



Leaf Nutrients and Macroinvertebrates Control Litter Mixing Effects on Decomposition in Temperate Streams

Mathieu Santonja, Hector Rodriguez-Perez, Nathalie Le Bris, Christophe Piscart

► To cite this version:

Mathieu Santonja, Hector Rodriguez-Perez, Nathalie Le Bris, Christophe Piscart. Leaf Nutrients and Macroinvertebrates Control Litter Mixing Effects on Decomposition in Temperate Streams. *Ecosystems*, 2020, 23 (2), pp.400-416. 10.1007/s10021-019-00410-9 . hal-02165628

HAL Id: hal-02165628

<https://amu.hal.science/hal-02165628>

Submitted on 9 Jul 2019

HAL is a multi-disciplinary open access archive for the deposit and dissemination of scientific research documents, whether they are published or not. The documents may come from teaching and research institutions in France or abroad, or from public or private research centers.

L'archive ouverte pluridisciplinaire **HAL**, est destinée au dépôt et à la diffusion de documents scientifiques de niveau recherche, publiés ou non, émanant des établissements d'enseignement et de recherche français ou étrangers, des laboratoires publics ou privés.

TITLE: Leaf nutrients and macroinvertebrates control litter mixing effects on decomposition in temperate streams

RUNNING HEAD: Litter mixture decomposition in temperate streams

AUTHORS: Mathieu Santonja^{1,2*}, Héctor Rodríguez-Pérez¹, Nathalie Le Bris¹, Christophe Piscart¹

ADDRESSES

1. Univ Rennes, CNRS, ECOBIO UMR 6553, F-35000 Rennes, France.

2. Aix Marseille Univ, Avignon Université, CNRS, IRD, IMBE, Marseille, France.

***CORRESPONDING AUTHOR**

Email: mathieu.santonja@gmail.com

ORCID: 0000-0002-6322-6352

ABSTRACT

Plant litter decomposition is an essential ecosystem function in temperate streams. Both riparian vegetation and decomposer communities are major determinants of the decomposition efficiency and the interactions occurring within litter mixtures. However, the extent to which such litter mixture interactions are affected by combined shifts in litter traits and decomposer community is not well understood. We used leaf litter from 10 European tree species in order to study litter decomposition and litter mixture effects occurring in two-species litter mixtures in a temperate forested stream of northwestern France. The study distinguished between (i) decomposition involving microorganisms alone or together with invertebrates, and (ii) decomposition involving litter mixtures of similar or dissimilar nutrient content. Increasing mean litter nutrient concentration favored both microbial activity and litter decomposition rate. Surprisingly, the highest litter mixture effects occurred in mixtures containing two nutrient-rich litters and occurred mainly in macroinvertebrate presence. Both the “mass-ratio hypothesis”, expressed as the community weighted mean traits ($Trait_{CWM}$), and the “niche complementarity hypothesis”, expressed as the functional dissimilarity of litter traits ($Trait_{FD}$), contributed to explain litter mixture effects. However, $Trait_{CWM}$ was found to be a better predictor than $Trait_{FD}$. Finally, when evaluating the individual contributions of litter nutrients, calcium and magnesium appeared as important drivers of litter mixture effects. Our findings suggest that the mass-ratio hypothesis overrules the niche complementarity hypothesis as a driver of litter diversity effects. Our study highlights the key importance of macroinvertebrates and of leaf nutrients, such as Ca and Mg, which are often neglected in decomposition studies in streams.

KEYWORDS: Biodiversity-ecosystem functioning; community weighted means; functional trait dissimilarity; litter traits; litter decomposition; litter nutrients; temperate stream

41 **HIGHLIGHTS**

- 42 - Macroinvertebrates and leaf nutrients drive microbial activity and litter decomposition
- 43 rate
- 44 - Macroinvertebrate presence and nutrient concentration control litter mixing effect
- 45 intensity
- 46 - Mean nutrient concentration is more important than concentration dissimilarity within
- 47 litter mixture

48

INTRODUCTION

Litter decomposition is an essential ecosystem function controlling the carbon (C) and nutrient cycles in both terrestrial (Swift and others 1979; Cadish and Giller 1997) and aquatic (Cummins 1974; Wallace and others 1997) ecosystems. Litter decomposition rate is jointly affected by the litter traits (e.g. Cornwell and others 2008; Garcia-Palacios and others 2016), the environmental conditions, such as the temperature (e.g. Fierer and others 2005; Follstad Shah and others 2017), and the decomposer communities (e.g. Hättenschwiler and others 2015; Gessner and others 2010; Jabiol and others 2013; Santonja and others 2018a). Rates of litter decomposition are essentially controlled by litter chemistry (Cornwell and others 2008; Garcia-Palacios and others 2016). In addition to secondary compounds (e.g. tannins) and fiber components (e.g. lignin), concentrations of nutrients such as nitrogen (N) and phosphorus (P) are useful litter traits for the prediction of the decomposition rates of single-species litter (Gessner and Chauvet 1994; Cornwell and others 2008; Schindler and Gessner 2009). Nevertheless, as mainly reported in terrestrial ecosystems, other nutrients such as calcium (Ca), magnesium (Mg), potassium (K), sodium (Na), and sulfur (S), can also affect the decomposition efficiency (Kaspari and others 2008; Makkonen and others 2012; Garcia-Palacios and others 2016). However, the potential importance of these nutrients is often neglected in decomposition studies in streams compared to N and P.

In the majority of natural ecosystems, litter material from different plant species decomposes together. Given that forested stream food webs are dependent upon allochthonous leaf litter (Vannote and others 1980; Wallace and others 1997; Gessner and others 1999), understanding the relationship between riparian tree species composition (and its associated litter traits) and the decomposition rates in streams draining forested watersheds is of considerable ecological importance (Swan and Palmer 2004; Kominoski and others 2007). Changing plant species composition can modify litter trait control over decomposition in two

ways. First, according to the “mass-ratio hypothesis” (Grime 1998; Garnier and others 2004), litter trait control over decomposition changes along community-weighted mean (CWM) trait values (Fig. 1a, Quested and others 2007; Mokany and others 2008; Laughlin 2011). Second, according to the “niche complementarity hypothesis” (Petchey and Gaston 2006; Diaz and others 2007), a change in the functional dissimilarity (FD) of litter trait-values affects the degree to which resource-use complementary occurs within the decomposer community, and its capacity to induce non-additive litter mixing effects on decomposition (i.e. litter mixtures that decompose at different rates than predicted by the mass-ratio hypothesis) (Fig. 1b, Wardle and others 1997; Vos and others 2013; Handa and others 2014). These two mechanisms can operate simultaneously by affecting both decomposer community and decomposition efficiency as a result of plant (litter) composition change (e.g. Garcia-Palacios and others 2017; Santonja and others 2018b). However, our knowledge about the relative importance of these two mechanisms in controlling the decomposer communities and the efficiency of the decomposition process in streams is very limited.

Mechanisms involved in positive litter mixing effects likely include nutrient transfer, such as N or P, from nutrient-rich to nutrient-poor litter species. For example, nutrients may be transported through fungal hyphae connecting two different leaf litter species (Schimel and Hättenschwiler 2007; Gessner and others 2010; Lummer and others 2012), in which case an acceleration of decomposition in species mixtures is expected since the nutrients are used more efficiently overall. According to the literature (Wardle and others 1997; Sanpera-Calbet et al. and others 2009; Santonja and others 2015a, 2015b; Santschi and others 2018), positive mixing effects are most likely expected in the mixtures including one poor-quality litter species and a high-quality one (Fig 1b; but see Frainer and others (2015) for contrasting results). However, whether, and to what extent, mixing litter affects decomposition rates in streams remains much debated, since litter mixture effects appear to be idiosyncratic (Schindler and Gessner 2009;

99 [Gessner and others 2010](#); [Cardinale and others 2011](#); [Lecerf and others 2011](#)). Previous studies
100 highlighting contrasting results have focused on the effects of N and P in explaining such litter
101 mixture effects (e.g. [Frainer and others 2015](#); [Santschi and others 2018](#)). However, other key
102 nutrients, such as Mg or Ca, could potentially affect decomposition rates in litter mixtures.

103 Additionally, it is also not clear to what extent macroinvertebrates contribute to
104 decomposition in litter mixtures ([Hättenschwiler and Gasser 2005](#); [Swan and Palmer 2006b](#);
105 [Swan 2011](#)). Shredder macroinvertebrates can play a prominent role in the decomposition
106 process in stream ecosystems ([Anderson and Sedell 1979](#); [Cummins and Klug 1979](#); [Handa and](#)
107 [others 2014](#); [Garcia-Palacios and others 2016](#)), by consuming and fragmenting the litter
108 material ([Allan 1996](#); [Graça 2001](#)), by stimulating microbial decomposition ([Wetzel 1995](#);
109 [Villanueva and others 2012](#)), and by mediating plant diversity effects on litter decomposition
110 ([Lecerf and others 2005](#); [Kominoski and others 2007](#); [Sanpera-Calbet and others 2009](#)).
111 According to [Cummins and others \(1989\)](#), such a role played by macroinvertebrates would be
112 higher in litter mixtures of distinct degradability than in more homogeneous litter mixtures.
113 Indeed, this type of litter mixture could simultaneously create a suitable microhabitat (more
114 refractory litter) and a food resource (more labile litter). Additionally, litter mixtures could
115 promote an increase in both growth rate ([Swan and Palmer 2006a](#)) and macroinvertebrate
116 population due to their aggregative behavior ([Presa-Abós and others 2006](#)), enhancing therefore
117 the decomposition of litter mixtures ([Sanpera-Calbet and others 2009](#)).

