

## RESEARCH ARTICLE

# Relationships between fungal community composition in decomposing leaf litter and home-field advantage effects

G. F. (Ciska) Veen<sup>1</sup>  | Basten L. Snoek<sup>1,2,3</sup> | Tanja Bakx-Schotman<sup>1</sup> | David A. Wardle<sup>4,5</sup>  | Wim H. van der Putten<sup>1,3</sup>

<sup>1</sup>Department of Terrestrial Ecology, Netherlands Institute of Ecology, Wageningen, The Netherlands

<sup>2</sup>Theoretical Biology and Bioinformatics, Utrecht University, Utrecht, The Netherlands

<sup>3</sup>Laboratory of Nematology, Wageningen University, Wageningen, The Netherlands

<sup>4</sup>Asian School of the Environment, Nanyang Technological University, Singapore, Singapore

<sup>5</sup>Department of Forest Ecology and Management, Swedish University of Agricultural Sciences, Umeå, Sweden

## Correspondence

G. F. (Ciska) Veen  
Email: c.veen@nioo.knaw.nl

## Funding information

Nederlandse Organisatie voor Wetenschappelijk Onderzoek, Grant/Award Number: 863.14.013

Handling Editor: Faming Wang

## Abstract

1. Increasing evidence suggests that specific interactions between microbial decomposers and plant litter, named home-field advantage (HFA), influence litter breakdown. However, we still have limited understanding of whether HFA relates to specific microbiota, and whether specialized microbes originate from the soil or from the leaf microbiome. Here, we disentangle the roles of soil origin, litter types and the microbial community already present on the leaf litter in determining fungal community composition on decomposing leaf litter and HFA.
2. We collected litters and associated soil samples from a secondary succession gradient ranging from herbaceous vegetation on recently abandoned ex-arable fields to forest representing the end stage of succession. In a greenhouse, sterilized and unsterilized leaf litters were decomposed for 12 months in soils from early- to late-successional stages according to a full-factorial design. At the end, we examined fungal community composition on the decomposing litter.
3. Fungal communities on decomposed late-successional litter in late-successional soil differed from those in early- and mid-successional stage litter and soil combinations. Soil source had the strongest impact on litter fungal composition when using sterilized litter, while the impact of litter type was strongest when using unsterilized litter. Overall, we observed HFA, as litter decomposition was accelerated in home soils. Increasing HFA did not relate to the dissimilarity in overall fungal composition, but there was increasing dissimilarity in the relative abundance of the most dominant fungal taxon between decomposing litter in home and away soils.
4. We conclude that early-, mid- and late-succession litter types did not exert strong selection effects on colonization by micro-organisms from the soil species pool. Instead, fungal community composition on decomposing litter differed substantially between litter types for unsterilized litter, suggesting that the leaf microbiome, either directly or indirectly, is an important determinant of fungal community composition on decomposing leaves. HFA related most strongly to the abundance of the most dominant fungal taxa on the decomposing litter, suggesting that HFA

This is an open access article under the terms of the Creative Commons Attribution License, which permits use, distribution and reproduction in any medium, provided the original work is properly cited.

© 2019 The Authors. *Functional Ecology* published by John Wiley & Sons Ltd on behalf of British Ecological Society.

may be attributed to some specific dominant fungi rather than to responses of the whole fungal community.

#### KEY WORDS

decomposition, fungi, ITS, microbial community, sequencing, succession

## 1 | INTRODUCTION

The home-field advantage (HFA) hypothesis predicts that plant litter decomposition is accelerated in soil from the plant community where litter originates from compared to soil from other plant communities (Ayres et al., 2009; Gholz, Wedin, Smitherman, Harmon, & Parton, 2000; Hunt, Ingham, Coleman, Elliott, & Reid, 1988; Veen, Freschet, Ordóñez, & Wardle, 2015). It has been proposed that HFA is the result of specific decomposer communities (Austin, Vivanco, González-Arzac, & Pérez, 2014; Ayres et al., 2009; Palozzi & Lindo, 2018). However, despite many studies on HFA (e.g. Freschet, Aerts, & Cornelissen, 2012; Li et al., 2017; Milcu & Manning, 2011; Veen, Sundqvist, & Wardle, 2015; Vivanco & Austin, 2008), few studies have attempted to demonstrate the involvement of specific decomposer communities (but see Chomel, Guittonny-Larchevêque, DesRochers, & Baldy, 2015; Lin et al., 2019). As a result, we still have limited understanding as to what extent litter types accumulate specific decomposer communities during decomposition, and how HFA may relate to the microbiome of decomposing leaf litter. Here, we examine how variation in HFA relates to differences in fungal microbiome composition. Thus, we aim at a better understanding of the possible role of specialist decomposers in driving decomposition processes. We tested how soil source, litter type and priority effects of microbial communities on the leaf litter shape the microbiome of decomposed litter and how this relates to HFA effects.

