus globulus* leaves and three native leaf species (*Alnus glutinosa*, *Castanea sativa* and *Quercus faginea*) in a Portuguese low order stream. Hydrobiologia 333, 79–85.

Carvalho, C., Hepp, L.U., Palma-Silva, C., Albertoni, E.F., 2015. Decomposition of macrophytes in a shallow subtropical lake. Limnologica 53, 1–9.

Chomel, M., Guittonny-Larchevêque, M., DesRochers, A., Baldy, V., 2015. Home field advantage of litter decomposition in pure and mixed plantations under boreal climate. Ecosystems 18, 1014–1028.

Debusk, W.F., Reddy, K.R., 2005. Litter decomposition and nutrient dynamics in a phosphorus enriched everglades mars. Biogeochemistry 75, 217–240.

van der Krift, T., Kuikman, P., Berendse, F., 2002. The effect of living plants on root decomposition of four grass species. Oikos 96, 36–45.

Eisenhauer, N., Dobies, T., Cesarz, S., Hobbie, S.E., Meyer, R.J., Worm, K., Reich, P.B., 2013. Plant diversity effects on soil food webs are stronger than those of elevated CO_2 and N deposition in a long-term grassland experiment. Proc. Natl. Acad. Sci. U. S. A. 110, 6889–6894.

Fenoy, E., Casas, J.J., Díaz-López, M., Rubio, J., Guil-Guerrero, J.L., Moyano-López, F.J., 2016. Temperature and substrate chemistry as major drivers of interregional variability of leaf microbial decomposition and cellulolytic activity in headwater streams. FEMS Microbiol. Ecol. 92, fiw169.

Franzitta, G., Hanley, M.E., Airoldi, L., Baggini, C., Bilton, D.T., Rundle, S.D., Thompson, R.C., 2015. Home advantage? Decomposition across the freshwater-estuarine transition zone varies with litter origin and local salinity. Mar. Environ. Res. 110, 1–7.

Freschet, G.T., Aerts, R., Cornelissen, J.H.C., 2012. Multiple mechanisms for trait effects on litter decomposition: moving beyond home-field advantage with a new hypothesis. J. Ecol. 100, 619–630.

Gessner, M.O., Swan, C.M., Dang, C.K., McKie, B.G., Bardgett, R.D., Wall, D.H., Hättenschwiler, S., 2010. Diversity meets decomposition. Trends Ecol. Evol. 25, 372–380.

Gholz, H.L., Wedin, D.A., Smitherman, S.M., Harmon, M.E., Parton, W.J., 2000. Long-term dynamics of pine and hardwood litter in contrasting environments: toward a global model of decomposition. Glob. Chang. Biol. 6, 751–765.

Gießelmann, U.C., Martins, K.G., Brändle, M., Schädler, M., Marques, R., Brandl, R., 2011. Lack of home-field advantage in the decomposition of leaf litter in the Atlantic Rainforest of Brazil. Appl. Soil Ecol. 49, 5–10.

Gulis, V., Suberkropp, K., 2003. Interactions between stream fungi and bacteria associated with decomposing leaf litter and different levels of nutrient availability. Aquat. Microb. Ecol. 30, 149–157.

Hobbie, S.E., 1992. Effects of plant species on nutrient cycling. Trends Ecol. Evol. 7, 336–339.

Hunt, H.W., Ingham, E.R., Coleman, D.C., et al., 1988. Nitrogen limitation of production and decomposition in prairie, mountain meadow, and pine forest. Ecology 69, 1009–1016.

Jackrel, S.L., Wootton, J.T., 2014. Local adaptation of stream communities to intraspecific variation in a terrestrial ecosystem subsidy. Ecology 95, 37–43.

Jewell, M.D., Shipley, B., Paquette, A., Messier, C., Reich, P.B., 2015. A traits-based test of the home-field advantage in mixed-species tree litter decomposition. Ann. Bot. 116, 781–788.