118 In order to address these gaps, we used leaf litter from 10 common European tree species
119 to evaluate how litter diversity effects are affected by litter nutrients (Ca, K, Mg, N, Na, P, and
120 S) in a temperate stream. Moreover, in order to elucidate which organisms may be responsible
121 for mediating such litter diversity effects, we distinguished between decomposition involving
122 microorganisms alone (with decomposition in fine-mesh litterbags) and both microorganisms
123 and macroinvertebrates (with decomposition in coarse-mesh litterbags) ([Boulton and Boon](#)

1991). Five of the plant species showed nutrient-rich litter, while the other five showed nutrient-poor litter (Fig. 2). Mixtures of two litter species were created in all possible pairwise combinations, in order to distinguish between litter mixtures of similar nutrient content (i.e. in mixing two nutrient-poor or two nutrient-rich litters) and litter mixtures of dissimilar nutrient content (i.e. in mixing one nutrient-poor litter and one nutrient-rich litter). Despite the fact that both litter traits, litter mass loss and microbial activity are continuous variables, such artificial distinction into two discrete nutrient categories was necessary to better understand where and why litter diversity effects occur. We evaluated (i) the litter decomposition rate and the microbial activity (i.e. CO₂ production) in the two-species litter mixtures, and (ii) the litter diversity effects occurring on the decomposition rate and on the microbial activity in these litter mixtures after 30 days of immersion in a temperate forested stream in northwestern France (Piscart and others 2009).

First, according to the “mass-ratio hypothesis” (Grime 1998; Garnier and others 2004), the increase in both litter decomposition rate and microbial activity would be positively correlated to the increase in CWM scores in nutrient concentration within the litter mixture (Cornwell and others 2008; Foucreau and others 2013). Therefore, in our first hypothesis, we predicted higher litter decomposition rate and microbial activity in the mixture of two nutrient-rich litters (Fig. 1a). Second, according to the “niche complementarity hypothesis” (Petchey and Gaston 2006; Diaz and others 2007), mixtures of very dissimilar litter species yield the highest FD scores and are related to higher litter diversity effects (Wardle and others 1997; Lecerf and others 2011; Santschi and others 2018). Thus, in our second hypothesis, we predicted the highest litter diversity effects in the mixtures of nutrient-poor and nutrient-rich litters (Fig. 1b), which would be shown by an increase in litter decomposition rate and microbial activity due to synergistic effects between the two litters in mixtures in the present study. Finally, macroinvertebrates may contribute up to 6 times more influence than microorganisms

on litter decomposition in temperate unaltered forested streams (Piscart and other 2009), favor microbial community development (Wetzel 1995; Villanueva and others 2012), and mediate litter diversity effects (Lecerf and others 2005), we hypothesized that litter decomposition rate, microbial activity, and litter diversity effects would be enhanced by macroinvertebrate presence (Figs. 1a and 1b).

MATERIALS AND METHODS

Study site and material collection

The experiment was conducted in the Hermitage stream, located in the Villecartier Forest in northwestern France (48°28' N, 1°33' W). The stream bed was dominated by sand and leaf litter (site H1 in Piscart and others 2009). The stream water was circumneutral, well-oxygenated, and possessed moderate nutrient concentrations (Piscart and others 2009): 11 ± 1 mg l⁻¹ dissolved oxygen, 49 ± 18 µg N l⁻¹ ammonium, 510 ± 19 µg N l⁻¹ nitrate, 10 ± 8 µg N l⁻¹ nitrite, 19 ± 8 µg P l⁻¹ phosphate. The macroinvertebrate community of the Hermitage stream was composed of species belonging to the Amphipoda, Isopoda and Trichoptera orders, with crustaceans representing 74% of this macroinvertebrate community (Supplementary Table S1). *Fagus sylvatica* was the dominant tree species in the forested watershed, but the riparian vegetation was composed of diverse deciduous tree species, including those employed in the experiment.

The leaf litter of 10 common European tree species was collected: *Acer platanoides*, *Alnus glutinosa*, *Betula pendula*, *Castanea sativa*, *Corylus avellana*, *Carpinus betulus*, *Fagus sylvatica*, *Quercus robur*, *Salix atrocinerea* and *Tilia cordata*, hereafter referred to by their genus name. These 10 species were selected using the existing literature (e.g. Lecerf and others 2007; Schindler and Gessner 2009; Santonja and others 2018a), based on the nutrient

concentrations of their leaves, to represent five species with nutrient-rich litter (NRL), and five with nutrient-poor litter (NPL) (Fig. 2; a cluster analysis well discriminates these two groups). Freshly abscised leaves were collected over the entire period of maximum litter fall, from October to November 2015. They were dried at room temperature, and stored until the beginning of the experiment.

Litter decomposition experiment

Leaf litter decomposition was studied for a period of 30 days, using the litterbag method (Boulton and Boon 1991). Coarse- and fine-mesh litterbags (5 mm and 0.5 mm mesh size, respectively) were used and filled with 2 g of dry leaves. The leaf litter enclosed in fine-mesh litterbags was only accessible to microorganisms, whereas the coarse-mesh bags also allowed access to macroinvertebrates. The litterbags contained either a single species (10 treatments) or a mixture of two species in all possible pairwise combinations (45 treatments). The mixed-species litterbags received equal amounts of both species (i.e. 1 g). A total of 440 litterbags ([10 single-species + 45 two-species mixtures] \times 2 mesh sizes \times 4 replicates) were used for the experiment.

In December 2016, the litterbags were immersed for 30 days in the Hermitage stream. After removal, the litterbags were immediately sealed in plastic bags to prevent the loss of litter material and were transported to the laboratory. The leaves were separated by species, which was possible even with small fragments of litter (owing to marked morphological differences among the species) and were carefully cleaned under water to remove macroinvertebrates and attached sediment particles.

Avoiding the central veins, six leaf discs (10 mm-diameter) for the single-species litters, or three disks for each species in the two-species litter mixtures, were cut for the purpose of microbial activity measurement. The remaining leaf material was frozen at -20 °C.

199

200 ***Microbial activity measurement***

201 Microbial CO₂ production measurements, as a measure of overall heterotrophic
202 microbial activity, were performed in order to evaluate the activity of the microbial
203 communities that colonized the leaves after 30 days of decomposition. We adapted the protocol
204 proposed by [Anderson and Domsch \(1973\)](#). Briefly, the six leaf disks taken in each litterbag
205 were placed in 125 ml glass bottles with 25 ml of filtered water (GF/F glass microfiber filter,
206 WhatmanTM) from the Hermitage stream and then preincubated for 12 h at 20 °C in the dark
207 allowing the microbial respiration to saturate the water with CO₂. In the matter of fact, at
208 constant temperature, it is only possible to increase the CO₂ concentration in the air phase of
209 the respiration chamber if the aqueous phase is oversaturated in CO₂. After a night of remaining
210 open to the atmosphere, the respiration chambers were hermetically sealed and the first sample
211 of air was taken, the second air sample was sampled after a 4 h incubation at 20 °C in the dark.
212 CO₂ production in a given time (i.e. 4 h) was then calculated as the difference between the final
213 CO₂ concentration and the initial CO₂ concentration. Preliminary studies with our experimental
214 conditions demonstrated that the CO₂ production was linear and, since all the process of
215 incubation occurred in the dark, there was no photosynthesis and thus no alteration of the
216 dissolved CO₂ partial pressure. In total, 440 respiration chambers, corresponding to the 440
217 litterbags, were prepared. At the beginning and at the end of the incubation period, 1 ml of air,
218 taken with a syringe, was injected into a gas chromatograph (µGC SRA A 3000) in order to
219 estimate the microbial activity (i.e. CO₂ air content). The leaf discs were then dried at 65 °C for
220 72 h and the CO₂ production was calculated as µg C-CO₂ per h and per g of litter dry mass (±
221 0.1 mg).

222 In parallel to the measurement of the microbial activity, the leaves remaining in the 440
223 litterbags were dried at 65 °C for 72 h, and then weighed to the nearest 0.1 mg. After weighing

the leaf disks and remaining leaf litter, all the litter material from a given litterbag was combined and ground to a fine powder using a ball mill, before measuring the litter ash content. We obtained ash-free dry mass by burning the combined sample at 550 °C for 4 h. Additional samples were also used to estimate ash-free dry mass of the initial litter material. The percentage of ash-free dry mass data was used to correct both the initial and the remaining leaf material before the computation of (i) the percentage of litter mass loss after 30 days of field decomposition and (ii) the microbial CO₂ production per h and per g of litter.

Litter trait measurement

The initial litter traits were determined from four samples of each of the 10 litter species ([Supplementary Table S2](#)). Prior to the chemical analysis, each litter sample was ground into powder using a ball mill. The carbon (C), nitrogen (N), and sulfur (S) concentrations were determined by thermal combustion, using a Vario Pyro cube CNS analyzer (Elementar France SARL, Lyon, France). The phosphorus (P) concentration was measured colorimetrically using the molybdenum blue method ([Grimshaw and others 1989](#)). To 80 mg of ground litter sample 8 ml of HNO₃ and 2 ml of H₂O₂ were added and the mixture was heated at 175 °C for 40 min using microwaves (Ethos One, Milestone SRL, Italy). After this mineralization step, the sample was diluted to a total of 50 ml. A hundred µl of sample, 100 µl of NaOH, 50 µl of mixed reagent (emetic tartar and ammonium molybdate solution), and 50 µl of ascorbic acid were mixed directly in a 96 well microplate. After 30 min at 40 °C, the reaction was completed, and the P concentration was measured at 720 nm using a microplate reader (Victor, PerkinElmer, Singapore). Following the mineralization step (i.e. the same as for P analysis), calcium (Ca), magnesium (Mg), potassium (K), and sodium (Na) concentrations were measured using an atomic absorption spectrometer (AAS, iCE 3000 series, ThermoScientific, China).