Previous work has shown that the composition of the species pool, environmental filtering and selection effects, as well as priority effects, all can be important drivers of community assembly and species composition in ecological communities (e.g. Berg & Smalla, 2009; Fukami et al., 2010; Grime, 2006; Raaijmakers, Paulitz, Steinberg, Alabouvette, & Moëgne-Loccoz, 2009; van der Wal, Ottosson, & de Boer, 2015; Zobel, 2016). However, little is known about their importance in shaping the litter microbiome (Austin et al., 2014; Palozzi & Lindo, 2018). The local species pool of the soil (Fierer & Jackson, 2006) determines which species are available for colonizing plant litter during composition. Meanwhile, the quality of the litter may act as environmental filter and prevent the establishment of certain micro-organisms (Kraft et al., 2015) resulting in litter-type-specific microbial communities. In addition, micro-organisms on plant litter may compete for nutrients and energy, creating a selection pressure (Austin et al., 2014; Ayres et al., 2009) that further shapes the litter community. Finally, the composition of the litter microbiome may be determined by the order in which species immigrate and colonize the litter during community assembly, leading to priority effects (Fukami, 2015). These effects cause historical

contingencies that control which species are able to colonize the litter at any moment in time (van der Wal et al., 2015).

The extent to which the litter microbiome depends on soil source, litter type and historical contingencies in the microbial community may help to understand the extent to which the various factors contribute to drive HFA effects. When litters have very distinct microbial communities in their home soils, they may experience stronger HFA effects than when communities in home and away soils have overlapping species composition. In addition, if litters take along their home phyllosphere microbiome (which can play a key role in shaping litter decomposer communities; Lin, He, Ma, Han, & Xiang, 2015; Voříšková & Baldrian, 2013) to the away soils, then HFA may be reduced because some specialist decomposers could then be introduced into the new habitat on the litter. To date, little is known about the composition of the litter microbiome in home vs. away soils in the context of HFA effects, particularly with regard to the role of the phyllosphere community. Nevertheless, previous work has indicated that plants can have specific saprophytic soil communities (Bezemer et al., 2010) and that their litters can support specific microbial communities (Keiser, Strickland, Fierer, & Bradford, 2011; Lin et al., 2019). As a result, when plant litters end up in a novel soil environment away from the plant species that has produced the litter, for example as a result of wind dispersal of dead leaves or of plant dispersal to novel environments, specific interactions between local soil decomposer communities and litter types may become disrupted resulting in reduced litter breakdown (Keiser et al., 2011; van der Putten, 2012). Alternatively, if the phyllosphere community plays a key role in driving HFA effects, introducing the litter together with its original microbiome may maintain the connection with specific decomposers and hence rates of litter breakdown.

The aim of our study was to understand how soil source, litter type and historical contingencies influence the fungal community composition of leaf litter during decomposition. In this study, we regard historical contingencies as the presence of an initial litter microbiome, which we modify by litter sterilization. In addition, we aimed to understand how variation in the composition of the fungal community on the decomposed litter might explain the observed patterns in HFA effects. In our study, we focus on fungal communities, as these are primary decomposers and many of them can degrade recalcitrant organic compounds that are abundant in plant residues (de Boer, Folman, Summerbell, & Boddy, 2005; van der Wal, Geydan, Kuyper, & Boer, 2013; Voříšková & Baldrian, 2013). We tested two hypotheses. The first hypothesis is that each of three factors, that is, soil origin, litter type and historical contingency, drives the microbiome of decomposed litters. This is based on our expectation that

each litter type would select a unique microbiome from the soil, because the HFA hypothesis predicts that HFA results from litters having specific decomposer communities. We also expected that litter microbiomes would be shaped in part by colonization from the soil in which they are incubated. As a result, when litters are placed away from their home soil they may not have access to all the decomposer specialists present in their home conditions. Finally, we expected that fungal communities initially present on litter would prevent colonization from the soil microbiome and exert a priority effect on fungal composition, compared to litters without fungi (as modified by sterilization). The second hypothesis is that the magnitude of the HFA is greater when the fungal composition on the decomposed home and away litter types is more dissimilar. Understanding which factors shape the microbiome on leaf litter and how differences between decomposed litters in their microbiomes explain the magnitude of home-field effects will help us to better understand how specific fungi may have contributed to HFA.