Leroy, C., Corbara, B., Dézerald, O., Trzcinski, M.K., Carria, J.-F., Dejean, A., Céréghino, R., 2017. What drives detrital decomposition in neotropical tank bromeliads? Hydrobiologia 802, 85–95.
Li, X., Cui, B., Yang, Q., Tian, H., Lan, Y., et al., 2012. Detritus quality controls macrophyte decomposition under different nutrient concentrations in a eutrophic shallow lake, North China. PLoS One 7 (7), e42042.
Li, Y.B., Li, Q., Yang, J.J., Lü, X.T., Liang, W.J., Han, X.G., Bezemer, T.M., 2017. Home-field advantages of litter decomposition increase with increasing N deposition rates: a litter and soil perspective. Funct. Ecol. 31, 1792–1801.
Lopes, M.L., Martins, P., Rodrigues, A.M., Quintino, V., 2013. Macroinvertebrates communities associated with the decomposition of *Phragmites australis* and *Fucus vesiculosus* in transitional systems. J. Sea Res. 83, 72–82.
Merritt, R.W., Cummins, K.W., Berg, M.B., 2008. An Introduction to the Aquatic Insects of North America. 4th Edition. Kendall/Hunt Publishing Co., Dubuque.
Milcu, A., Manning, P., 2011. All size classes of soil fauna and litter quality control the acceleration of litter decay in its home environment. Oikos 120, 1366–1370.
Mormul, R.P., Thomaz, S.M., Silveira, M.J., Rodrigues, L., 2010. Epiphyton or macrophyte: which primary producer attracts the snail *Hebetancylus moricandi*? Am. Malacol. Bull. 133, 127–133.
Nelson, D.W., Sommers, L.E., 1982. Part 2. Total carbon, organic carbon, and organic matter. Methods of Soil Analysis. American Society of Agronomy, Madison, WI, USA, pp. 539–579.
Olson, J.S., 1963. Energy storage and the balance of producers and decomposers in ecological systems. Ecology 44, 322–331.
Palozzi, J.E., Lindo, Z., 2017. Pure and mixed litters of *Sphagnum* and *Carex* exhibit a home-field advantage in Boreal peatlands. Soil Biol. Biochem. 115, 161–168.
Prescott, C.E., Grayston, S.J., 2013. Tree species influence on microbial communities in litter and soil: current knowledge and research needs. For. Ecol. Manag. 309, 19–27.
Rejmánková, E., Houdková, K., 2006. Wetland plant decomposition under different nutrient conditions: what is more important, litter quality or site quality? Biogeochemistry 80, 245–262.
Rowland, A.P., Roberts, J.D., 1994. Lignin and cellulose fractionation in decomposition studies using acid-detergent fibre methods. Commun. Soil Sci. Plan. 25, 269–277.
Scherer-Lorenzen, M., 2008. Functional diversity affects decomposition processes in experimental grasslands. Funct. Ecol. 22, 547–555.
Shi, R.H., 1994. Agricultural and Chemical Analysis for Soil. Science Press, Beijing, pp. 212–218 (in Chinese).
St. John, M.G., Orwin, K.H., Dickie, I.A., 2011. No 'home' versus 'away' effects of decomposition found in a grassland–forest reciprocal litter transplant study. Soil Biol. Biochem. 43, 1482–1489.
Strickland, M.S., Osburn, E., Lauber, C., Fierer, N., Bradford, M.A., 2009. Litter quality is in the eye of the beholder: initial decomposition rates as a function of inoculum characteristics. Funct. Ecol. 23, 627–636.
Veen, G.F., Freschet, G.T., Ordonez, A., Wardle, D.A., 2015a. Litter quality and environmental controls of home-field advantage effects on litter decomposition. Oikos 124, 187–195.
Veen, G.F., Sundqvist, M.K., Wardle, D.A., Briones, M.J., 2015b. Environmental factors and traits that drive plant litter decomposition do not determine home-field advantage effects. Funct. Ecol. 29, 981–991.
Wang, J., You, Y., Tang, Z., Liu, S., Sun, O.J., 2015. Variations in leaf litter decomposition across contrasting forest stands and controlling factors at local scale. J. Plant Ecol. 8, 261–272.
Ward, S.E., Orwin, K.H., Ostle, N.J., Briones, M.J.I., Thomson, B.C., Griffiths, R.I., Oakley, S., Quirk, H., Bardgett, R.D., 2015. Vegetation exerts a greater control on litter decomposition than climate warming in peatlands. Ecology 96, 113–123.
Wollum II, A.G., 1982. Cultural methods for soil microorganisms. Methods of soil analysis. Part 2. Chemical and microbiological properties. Agron. Monogr. 9 (2), 781–802.
Xiao, K.Y., Yu, D., Wang, J.W., 2006. Habitat selection in spatially heterogeneous environments: a test of foraging behavior in the clonal submerged macrophyte *Vallisneria spiralis* L. Freshw. Biol. 51, 1552–1559.
Yang, Y.F., Huang, X.F., Liu, J.K., 1999. Long-term changes in crustacean zooplankton and water quality in a shallow, eutrophic Chinese lake densely stocked with fish. Hydrobiologia 391, 195–203.