To assess the “mass-ratio hypothesis”, we calculated the community-weighted mean (CWM) trait values of litter mixtures as the average trait values of litter mixtures following Garnier and others (2004) as: $Trait_{CWM} = \sum_{i=1}^n p_i \times trait_i$, where p_i is the relative abundance for species i in the litter mixture and $trait_i$ is the trait value for species i . These calculations were performed for each of the 8 litter traits. The highest scores of CWM were reached for litter mixtures containing two litter species with the highest nutrient concentrations (Fig. 1a). Since we predicted that both microbial activity and litter decomposition rate respond to the “mass-ratio hypothesis”, we expected the increase in both decomposition rate and microbial activity to be positively correlated to the increase in CWM values.

To assess the “niche complementarity hypothesis”, we calculated the functional dissimilarity (FD) of litter mixtures according to Rao’s quadratic entropy (Botta Dukat 2005; Epps and others 2007) as: $Trait_{FD} = \sum_{i=1}^n \sum_{j=1}^n p_i p_j * dij$, where p_i and p_j are the relative abundance for species i and j in the litter mixture, and d_{ij} the Euclidian distance between species i and j for the trait considered. These calculations were performed for each of the 8 litter traits. The highest scores of FD were reached for litter mixtures containing two species with very dissimilar nutrient concentrations (Fig. 1b). Since we predicted that the litter diversity effects respond to the “niche complementarity hypothesis”, we expected the increase in relative litter mixture effects to be positively correlated to the increase in FD values.

Statistical analyses

All of the statistical analyses were conducted using R software (R Core Team 2013), with significance levels indicated as * for $P < 0.05$, ** for $P < 0.01$, and *** for $P < 0.001$.

A principal component analysis (PCA) was conducted using the values of the eight measured litter traits of the 10 tree species (Supplementary Table S2) in order to discriminate the five tree species with nutrient-rich litter (NRL) from the other five tree species with nutrient-

poor litter (NPL). The differences in the initial litter traits were assessed using one-way ANOVAs, followed by Tukey's tests to carry out post-hoc pairwise comparisons.

For the single-species litter, three-way ANOVAs, followed by Tukey's post hoc tests, were used to test for the effects of litter type (separated in NPL *vs.* NRL), litter species identity (10 litters), macroinvertebrate presence (fine-mesh bag [FMB] *vs.* coarse-mesh bag [CMB]), and their interactions on litter decomposition rate and microbial activity.

In order to further test whether litter decomposition and microbial activity differed between litter mixtures and single litter species, the relative litter mixture effects (RME) on litter decomposition rate and microbial activity were calculated. The RME was calculated as the relative difference between the observed litter decomposition rate/microbial activity (O) from the litter mixtures compared to those expected based on the respective single litter species treatments (E), following the formula $(O - E) / E \times 100\%$ (Wardle and others 1997). One-sample Student's *t*-tests were used to test whether the RME were significantly different from zero.

For the two-species litter mixtures, two-way ANOVAs, followed by Tukey's post hoc tests, were used to test for the effects of litter mixing (NPL-NPL, NPL-NRL and NRL-NRL), macroinvertebrate presence (FMB *vs.* CMB), and their interactions i) on litter decomposition rate and microbial activity and ii) on the RME on litter decomposition rate and on microbial activity.

For a more detailed understanding of how the mixture of leaf litter affected the litter decomposition rate and microbial activity, we evaluated the effects of the mean litter traits (Trait_{CWM}) and functional litter trait dissimilarities (Trait_{FD}) of the eight measured litter traits (Supplementary Table S2). First, a principal component analysis (PCA) was conducted using the CWM or the FD values of the eight measured litter traits across the 45 two-species litter mixtures. CWM1 and CWM2, and FD1 and FD2 represented the two first components of the

PCAs conducted using the CWM or the FD values across the litter mixtures, respectively. Second, multiple linear regression models were performed in order to decipher the relative contributions of Trait_{CWM} and Trait_{FD}. In these models the effects of the Traits_{CWM} (i.e. CWM1 and CWM2), Traits_{FD} (i.e. FD1 and FD2), macroinvertebrate presence (FMB vs. CMB), and their interactions were tested i) on litter decomposition rate and microbial activity and ii) on the RME on litter decomposition rate and on microbial activity.

RESULTS

Litter traits

The PCA based on the element concentrations showed that the first PCA axis explained 50.2% of the variation and discriminated between the nutrient-rich litters (NRL) on the left and the nutrient-poor litters (NPL) on the right of the axis (Fig. 2).

Litter species incubated individually

The NRL exhibited two times more litter mass loss and microbial activity than the NPL (Table 1; Fig. 3). The presence of macroinvertebrates increased the litter mass loss and the microbial activity, but this effect was dependent on the litter type (litter type \times mesh size interaction, Table 1). This significant interaction was explained by a stronger increase in litter mass loss and microbial activity in the NRL category (+59% litter mass loss and +33% microbial activity) compared to the NPL category (+23% litter mass loss and +28% microbial activity) (Fig. 3).

In addition, litter mass loss and microbial activity were significantly affected by litter species identity (Table 1). Within the NRL category, *Alnus* showed the highest litter mass loss compared to the four other species (Supplementary Fig. S1a), as well as a higher microbial

activity than *Carpinus*, *Acer*, and *Tilia* (Supplementary Fig. S1b). Within the NPL category, *Salix* and *Betula* showed both a higher litter mass loss and a higher microbial activity than the three other species (Supplementary Fig. S1a and S1b). The positive effect of the macroinvertebrate presence on the litter mass loss also varied according to the litter species identity (litter species \times mesh size interaction, Table 1), as the effects ranged from +68% for *Alnus* to an absence of effect for *Fagus* (Supplementary Fig. S1a).

Effects of mixing low- and high-quality litter

Litter mass loss and microbial activity increased according to the gradient NPL-NPL < NPL-NRL < NRL-NRL mixtures (Table 2; Fig. 4a and 4c), with two times more litter mass loss and microbial activity in mixtures of two nutrient-rich litters compared to the mixtures of two nutrient-poor litters. The relative mixture effects (RME) on litter mass loss and on microbial activity showed a similar trend, with an increase in RME according to the same gradient (Table 2; Fig. 5). The NPL-NPL mixtures lost -3.5% litter mass compared with the expected values from the respective single litter species (Fig. 5a), while the observed microbial activity of the NPL-NPL mixtures did not differ significantly from the expected values (Fig. 5c). The NPL-NRL mixtures lost +5.3% litter mass than expected from the respective single litter species (Fig. 5a), while the observed microbial activity of the NPL-NRL mixtures did not differ from the expected values (Fig. 5c). The NRL-NRL mixtures showed +8.9% litter mass and +16.3% microbial activity than expected from the respective single litter species (Figs. 5a and 5c).

Litter mass loss and microbial activity were respectively 41% and 30% higher with macroinvertebrates than without (Table 2; Fig. 4b and 4d). The RME on litter mass loss and on microbial activity were also higher with macroinvertebrates than without (Table 2). Specifically, the litter mixtures exhibited synergistic effects in the presence of

macroinvertebrates, with +7.3% litter mass loss and +9.9% microbial activity than expected from the respective single litter species (Figs. 5b and 5d).

CWM- versus FD-trait control over litter decomposition rate and microbial activity

The PCA of the CWM traits showed that the first PCA axis (CWM1), explaining 50.2% of the variation, was determined by high scores of P and Ca concentrations, and, to a lower extent, by the scores of the K, Mg, Na, and S concentrations (Supplementary Fig. S2a). The low scores of the second PCA axis (CWM2), explaining 21.9% of the variation, were related to high values of N and C concentrations (Supplementary Fig. S2a). Regarding functional trait dissimilarity, high scores of the first PCA axis (FD1), explaining 28.5% of the variation, were related to the increase in dissimilarity in the Mg, S, and C concentrations, while the low scores were related to the increase in dissimilarity in the N and Ca concentrations (Supplementary Fig. S2b). The low scores along the second axis (FD2), explaining 22.0% of the variation, were largely dependent on the increase in dissimilarity in the K, Na, and P concentrations (Supplementary Fig. S2b).

When simultaneously evaluating the effects of the CWM and FD traits (Table 3), we found that both litter mass loss and microbial activity were mainly controlled by the CWM values (i.e. increasing mean nutrient concentrations within the litter mixture) compared to the FD values (i.e. increasing dissimilarity in nutrient concentrations within the litter mixture). In fact, CWM values (CWM1 + CWM2) explained respectively 9 and 15 times more of the overall variance in litter mass loss and in microbial activity than FD values (FD1 + FD2) (Table 3).

When simultaneously evaluating the effects of the CWM and FD traits on RME, we found that the RME on litter mass loss was controlled by the CWM (CWM1), the presence of macroinvertebrates, and the interaction between the FD and the presence of macroinvertebrates (FD2 \times mesh size) (Table 3). Increasing CWM1 scores was related to higher RME (Fig. 6a).

The significant interaction between FD2 and mesh size (Table 3) showed that increasing FD in the initial K, Na, and P concentrations stimulated microbial-driven RME (Fig. 6d). Concerning the RME on microbial activity, it was found to be significantly affected only by CWM1 and the presence of macroinvertebrates (Table 3). Similar to what we observed for litter mass loss, increasing CWM1 scores was related to higher RME on microbial activity (Fig. 6e).

The RME on litter mass loss appeared to be more strongly correlated with the increase in P, Ca, Mg and Na concentrations than with the K, N, or S concentrations, and these relationships were more marked in the coarse-mesh litterbags in which macroinvertebrates were present, than in the fine-mesh litterbags in which macroinvertebrates were absent (Table 4). The RME on microbial activity appeared to be more strongly correlated with the increase in Ca, K, and Mg concentrations than in N and P concentrations, and these relationships were also more marked in the presence than in the absence of macroinvertebrates (Table 4).