To test our hypotheses, we set up a reciprocal transplant experiment using soils and litters from a well-repeated successional gradient of ex-arable fields in the Netherlands (Veen, Keiser, Putten, & Wardle, 2018), in which early and mid-secondary successional stages were represented by ex-arable grasslands and late-successional stages were consisting of mixed pine and birch forest. Litters from each successional stage were left unsterilized or sterilized, in order to represent different historical contingencies of leaf microbiome, and incubated in soils from all stages in the greenhouse for 12 months, after which we determined litter mass loss and fungal community composition of the decomposed litter. Fungal community composition was measured at the end of the experiment, which integrates community development over the full course of the experimental period.

## 2 | MATERIALS AND METHODS

We set up a reciprocal litter transplant experiment in a greenhouse using soils and litters from early-, mid- and late-successional stages from a secondary successional gradient on ex-arable fields. The successional gradient was situated on nutrient-poor sandy soils in Central Netherlands, in the nature area "De Veluwe," between Ede (52°04'20" N, 5°44'12") and Wolfheze (52°00'77" N, 5°48'58"). It was part of a well-established successional gradient of old fields (e.g. Kardol, Bezemer, & van der Putten, 2006; van de Voorde, van der Putten, & Bezemer, 2011; Morriën et al., 2017). Mean annual temperature was 10.7°C and mean annual precipitation approx. 840 mm in the ten years prior to our study, that is 2003–2012 (Royal Netherlands Meteorological Institute [KNMI]). Early-successional stages consisted of early-successional grasslands, dominated by fast-growing plant species such as the grasses *Elytrigia repens* and *Lolium perenne* and the forbs *Jacobaea vulgaris* and *Myosotis arvensis*. Mid-successional stages consisted of grasslands dominated by slower growing grasses such as *Anthoxanthum odoratum* and *Agrostis capillaris* and forbs, such as *Leucanthemum vulgare* and *Plantago*

*lanceolata*. Late-successional stages consisted of mixed forest dominated by the trees *Betula pendula*, *Pinus sylvestris* and *Quercus robur* (Veen et al., 2018).

### 2.1 | Experimental design

Soil and litter material was collected at the same time as material used by Veen et al. (2018), and we followed the same sampling design. Along the successional gradient, we laid out six replicate transects that included early-, mid- and late-successional stages, resulting in 18 sampling locations (3 successional stages × 6 transects) from which litter and soil were collected. Successional stages within any given transect were at a distance of 300–2,200 m from each other, and transects were between 100 m and 7.5 km apart. For each sampling location, we used 4–8 randomly placed quadrats of 10 cm × 10 cm from an area of 2 m × 2 m. Between 25 October 2012 and 9 November 2012, in each sampling location we collected ~10 g of freshly senesced leaf litter from the soil surface, resulting in a mixed litter sample that is representative of the local plant community. Then, in each quadrat we collected the soils underneath the litter from a depth of 0–10 cm below the soil surface. Samples from quadrats within a sampling location were homogenized, both for soil and litters, so that we had one homogenized litter and soil sample for each of the 18 sampling locations, which were kept as independent replicates throughout the whole experiment.