DISCUSSION

Niche complementarity hypothesis vs. mass-ratio hypothesis for explaining litter diversity effects

We found evidence of leaf litter diversity effects on decomposition rates, including additive, synergistic, and antagonistic effects, as previously shown in other studies (Kominoski and others 2007; Srivastava and others 2009; Lecerf and others 2011; Handa and others 2014). Surprisingly, the pattern observed in our study highlighted that the largest mixture effects occurred in mixtures containing the combination of two nutrient-rich litters (NRLs), rather than in mixtures of one nutrient-rich litter (NRL) and one nutrient-poor litter (NPL), as predicted in our second hypothesis (Fig. 1b). Such a finding is in agreement with the mass-ratio hypothesis, and emphasizes the fact that increasing the mean value of the nutrient pool in litter mixtures

favors the occurrence of litter diversity effects. In any case our results pointed out that the litter diversity effects on decomposition were strongly litter-quality dependent. This finding is congruent with the recent study of [Jabiou and Chauvet \(2015\)](#) in which higher synergistic effects on litter decomposition occurred when *Alnus glutinosa* was combined with *Fraxinus angustifolia* (i.e. two NRLs) than when *Alnus glutinosa* was combined with *Quercus ilex* (i.e. one NRL and one NPL).

Surprisingly, $Trait_{CWM}$ proved to be a better predictor of litter diversity effects than $Trait_{FD}$, for both the coarse- and fine-mesh litterbags, contrasting with our second hypothesis. Indeed, based on the well-developed literature on the drivers of the litter decomposition process (e.g. [Hättenschwiler and others 2005](#); [Cornwell and others 2008](#); [Gessner and others 2010](#); [Makkonen and others 2012](#); [Vos and others 2013](#); [Handa and others 2014](#); [Garcia-Palacios and others 2017](#)), we expected that litter diversity effects occur when there are increases in the nutrient concentration dissimilarity of the litter mixtures (i.e. according to the niche complementary hypothesis, [Fig. 1b](#)), while the mean nutrient concentration within the litter mixture must only explain the litter decomposition rate (i.e. according to the mass-ratio hypothesis, [Fig. 1a](#)). The results from the coarse-mesh litterbags showed no evidence of nutrient dissimilarity effect ($Trait_{FD}$), conforming with the conclusion of [Frainer and others \(2015\)](#), which also highlighted, over a lower range of litter traits (N, P, and lignin), that no litter dissimilarity effect was involved in the litter diversity effects occurring in two-species mixtures. In contrast, $Trait_{FD}$ was linked to the litter diversity effects for the fine-mesh litterbags. In this case, litter mixtures with contrasting nutrient concentrations may have improved the availability of different nutrient sources for microorganisms ([Schimel and Hättenschwiler 2007](#); [Handa and others 2014](#)), as suggested by the niche complementarity hypothesis. Filamentous fungi, including the aquatic hyphomycetes that dominate fungal communities on decomposing leaves in streams ([Krauss and others 2011](#)), can indeed extend their hyphae over considerable distances

in order to acquire remote resources that they transport to the locations of active hyphal growth (Ritz 2006). It has previously been suggested that fungi-driven N transfer among litter species varying in their initial N concentration may contribute to litter mixture effects (Schimel and Hättenschwiler 2007; Vos and others 2013; Handa and others 2014). Thus, nutrients from a nutrient-rich litter may be translocated to another litter depleted in nutrient (i.e. a nutrient-poor litter; Schimel and Hättenschwiler 2007; Handa and others 2014). Therefore, our findings highlighted that both Trait_{CWM} and Trait_{FD} contributed to explain litter diversity effects driven by microorganisms alone. Interestingly, Trait_{FD} did not explain litter diversity effects driven by microorganisms and macroinvertebrates together. In this case, it could be hypothesized that the additional nutrients provided by macroinvertebrate feces alleviated the nutrient limitation for microorganisms (Wetzel 1995; Joyce and Wotton 2008), and consequently the Trait_{FD} contribution to litter diversity effects.

Macroinvertebrates control over decomposition and litter diversity effects

Across all of the 45 litter mixtures, litter mixture decomposition was not found to be enhanced in the fine-mesh litterbags (i.e. additive effects), while the litter mixtures in the coarse-mesh litterbags lost on average 7.3% more mass than the litter in the single-species treatments (i.e. synergistic effects). The absence of litter mixture effects when macroinvertebrates were excluded suggests that the activity of microbial decomposers alone did not induce any effect of litter mixture on decomposition in the studied temperate stream. Previous studies performed in lotic systems, where decomposition primarily involves microorganisms, have also reported a lack of a synergism (Ferreira and others 2012; Bruder and others 2014). For instance, similar to our findings, Jabiol and Chauvet (2015) reported that no effect of litter mixture was observed when detritivores were excluded during the litter decomposition in a Mediterranean stream in southern France. In a laboratory experiment, Swan

and Palmer (2006b) also reported that litter mixture effects were contingent on the feeding activity of the isopod *Caecidotea communis*.

In homogenous species mixtures comprising litters of a single litter category, macroinvertebrates have little choice to select among different qualities of leaves, as opposed to the choice offered between leaf litter species in heterogeneous litter mixtures (Swan and Palmer 2006b). However, as macroinvertebrates can simultaneously exploit multiple litter species to meet their elemental demands (Leroy and Marks 2006), combining two NRLs that nutritionally complement one another might stimulate the feeding activity of detritivores (Vos and others 2013). For example, the mixing of one P-rich with one N-rich species in the present study (such as *Acer platanoides* and *Alnus glutinosa*, respectively) increased litter mass loss of the mixture by 14%, probably due to the increased nutritional value of such litter mixing. In contrast, mixing two P-rich species (e.g. *Acer platanoides* and *Carpinus betulus*) increased litter mass loss of the mixture by only 4%, while mixing two P-poor species (e.g. *Castanea sativa* and *Fagus sylvatica*) decreased litter mass loss of the mixture by 10%. Alternatively in NRL+NPL mixtures, high detritivore density may be promoted by the co-occurrence of a suitable resource, such as that provided by the nutrient-rich litter, and a complex and structured habitat provided by the nutrient-poor litter (Sanpera-Calbet and others 2009; Jabiol and others 2014).

Litter nutrients control over decomposition and litter diversity effects

In agreement with our predictions and the literature, nutrient-rich litters decomposed faster than nutrient-poor litters, confirming that nutrient concentration is an important determinant of leaf litter decomposition (Kaspari and others 2008; Makkonen and others 2012; Garcia-Palacios and others 2016). The rates of litter decomposition observed in the present study were similar to those reported in other studies concerning temperate streams (e.g. Lecerf

and others 2007; Schindler and Gessner 2009; Frainer and others 2015). For instance, Lecerf and others (2007) reported 80% and 42% mass loss for *Alnus glutinosa* (i.e. a nutrient-rich litter) and *Quercus robur* (i.e. a nutrient-poor litter), respectively, after 34 days of decomposition in a temperate stream in central France. The results of the present study were also similar to the 16.5% mass loss for *Fagus sylvatica* (i.e. a nutrient-poor litter) after 30 days of decomposition obtained in the same Petit Hermitage stream in February 2005 (Piscart and others 2009).

Moreover, our results also highlighted the important role of nutrients not usually considered in litter diversity experiments. The first principal component of the CWM-trait PCA was the main driver of litter diversity effects in both fine- and coarse-mesh litterbags, and was driven by P, Ca, K, Mg, Na, and S. Their relative content in the leaves is closely related to one other (Garcia-Palacios and others 2016), hampering a straightforward interpretation of combination of elements. Nevertheless, when evaluating the individual contributions of litter nutrients, Ca and Mg appeared to be important drivers of litter diversity effects compared to N (and to P only for microbial activity), suggesting for the first time that these two nutrients may play an important role in litter diversity effects in addition to, or independently of, the P or N content. The Ca content is known to positively affect the growth and activity of aquatic hyphomycetes, by enhancing the fungal capacity to transfer N between distinct litter types (Jenkins and Suberkropp 1995). Meanwhile, Ca and Mg are known to be essential elements in the diet of macroinvertebrates, since they are required for enzymatic reactions, nerve connections, muscle function, and skeleton formation (National Research Council 2005). Moreover, in some temperate streams, crustaceans could represent up to 95% of shredder biomass (Piscart and others 2011), and the Ca content of leaves could be a significant source of Ca for their cuticles (Cairns and Yan 2009), especially in streams with a low Ca content in the water (Glazier 1998), such as in Brittany, where the present study was conducted.

498

499 CONCLUSION

500 As expected, increasing mean nutrient concentration and macroinvertebrate presence
501 favored both microbial activity and litter decomposition rate. In addition, we experimentally
502 demonstrated for the first time that the mass-ratio hypothesis (i.e. Trait_{CWM}) overrules the niche
503 complementarity hypothesis (Trait_{FD}) as a driver of litter diversity effects in a temperate stream.
504 In fact, in strong contrast to our expectations, the combination of two nutrient-rich litters yielded
505 the highest litter mixture effects. In addition, the synergistic effects of litter mixing were mainly
506 evident in the presence of macroinvertebrates. Both community weighted mean traits
507 (Trait_{CWM}) and the functional dissimilarity of litter traits (Trait_{FD}) contributed to explain the
508 litter mixture effects. There was no support for Trait_{FD} explaining litter diversity effects in the
509 presence of macroinvertebrates, while both Trait_{CWM} and Trait_{FD} contributed to explain litter
510 mixture effects driven by microorganisms alone. Finally, Ca and Mg, which are often neglected
511 in decomposition studies in streams, were found to be important drivers of litter mixture effects.

512

513 ACKNOWLEDGMENTS

514 The chemical analyses were performed at the Plateforme d'Analyses Chimiques en
515 Ecologie (PACE, LabEx Centre Méditerranéen de l'Environnement et de la Biodiversité,
516 Montpellier, France), as well as at the PLateforme AnalYtique (PLAY, UMR CNRS 6553
517 ECOBIO, Rennes, France). We thank Raphaëlle Leclerc, Bruno Buatois and Nicolas Barthes
518 for technical assistance during the chemical analyses as well as Pierre Mariotte for his reviewing
519 of the English. This study was funded by the Selune project (project number 1 053 864) and by
520 the Rennes Metropole's project AIS 2015. The study was also supported by the LTSER France
521 Zone Atelier Armorique. Finally, the authors declare no conflict of interest.

522

AUTHORS' CONTRIBUTIONS

M.S., H.R.P., N.L.B. and C.P. conceived and performed the experiments. M.S. analyzed the data and led the writing of the manuscript. All authors contributed critically to the drafts and gave final approval for publication.

REFERENCES

- Allan DJ. 1996. Stream Ecology. Chapman and Hall, New York.
- Anderson, JPE, Domsch KH. 1973. Quantification of bacterial and fungal contributions to soil respiration. Archiv für Mikrobiologie 93:113-127. Doi: 10.1007/BF00424942
- Anderson NH, Sedell JR. 1979. Detritus processing by macroinvertebrates in stream ecosystems. Annual Review of Entomology 24:351-377. Doi: 10.1146/annurev.en.24.010179.002031
- Botta Dukát Z. 2005. Rao's quadratic entropy as a measure of functional diversity based on multiple traits. Journal of Vegetation Science 16:533-540. Doi: 10.1658/1100-9233(2005)16[533:RQEAAM]2.0.CO;2
- Boulton AJ, Boon PI. 1991. A review of methodology used to measure leaf litter decomposition in lotic environments: time to turn over an old leaf? Australian Journal of Marine Freshwater Resources 42:1-43. Doi: 10.1071/MF9910001
- Bruder A, Schindler MH, Moretti MS, Gessner MO. 2014. Litter decomposition in a temperate and a tropical stream: the effects of species mixing, litter quality and shredders. Freshwater Biology 59:438-449. <https://doi.org/10.1111/fwb.12276>
- Cadish G, Giller KE. 1997. Driven by Nature: Plant Litter Quality and Decomposition. CAB international, Wallingford.

546 Cairns A, Yan N. 2009. A review of the influence of low ambient calcium concentrations on
547 freshwater daphniids, gammarids, and crayfish. *Environmental Reviews* 17:67-79. Doi:
548 10.1139/A09-005

549 Cardinale BJ, Matulich KL, Hooper DU, Byrnes JE, Duffy E, Gamfeldt L, ... Gonzalez A.
550 2011. The functional role of producer diversity in ecosystems. *American Journal of Botany*
551 98:572-592. Doi: 10.3732/ajb.1000364

552 Cornwell WK, Cornelissen JH, Amatangelo K, Dorrepaal E, Eviner VT, Godoy O, ... Westoby
553 M. 2008. Plant species traits are the predominant control on litter decomposition rates within
554 biomes worldwide. *Ecology letters* 11:1065-1071. Doi: 10.1111/j.1461-0248.2008.01219.x

555 Cummins KW, Klug MJ. 1979. Feeding ecology of stream invertebrates. *Annual review of*
556 *ecology and systematics* 10:147-172. Doi: 10.1146/annurev.es.10.110179.001051

557 Cummins KW, Wilzbach MA, Gates DM, Perry JB, Taliferro WB. 1989. Shredders and riparian
558 vegetation: Leaf litter that falls into streams influences communities of stream invertebrates.
559 *BioSciences* 39:24-30 Doi: 10.2307/1310804

560 Cummins KW. 2012. Structure and function of stream ecosystems. *Bioscience* 24:631-641.
561 <https://doi.org/10.2307/1296676>

562 Diaz S, Lavorel S, De Bello F, Quetier F, Grigulis K, Robson TM. 2007. Incorporating plant
563 functional diversity effects in ecosystem service assessments. *Proceedings of the National*
564 *Academy of Sciences of the USA* 104:20684-20689. Doi: 10.1073/pnas.0704716104

565 Epps KY, Comerford NB, Reeves III JB, Cropper Jr WP, Araujo QR. 2007. Chemical diversity-
566 highlighting a species richness and ecosystem function disconnect. *Oikos* 116:1831-1840.
567 Doi: 10.1111/j.0030-1299.2007.15853.x

568 Ferreira V, Encalada AC, Graça MA. 2012. Effects of litter diversity on decomposition and
569 biological colonization of submerged litter in temperate and tropical streams. *Freshwater*
570 *Science* 31:945-962. Doi: 10.1899/11-062.1

571 Fierer N, Craine JM, McLauchlan K, Schimel JP. 2005. Litter quality and the temperature
572 sensitivity of decomposition. *Ecology* 86:320-326. Doi: 10.1890/04-1254

573 Follstad Shah JJ, Kominoski JS, Ardón M, Dodds WK, Gessner MO, Griffiths NA, ... Manning
574 DW. 2017. Global synthesis of the temperature sensitivity of leaf litter breakdown in streams
575 and rivers. *Global Change Biology* 23:3064-3075. Doi: 10.1111/gcb.13609

576 Foucreau N, Puijalon S, Hervant F, Piscart C. 2013. Effect of leaf litter characteristics on leaf
577 conditioning and on consumption by *Gammarus pulex*. *Freshwater Biology* 58:1672-1681.
578 Doi: 10.1111/fwb.12158

579 Frainer A, Moretti MS, Xu W, Gessner MO. 2015. No evidence for leaf-trait dissimilarity
580 effects on litter decomposition, fungal decomposers, and nutrient dynamics. *Ecology*
581 96:550-561. Doi: 10.1890/14-1151.1

582 Garcia-Palacios P, McKie BG, Handa IT, Frainer A, Hättenschwiler S. 2016. The importance
583 of litter traits and decomposers for litter decomposition: a comparison of aquatic and
584 terrestrial ecosystems within and across biomes. *Functional Ecology* 30:819-829. Doi:
585 10.1111/1365-2435.12589

586 Garcia-Palacios P, Shaw EA, Wall DH, Hättenschwiler S. 2017. Contrasting mass-ratio vs.
587 niche complementarity effects on litter C and N loss during decomposition along a regional
588 climatic gradient. *Journal of Ecology* 105:968-978. Doi: 10.1111/j.1466-8238.2011.00651.x

589 Garnier E, Cortez J, Billès G, Navas ML, Roumet C, Debussche M, ... Toussaint JP. 2004.
590 Plant functional markers capture ecosystem properties during secondary
591 succession. *Ecology* 85:2630-2637. Doi: 10.1890/03-0799

592 Gessner MO, Chauvet E. 1994. Importance of stream microfungi in controlling breakdown rates
593 of leaf litter. *Ecology* 75:1807-1817. Doi: 10.2307/1939639

594 Gessner MO, Chauvet E, Dobson M. 1999. A perspective on leaf litter breakdown in streams.
595 *Oikos* 85:377-384. Doi: 10.2307/3546505

596 Gessner MO, Swan CM, Dang CK, Mckie BG, Bardgett RD, Wall DH, Hättenschwiler S. 2010.
597 Diversity meets decomposition. Trends in Ecology and Evolution 25:372-380. Doi:
598 10.1016/j.tree.2010.01.010

599 Glazier DS. 1998. Springs as model systems for ecology and evolutionary biology: a case study
600 of *Gammarus minus* Say (Amphipoda) in mid-Appalachian springs differing in pH and ionic
601 content. In: Botosaneanu L (eds) Studies in crenobiology: the biology of springs and
602 springbrooks, Backhuys Publishers, Leiden, pp 41-54.

603 Graça MAS. 2001. The role of invertebrates on leaf litter decomposition in streams-a review.
604 International Review of Hydrobiology 86:383-393. Doi: 10.1002/1522-
605 2632(200107)86:4/5<383::AID-IROH383>3.0.CO;2-D

606 Grime JP. 1998. Benefits of plant diversity to ecosystems: immediate, filter and founder effects.
607 Journal of Ecology 86:902-910. Doi: 10.1046/j.1365-2745.1998.00306.x

608 Grimshaw HM, Allen SE, Parkinson JA. 1989. Nutrient elements. In: Allen SE (eds) Chemical
609 analysis of ecological materials, 2nd Edition, Blackwell Scientific, Oxford, pp 81-159.

610 Handa IT, Aerts R, Berendse F, Berg MP, Bruder A, Butenschoen O, ... Hättenschwiler S.
611 2014. Consequences of biodiversity loss for litter decomposition across
612 biomes. Nature 509:218-221. Doi: 10.1038/nature13247

613 Hättenschwiler S, Gasser P. 2005. Soil animals alter plant litter diversity effects on
614 decomposition. Proceedings of the National Academy of Sciences, 102:1519-1524.
615 <https://doi.org/10.1073/pnas.0404977102>

616 Hättenschwiler S, Tiunov AV, Scheu S. 2005. Biodiversity and litter decomposition in
617 terrestrial ecosystems. Annual Review of Ecology, Evolution and Systematics 36:191-218.
618 Doi: 10.1146/annurev.ecolsys.36.112904.151932