For each of the 18 soil samples, we manually removed stones and large roots. A subsample of the soil from each location was used for analysing moisture, organic matter, carbon, nitrogen and phosphorous content, as well as nitrate and ammonium availability (see Veen et al., 2018 for details). The rest of the soil was used to fill six 1-L pots, with each of those pots receiving either sterilized or non-sterilized litter from early-, mid- or late-successional stages. The litter collected from each location was air-dried for at least 48 hr until constant weight and cut into 5-mm fragments. A subsample of litter from each location was used to determine C, N and P contents (see Veen et al., 2018 for details). Half of the remaining litter from each location was sterilized using gamma-irradiation (25 KGray) to remove micro-organisms, whereas the other half was left intact. For litter from each location, we filled three nylon mesh bags of 5 cm × 10 cm and mesh size 0.9 × 1 mm with 1 g of sterilized litter, and three with 1 g of non-sterilized litter. Mesh bags were buried in the pots according to a reciprocal transplant approach: within each transect, sterilized and non-sterilized litters originating from each successional stage (early, mid or late) were incubated in soils from all three successional stages (early, mid and late). This resulted in a total of two sterilization treatments (sterilized vs. non-sterilized) × 3 soil sources (early-/mid-/late-successional stage) × 3 litter sources (early-/mid-/late-successional stage) × 6 replicate transects = 108 pots, with 1 mesh bag per pot.

Pots were placed in a greenhouse under constant temperature and moisture conditions (see Veen et al., 2018 for details). After 12 months, mesh bags were harvested from all pots. From each bag, a subsample was collected to measure fungal community composition.

We measure fungal community composition at 12 months because this integrates community development during the course of decomposition. The rest of the litter in the bag was gently rinsed on a 0.5-mm sieve to remove soil particles, dried at 60°C until constant weight and weighed to determine mass loss (which was corrected by adding the dry weight equivalent of the subsample that was used for analysing fungal community composition).

## 2.2 | DNA extraction, amplification and sequencing

Litter subsamples were frozen in liquid nitrogen and ground. Per subsample, we used 0.25 g fresh weight to isolate DNA using the PowerSoil DNA Isolation kit according to the manufacturer's instructions (MO BIO Laboratories). For each subsample, we amplified the nuclear rDNA internal transcribed spacer (ITS) region using the fungal-specific primer pair with forward primer ITS 9 and reverse primer ITS 4 (Ihrmark et al., 2012), and added adapter sequences and 6-base-pair (bp) tags to each subsample. Polymerase chain reactions (PCRs) were performed in 25- $\mu$ l reaction mixtures and contained of each 2.5  $\mu$ l dNTP (400  $\mu$ M), 0.2  $\mu$ l of FastStart Expand High Fidelity polymerase (Roche Applied Sciences), 2.5  $\mu$ l 10 $\times$  PCR buffer with MgCl<sub>2</sub>, 10  $\mu$ mol/L of each of the two primers and 1  $\mu$ l DNA (~5–20 ng). Before performing PCRs, all samples were diluted ten times. For samples that did not produce (or produce enough) PCR products, we added 2.5  $\mu$ l BSA (4 mg/ml) to the reaction mixture (Farell & Alexandre, 2012). The temperature cycling PCR conditions involved denaturation at 95°C for 5 min, followed by 35 cycles of 95°C for 30 s and 55°C for 30 s, and then 72°C for 1 min. The final extension step was 72°C for 10 min. We confirmed the presence of expected sizes of PCR products by agarose gel electrophoresis. Products of PCRs were pooled per sample (between three–nine PCRs per sample) and purified using a QIAquick PCR Purification Kit (Qiagen, product number 28106). Quantification of DNA in the samples was done using a Fragment Analyzer (Advanced Analytical Technologies, Inc.) with an NGS Standard Sensitivity kit (DNF-473-0500), and we standardized the amount of DNA when pooling samples for sequencing by using 50 ng of DNA per sample. Samples were sequenced (Macrogen Company) on a Roche 454 automated sequencer and GS FLX system using titanium chemistry (454 Life Sciences).

## 2.3 | Bioinformatics

We analysed DNA sequences for each subsample using a Snakemake workflow (Köster & Rahmann, 2012) that follows the standard operating procedure for 454 data in MOTHUR version 1.33.2 (Schloss et al., 2009). We denoised and quality-filtered flowgrams using the “shhh.flows” command (Quince, Lanzen, Davenport, & Turnbaugh, 2011), which includes de-multiplexing and trimming of the flowgrams. We then analysed ITS sequences with ITSx (version 1.0.10). Chimeric sequences were removed using the “chimera.uchime” command (Edgar, Haas, Clemente, Quince, & Knight, 2011). Clustering of reads into OTUs was performed at a 97% identity threshold using EspritTree version 11152011 (2011). Sequences were aligned and classified