619 Jabiol J, McKie BG, Bruder A, Bernadet C, Gessner MO, Chauvet E. 2013. Trophic complexity
 620 enhances ecosystem functioning in an aquatic detritus-based model system. *Journal of*
 621 *Animal Ecology* 82:1042-1051. Doi: 10.1111/1365-2656.12079
 622 Jabiol J, Cornut J, Danger M, Jouffroy M, Elger A, Chauvet E. 2014. Litter identity mediates
 623 predator impacts on the functioning of an aquatic detritus-based food web. *Oecologia*,
 624 176:225-235. Doi: 10.1007/s00442-014-2990-y
 625 Jabiol J, Chauvet E. 2015. Biodiversity and litter decomposition: a case study in a
 626 Mediterranean stream. *Freshwater Science* 34:423-430. Doi: 10.1086/680094
 627 Jenkins CC, Suberkropp K. 1995. The influence of water chemistry on the enzymatic
 628 degradation of leaves in streams. *Freshwater Biology* 33:245-253. Doi: 10.1111/j.1365-
 629 2427.1995.tb01165.x
 630 Joyce P, Wotton RS. 2008. Shredder fecal pellets as stores of allochthonous organic matter in
 631 streams. *Journal of the North American Benthological Society* 27:521-528. Doi: 10.1899/07-
 632 102.1
 633 Kaspari M, Wright J, Yavitt J, Harms K, Garcia M, Santana M. 2008. Multiple nutrients limit
 634 litterfall and decomposition in a tropical forest. *Ecology Letters* 11:35-43. Doi:
 635 10.1111/j.1461-0248.2007.01124.x
 636 Kominoski JS, Pringle CM, Ball BA, Bradford MA, Coleman DC, Hall DB, Hunter MD. 2007.
 637 Nonadditive effects of leaf litter species diversity on breakdown dynamics in a detritus-based
 638 stream. *Ecology* 88:1167-1176. Doi: 10.1890/06-0674
 639 Krauss GJ, Solé M, Krauss G, Schlosser D, Wesenberg D, Bärlocher F. 2011. Fungi in
 640 freshwaters: ecology, physiology and biochemical potential. *FEMS microbiology reviews*
 641 35:620-651. Doi: 10.1111/j.1574-6976.2011.00266.x
 642 Laughlin DC. 2011. Nitrification is linked to dominant leaf traits rather than functional
 643 diversity. *Journal of Ecology* 99:1091-1099. Doi: 10.1111/j.1365-2745.2011.01856.x

644 Lecerf A, Dobson M, Dang CK, Chauvet E. 2005. Riparian plant species loss alters trophic
 645 dynamics in detritus-based stream ecosystems. *Oecologia* 146:432-442. Doi:
 646 10.1007/s00442-005-0212-3

647 Lecerf A, Risnoveanu G, Popescu C, Gessner MO, Chauvet E. 2007. Decomposition of diverse
 648 litter mixtures in streams. *Ecology* 88:219-227. [https://doi.org/10.1890/0012-](https://doi.org/10.1890/0012-9658(2007)88[219:DODLMI]2.0.CO;2)
 649 9658(2007)88[219:DODLMI]2.0.CO;2

650 Lecerf A, Marie G, Kominoski JS, LeRoy CJ, Bernadet C, Swan CM. 2011. Incubation time,
 651 functional litter diversity, and habitat characteristics predict litter-mixing effects on
 652 decomposition. *Ecology* 92:160-169. Doi: 10.1890/10-0315.1

653 Leroy CJ, Marks JC. 2006. Litter quality, stream characteristics and litter diversity influence
 654 decomposition rates and macroinvertebrates. *Freshwater biology* 51:605-617. Doi:
 655 10.1111/j.1365-2427.2006.01512.x

656 Lummer D, Scheu S, Butenschoen O. 2012. Connecting litter quality, microbial community
 657 and nitrogen transfer mechanisms in decomposing litter mixtures. *Oikos* 121:1649-1655.
 658 Doi: 10.1111/j.1600-0706.2011.20073.x

659 Makkonen M, Berg MP, Handa IT, Hättenschwiler S, van Ruijven J, van Bodegom PM, ...
 660 Aerts R. 2012. Highly consistent effects of plant litter identity and functional traits on
 661 decomposition across a latitudinal gradient. *Ecology Letters* 15:1033-1041. Doi:
 662 10.1111/j.1461-0248.2012.01826.x

663 Mokany K, Ash J, Roxburgh S. 2008. Functional identity is more important than diversity in
 664 influencing ecosystem processes in a temperate native grassland. *Journal of Ecology* 96:884-
 665 893. Doi: 10.1111/j.1365-2745.2008.01395.x

666 National Research Council. 2005. Mineral Tolerance of Animals, 2nd ed. National Academic
 667 Press, Washington.

668 Petchey OL, Gaston KJ. 2006. Functional diversity: back to basics and looking forward.

669 Ecology letters 9:741-758.doi: 10.1111/j.1461-0248.2006.00924.x

670 Piscart C, Genoel R, Dolédec S, Chauvet E, Marmonier P. 2009. Effects of intense agricultural
671 practices on heterotrophic processes in streams. *Environmental Pollution* 157:1011-1018.
672 Doi: 10.1016/j.envpol.2008.10.010

673 Piscart C, Navel S, Maazouzi C, Montuelle B, Cornut J, Mermillod-Blondin F, Creuse des
674 Chatelliers M, Simon L, Marmonier P. 2011. Leaf litter recycling in benthic and hyporheic
675 layers in agricultural streams with different types of land use. *Science of the Total*
676 *Environment* 409:4373-4380. Doi: 10.1016/j.scitotenv.2011.06.060

677 Presa Abos C, Lepori F, McKie BG, Malmqvist B. 2006. Aggregation among resource patches
678 can promote coexistence in stream-living shredders. *Freshwater Biology* 51:545-553.
679 <https://doi.org/10.1111/j.1365-2427.2006.01509.x>

680 Quested H, Eriksson O, Fortunel C, Garnier E. 2007 Plant traits relate to whole-community
681 litter quality and decomposition following land use change. *Functional Ecology* 21:1016-
682 1026. Doi: 10.1111/j.1365-2435.2007.01324.x

683 R Core Team. 2013. R: A Language and Environment for Statistical Computing. R Foundation
684 for Statistical Computing, Vienna, Austria.

685 Ritz K. 2006. Fungal roles in transport processes in soils. In MG Gadd eds, *Fungi in*
686 *biogeochemical cycles*. Cambridge University Press, Cambridge, UK, pp. 51-73.

687 Sanpera-Calbet I, Lecerf A, Chauvet E. 2009. Leaf diversity influences in-stream litter
688 decomposition through effects on shredders. *Freshwater Biology* 54:1671-1682. Doi:
689 10.1111/j.1365-2427.2009.02216.x

690 Santonja M, Baldy V, Fernandez C, Balesdent J, Gauquelin T. 2015a. Potential shift in plant
691 communities with climate change in a Mediterranean Oak forest: consequence on nutrients
692 and secondary metabolites release during litter decomposition. *Ecosystems* 18:1253-1268.
693 Doi: 10.1007/s10021-015-9896-3

694 Santonja M, Fernandez C, Gauquelin T, Baldy V. 2015b. Climate change effects on litter
 695 decomposition: intensive drought leads to a strong decrease of litter mixture interactions.
 696 Plant and Soil 393:69-82. Doi: 10.1007/s11104-015-2471-z
 697 Santonja M, Pellan L, Piscart C. 2018a. Macroinvertebrate identity mediates the effects of litter
 698 quality and microbial conditioning on the leaf litter recycling in temperate streams. Ecology
 699 and Evolution 8:2542-2553. Doi: 10.1002/ece3.3790
 700 Santonja M, Foucault Q, Rancon A, Gauquelin T, Fernandez C, Baldy V, Mirleau P. 2018b.
 701 Contrasting responses of bacterial and fungal communities to plant litter diversity in a
 702 Mediterranean oak forest. Soil Biology and Biochemistry 125:27-36. Doi:
 703 10.1016/j.soilbio.2018.06.020
 704 Santschi F, Gounand I, Harvey E, Altermatt F. 2018. Leaf litter diversity and structure of
 705 microbial decomposer communities modulate litter decomposition in aquatic systems.
 706 Functional Ecology 32:522-532. Doi: 10.1111/1365-2435.12980
 707 Schimel JP, Hättenschwiler S. 2007. Nitrogen transfer between decomposing leaves of different
 708 N status. Soil Biology and Biochemistry 39:1428-1436. Doi: 10.1016/j.soilbio.2006.12.037
 709 Schindler MH, Gessner MO. 2009. Functional leaf traits and biodiversity effects on litter
 710 decomposition in a stream. Ecology 90:1641-1649. Doi: 10.1890/08-1597.1
 711 Srivastava DS, Cardinale BJ, Downing AL, Duffy JE, Jouseau C, Sankaran M, Wright JP. 2009.
 712 Diversity has stronger top-down than bottom-up effects on decomposition. Ecology
 713 90:1073-1083. Doi: 10.1890/08-0439.1
 714 Swan CM, Palmer MA. 2004. Leaf diversity alters litter breakdown in a piedmont stream.
 715 Journal of the North America Benthological Society 23:15-28.
 716 Swan CM, Palmer MA. 2006a. Composition of speciose leaf litter alters stream detritivore
 717 growth, feeding activity and leaf breakdown. Oecologia 147:469-478. Doi: 10.1007/s00442-
 718 005-0297-8

719 Swan CM, Palmer MA. 2006b. Preferential feeding by an aquatic consumer mediates non-
720 additive decomposition of speciose leaf litter. *Oecologia* 149:107-114. Doi:
721 10.1007/s00442-006-0436-x

722 Swan C M. 2011. Consumer presence and resource diversity independently induce stability of
723 ecosystem function in a Piedmont stream. *Ecosphere* 2:136. Doi: 10.1890/ES11-00274.1

724 Swift MJ, Heal OW, Anderson JM. 1979. Decomposition in terrestrial ecosystems. Berkeley,
725 University of California press.