with RDP using the command `classify.seqs` (Wang, Garrity, Tiedje, & Cole, 2007) against the UNITE database version (v6\_sh\_97, release 01.08.2015). For each OTU, a consensus taxonomy was determined based on the lowest taxonomic ancestor, using the “classify.otu” command. Taxonomic classification and OTU clustering data were combined into the BIOM format (McDonald et al., 2012) for further downstream statistical analysis. Our data have been submitted to the European Nucleotide Archive (accession number: PRJEB24897; study name: ena-STUDY-NIOO-KNAW-09-02-2018-13-45:33:313-478). Prior to further analyses, we removed 15 samples that each had less than 230 reads, and all singletons from our dataset. We calculated the relative abundance for each OTU within a sample. Alpha-diversity (i.e. OTU richness and Shannon diversity index) was determined on rarefied data using the VEGAN package in R (Oksanen et al., 2018). We used FUNguild to determine the functional groups to which the OTUs in our dataset belong (Nguyen et al., 2016).

## 2.4 | Data analysis

To test how soil source, litter type (both: early-, mid-, late-successional stage) and sterilization (sterilized vs. not sterilized) affected fungal community composition, we used the Bray–Curtis dissimilarity index from the vegan package (Oksanen et al., 2018) in a PERMANOVA, with soil source, litter type, sterilization and their interactions and transect as a fixed factor. The relative abundances of the fungal OTUs were included as a multivariate response variable. To test for differences among treatments for each factor in the PERMANOVA test, we performed pairwise PERMANOVA comparisons between each combination of treatments as a post hoc analysis. *p*-Values were adjusted by the “BH” method in the `p.adjust` function in R (Benjamini & Hochberg, 1995). Fungal community data were displayed using nonmetric multidimensional scaling (NMDS) based on a Bray–Curtis distance matrix. In addition, we determined the number of reads per sample, OTU richness (total number of OTUs per sample) and OTU diversity (Shannon diversity) and we tested how soil source (early-, mid-, late-successional stage), litter type (early-, mid-, late-successional stage) and sterilization (sterilized vs. not sterilized) affected these indices using a general linear model. Here, soil source, litter type, sterilization and their interactions were tested as fixed factors, and transect as a random factor. To test how our treatments affected the relative abundance of each of the nine most dominant classes, we used general linear models with their relative abundance as a response variable, soil source, litter type, sterilization and their interactions as fixed factors, and transect as a random factor. To test how our treatments affected the relative abundance of functional groups, we used the same model, but with the relative abundance of functional groups as a response variable. To assess how taxa of fungi changed across our soil and litter treatments, we performed network analyses at the level of fungal genera. These analyses were based on Spearman rank correlations between all pairs of genera for each combination of soil source and litter type for unsterilized litter and sterilized litter separately. Correlations

between genera with  $r > 0.5$  or  $r < -0.5$  ( $p < 0.05$ ) were included in the network. We then determined clusters of fungal genera by k-means clustering on the correlation matrix. Networks were visualized using igraph in R (Csardi & Nepusz, 2006).

To determine HFA effects on litter mass loss after 12 months, we calculated the amount of additional decomposition at home (ADH) (Ayres et al., 2009; Veen, Sundqvist, et al., 2015). ADH was calculated for each litter sample and represents the percentage of additional mass loss in the home environment of that litter relative to all away environments within the same transect. We used t-tests to determine whether ADH values were significantly greater than zero for all treatments combined and for each of the treatment separately. To determine how soil source, litter type, litter sterilization and their interactions affected litter mass loss or ADH, we used a general linear model in which soil source, litter type and sterilization and their interactions were included as fixed factors and transect included as a random factor.

To test how shifts in fungal community composition relate to the observed litter mass loss, we used linear regression analyses with litter mass loss as a response variable and the NMDS scores for the fungal community as predictor variables. For this analysis, data from each pot were included as an individual observation, totalling 108 data points (i.e. 6 replicates  $\times$  3 sampling locations  $\times$  3 litter types  $\times$  2 sterilization treatments). To test how the relative abundance of dominant fungal taxa affected litter mass loss, we used a multiple regression analysis (again with each pot as an individual observation), with litter mass loss as a response variable and the relative abundance of the nine most dominant taxa (i.e. together composing >90% of the total community) as predictor variables. We selected only the nine most dominant fungi, because less abundant taxa occurred only in few plots.