726 Vannote RL, Minshall GW, Cummins KW, Sedell JR, Cushing CE. 1980. The river continuum
727 concept. *Canadian Journal of Fisheries and Aquatic Sciences* 37:130-137. Doi: 10.1139/f80-
728 017

729 Villanueva VD, Albarino R, Canhoto C. 2012. Positive effect of shredders on microbial
730 biomass and decomposition in stream microcosms. *Freshwater Biology* 57:2504-2513. Doi:
731 10.1111/fwb.12023

732 Vos VC, van Ruijven J, Berg MP, Peeters ET, Berendse F. 2013. Leaf litter quality drives litter
733 mixing effects through complementary resource use among detritivores. *Oecologia* 173:269-
734 280. Doi: 10.1007/s00442-012-2588-1

735 Wallace JB, Eggert SL, Meyer JL, Webster JR. 1997. Multiple trophic levels of a forest stream
736 linked to terrestrial litter inputs. *Science* 277:102-104. Doi: 10.1126/science.277.5322.102

737 Wardle DA, Bonner KI, Nicholson KS. 1997. Biodiversity and plant litter: Experimental
738 evidence which does not support the view that enhanced species richness improves
739 ecosystem function. *Oikos* 79:247-258. Doi: 10.2307/3546010

740 Wetzel RG. 1995. Death, detritus, and energy flow in aquatic ecosystems. *Freshwater Biology*
741 33:83-89. Doi: 10.1111/j.1365-2427.1995.tb00388.x

742

TABLES

Table 1. Effects of litter quality (NPL vs. NRL), litter species identity (10 species), litterbag mesh size (fine-mesh vs. coarse-mesh litterbags), and their interactions on litter decomposition rate and microbial activity in the single-species litterbags. d.f. = degrees of freedom, %SS = percentage of type III sums of squares. *F*-values and associated *P*-values (with the respective symbols * for $P < 0.05$, ** for $P < 0.01$, and *** for $P < 0.001$) are indicated.

	d.f.	Litter decomposition		Microbial activity	
		%SS	<i>F</i> -value	%SS	<i>F</i> -value
Litter quality (LQ)	1	59.1	854.8 ***	60.0	239.7 ***
Litter species (LS)	8	12.6	22.8 ***	11.3	5.6 ***
Mesh size (MS)	1	15.8	228.9 ***	11.1	44.3 ***
LQ × MS	1	6.6	95.9 ***	1.4	5.7 *
LS × MS	8	1.6	3.0 **	1.1	0.6
Residuals	60	4.2		15.0	

Table 2. Effects of litter quality mixing (NPL-NPL, NPL-NRL and NRL-NRL), litterbag mesh size (fine-mesh *vs.* coarse-mesh litterbags), and their interactions i) on litter decomposition rate and microbial activity and ii) on relative mixture effects (RME) on litter decomposition rate (LD) and on microbial activity (MA). d.f. = degrees of freedom, %SS = percentage of type III sums of squares. *F*-values and associated *P*-values (with the respective symbols * for $P < 0.05$, ** for $P < 0.01$, and *** for $P < 0.001$) are indicated.

		Litter decomposition		Microbial activity		RME on LD		RME on MA	
	d.f.	%SS	<i>F</i> -value	%SS	<i>F</i> -value	%SS	<i>F</i> -value	%SS	<i>F</i> -value
Litter quality (LQ)	2	49.1	617.5 ***	55.5	506.2 ***	21.6	58.0 ***	19.7	51.9 ***
Mesh size (MS)	1	30.8	776.2 ***	23.1	421.8 ***	11.5	61.5 ***	13.4	70.7 ***
LQ × MS	2	6.0	75.3 ***	2.1	18.6 ***	0.8	2.3	0.0	0.1
Residuals	354	14.1		19.3		66.0		66.8	

Table 3. Effects of community-weighted mean traits (CWM), functional trait dissimilarities (FD), litterbag mesh size (fine-mesh vs. coarse-mesh litterbags), and their interactions i) on litter decomposition rate and microbial activity and ii) on relative mixture effects (RME) on litter decomposition (LD) and on microbial activity (MA). CWM1 and CWM2, and FD1 and FD2 represented the two first components of the PCAs conducted using the CWM or the FD values in [Supplementary Fig. S2](#). d.f. = degrees of freedom, %SS = percentage of type III sums of squares. *F*-values and associated *P*-values (with the respective symbols * for $P < 0.05$, ** for $P < 0.01$, and *** for $P < 0.001$) are indicated.

	d.f.	Litter decomposition		Microbial activity		RME on LD		RME on MA	
		%SS	F-value	%SS	F-value	%SS	F-value	%SS	F-value
CWM1	1	48.8	2693.2 ***	55.1	1344.1 ***	15.4	81.8 ***	23.1	129.3 ***
CWM2	1	1.6	86.4 ***	1.6	38.1 ***	0.3	1.4	0.3	1.7
FD1	1	3.7	202.3 ***	1.8	43.2 ***	0.2	0.8	0.2	1.1
FD2	1	1.9	107.1 ***	1.4	33.9 ***	3.7	19.9 ***	0.5	2.9
Mesh size (MS)	1	30.8	1704.0 ***	22.9	559.5 ***	11.5	60.8 ***	13.3	74.6 ***
CWM1 × MS	1	6.0	332.0 ***	1.9	46.5 ***	0.3	1.3	0.0	0.3
CWM2 × MS	1	0.1	3.4	0.3	7.1 **	0.7	3.6	0.1	0.3
FD1 × MS	1	0.7	40.1 ***	0.7	17.5 ***	0.4	2.0	0.2	1.2
FD2 × MS	1	0.1	5.3 *	0.1	2.7	1.6	8.3 **	0.0	0.2
Residuals	350	6.3		14.3		66.0		62.2	

Table 4. Relationships between CWM values of chemical element concentrations and relative mixture effects (RME) on litter decomposition rate and on microbial activity in fine-mesh and coarse-mesh litterbags. Adjusted R^2 and associated P -values (* $P < 0.05$, ** $P < 0.01$, *** $P < 0.001$) are indicated.

	C	N	P	Ca	K	Mg	Na	S
RME on litter decomposition								
Fine-mesh litterbags	0.19 **	0.19 **	0.28 ***	0.26 ***	0.14 **	0.21 **	0.18 **	0.09 *
Coarse-mesh litterbags	0.17 **	0.12 *	0.27 ***	0.35 ***	0.24 ***	0.30 ***	0.33 ***	0.15 **
RME on microbial activity								
Fine-mesh litterbags	0.12 *	0.02	0.13 *	0.27 **	0.26 ***	0.31 ***	0.27 ***	0.21 ***
Coarse-mesh litterbags	0.07	0.16 **	0.27 ***	0.41 ***	0.30 ***	0.43 ***	0.16 **	0.16 **

FIGURES

Fig. 1. (a) Hypothetical relationship between the different categories of litter mixtures and the litter decomposition rate of these litter mixtures. According to the mass-ratio hypothesis, the decomposition rate increases with the increase in CWM values in nutrient concentration within a litter mixture. (b) Hypothetical relationship between the different categories of litter mixtures and the intensity of litter diversity effects occurring in these litter mixtures. According to the niche complementary hypothesis, the intensity of litter diversity effects increases with the increase in FD values in nutrient concentration, which would be shown by an increase in litter decomposition rate in the present study. More precisely, this increase in decomposition rate, due to synergistic effects between the two litters in mixtures, will be higher in litter mixtures exhibiting intermediate nutrient concentration compared to litter mixtures exhibiting low or high nutrient concentrations (Fig. 1a). NPL = nutrient-poor litter, NRL = nutrient-rich litter. For both relationships, red dotted lines illustrate hypothetical increases in both litter decomposition rate and litter diversity effects mediated by macroinvertebrate presence.

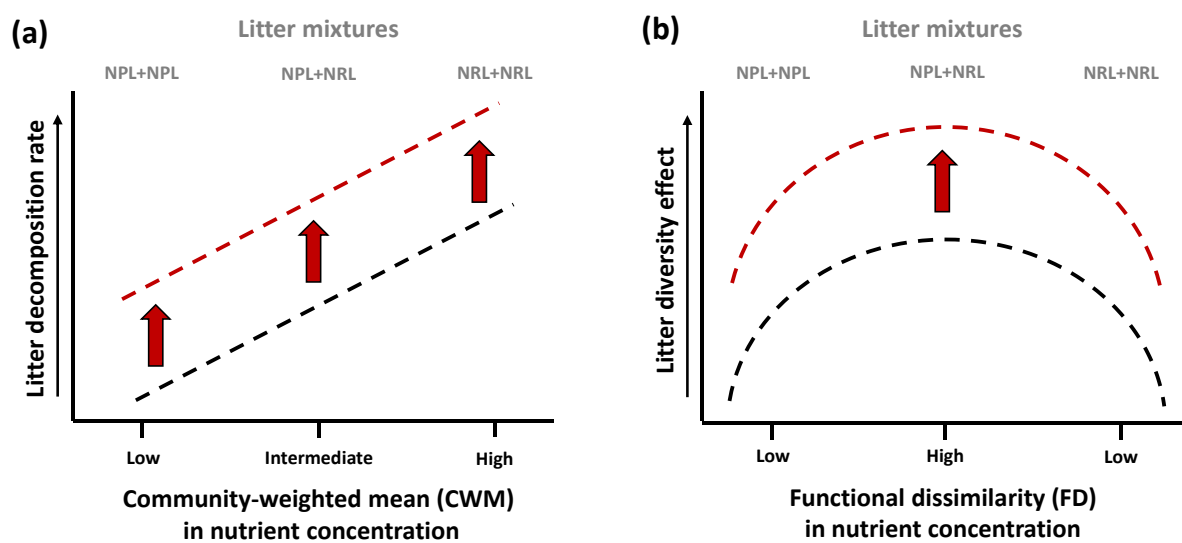


Fig. 2. Principal component analysis (PCA) based on the chemical element concentrations of the 10 litter species. Variance explained by each principal component and associated eigenvalues are shown in brackets.

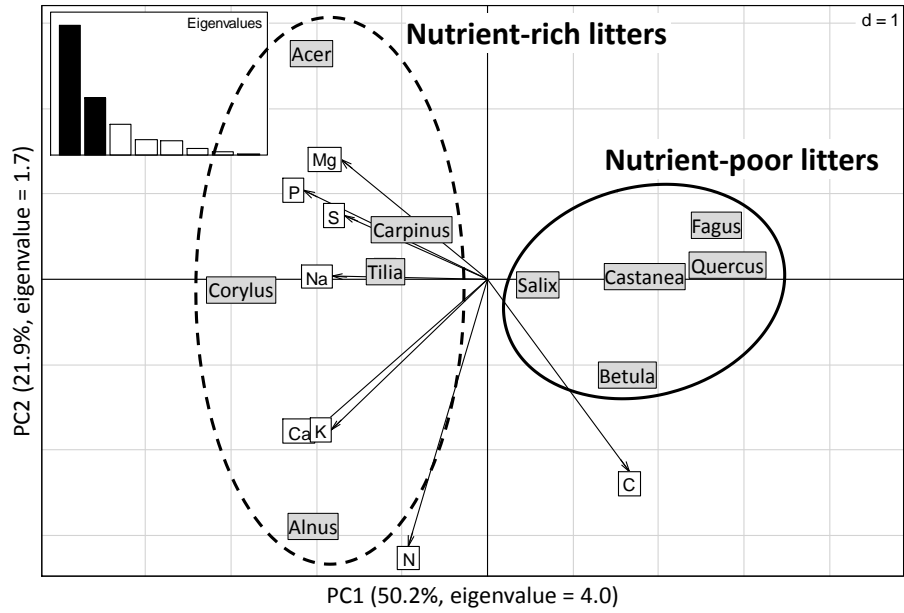


Fig. 3. Litter decomposition rate (a) and microbial activity (b) in the single-species litterbags according to litter quality (nutrient-poor litter [NPL] *vs.* nutrient-rich litter [NRL]) and to litterbag mesh size (fine-mesh *vs.* coarse mesh litterbags). Each bar represents the mean value \pm SE; n= 20. The litter decomposition rate is indicated in percent mass loss relative to the initial mass. The microbial activity is expressed as $\mu\text{g C-CO}_2$ production per h and per g of litter.

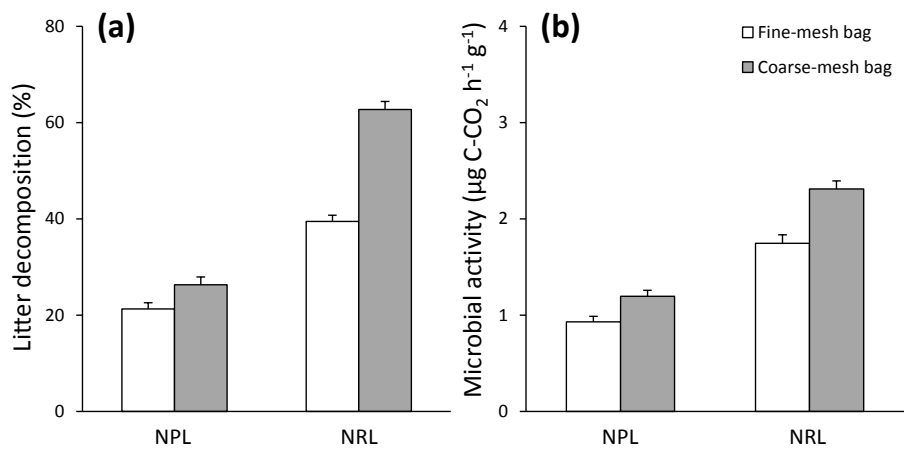


Fig. 4. Litter decomposition rate (a, b) and microbial activity (c, d) according to litter quality mixing (a, c) and to litterbag mesh size (b, d). Each bar represents the mean value \pm SE. NPL = nutrient-poor litter, NRL = nutrient-rich litter, FMB = fine-mesh litterbag, CMB = coarse-mesh litterbag. Different letters denote significant differences between treatments with $a < b < c$.

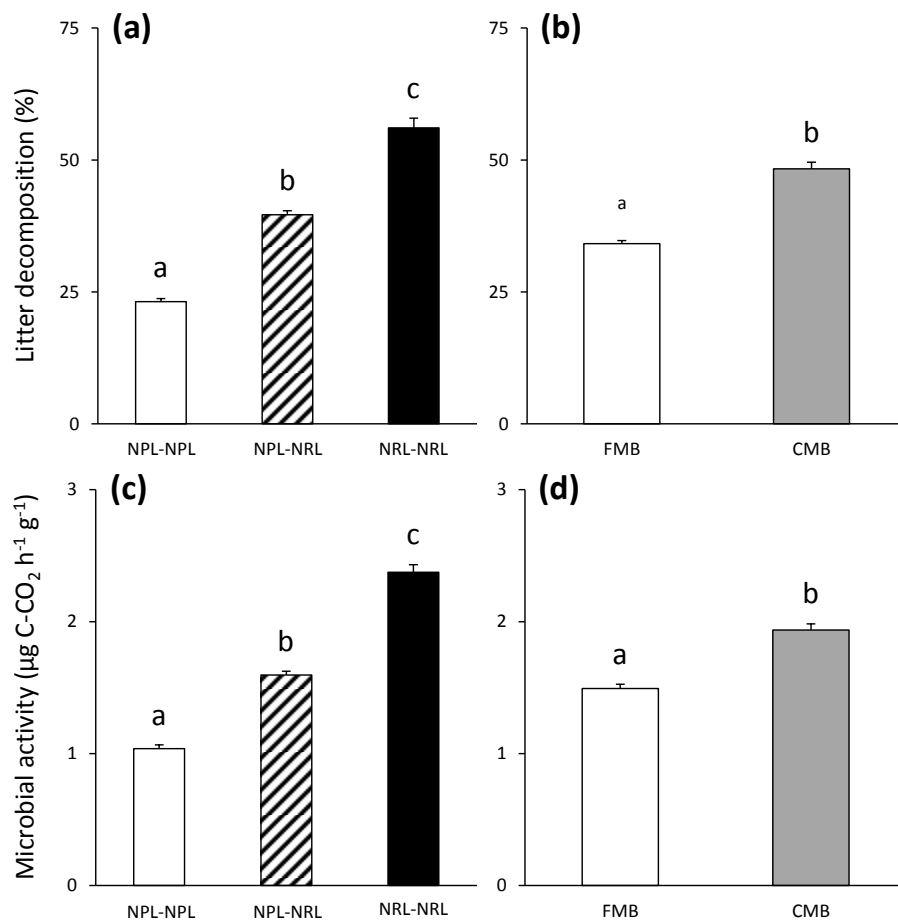


Fig. 5. Relative mixture effects (RME) on litter decomposition rate (a, b) and on microbial activity (c, d) according to litter quality mixing (a, c) and to litterbag mesh size (b, d). Each bar represents the mean value \pm SE. NPL = nutrient-poor litter, NRL = nutrient-rich litter, FMB = fine-mesh litterbag, CMB = coarse-mesh litterbag. The RME on litter decomposition rate and on microbial activity are indicated as the relative difference between the observed and the expected values from the respective single litter species treatments. Different letters denote significant differences between treatments with $a < b < c$.

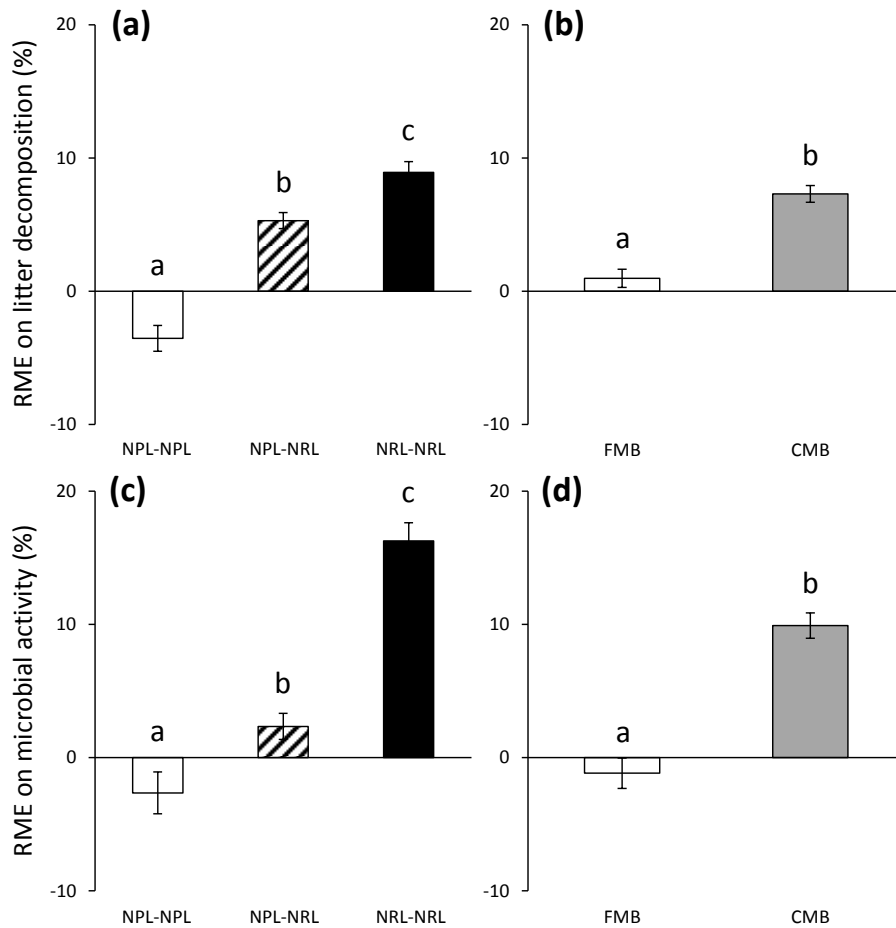


Fig. 6. Relative mixture effects (RME) on litter decomposition rate (panels a, b, c and d) and on microbial activity (panels e, f, g and h) as a function of community weighted mean traits (CWM1 and CWM2 from the PCA using the CWM values, [Supplementary Fig. S2a](#)) and functional trait dissimilarities (FD1 and FD2 from the PCA using the FD values, [Supplementary Fig. S2b](#)) in fine-mesh (white symbol) and coarse-mesh (grey symbol) litterbags. The RME on litter decomposition rate and on microbial activity are indicated as the relative difference between the observed and the expected values from the respective single litter species treatments. Significant relationships are indicated with dotted (fine-mesh litterbags) or grey (coarse-mesh litterbags) lines.