To test how fungal dissimilarity between pots, both for the whole community (using Bray–Curtis dissimilarity measures) and for each of the dominant fungal taxa, related to dissimilarity in litter mass loss, we compared pairs of pots receiving home litter. For example, pots with early-succession soil and early-succession litter were compared to pots with mid-succession soil and mid-succession litter and to pots with late-succession soil and late-succession litter within the same replicate transect. Fungal community dissimilarity was calculated using the Bray–Curtis dissimilarity index, and for dominant taxa, we calculated dissimilarity in abundance as:

$$\text{dissimilarity abundance} = \left| \log_2 \left( \frac{X_{ii}}{X_{jj}} \right) \right| \quad (1)$$

where  $X_{ii}$  is the relative abundance of a dominant fungal taxon  $X$  in soil  $I$  incubated with litter  $i$ , and  $X_{jj}$  is the relative abundance of fungal taxon  $X$  incubated in soil  $J$  incubated with litter  $j$ . To test how fungal community dissimilarity affected ADH, we used a linear regression with ADH as a response variable, and Bray–Curtis distance, sterilization and their interaction as predictor variables. To test how dissimilarity in the relative abundance of dominant fungal taxa related to ADH, we first used a multiple regression

analyses with ADH as the response variable, and dissimilarity in abundance of the nine most dominant taxa as predictor variables. When dissimilarity in abundance of one of the abundant dominant taxa significantly explained variation in ADH, we performed a linear regression analysis with ADH as the response variable and dissimilarity in abundance, sterilization treatment and their interaction as predictor variables. For both the analyses on fungal community dissimilarity and on the dissimilarity in the abundance of the dominant fungal taxa, we only included data from litter incubated in “home” soils, that is in soils from the successional stage where they were sourced from. As a result, we only used pots with early-successional litters and soils, pots with mid-successional litters and soils, and pots with late-successional litters and soils. This is because ADH is calculated per successional stage, and not for each individual transplant. In total, this resulted in 3 pots per replicate gradient  $\times$  6 gradients  $\times$  2 sterilization treatments = 36 data points. Within each replicate gradient, we then compared fungal composition in a given litter sample to the composition in the litter samples from the other two pots to calculate the average dissimilarity in community composition or relative abundance, respectively.

To test how litter and soil chemical properties differed between successional stages, we used general linear models for each litter or soil property, using successional stage (early, mid, late) as a fixed factor, and transect as a random factor.

All analyses were performed in R version 3.4 (R Development Core Team, 2013). We used the vegan package for multivariate statistics (Oksanen et al., 2018) and the lmerTest package for general linear models (Kuznetsova, Brockhoff, & Christensen, 2017). Data were tested for normality using a Q-Q plot and for homogeneity of variances with a Levene's test.

## 3 | RESULTS

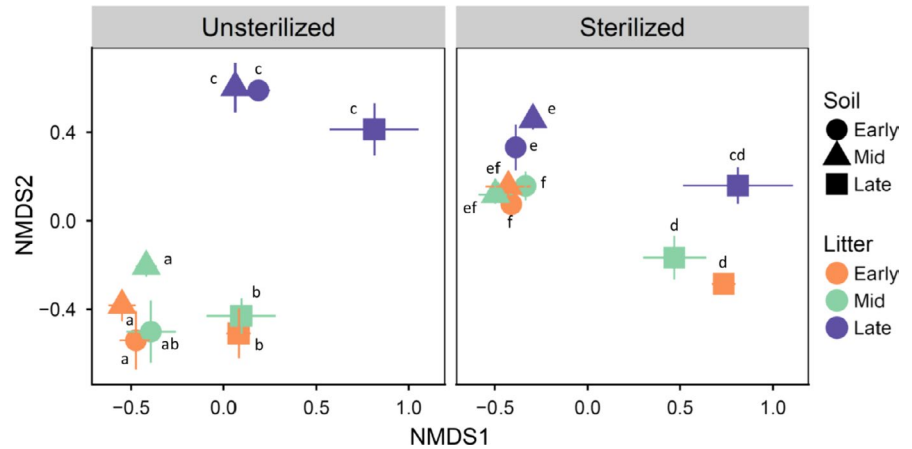
### 3.1 | Litter and soil chemical properties

Litter and soil chemical properties differed between successional stages (Supporting Information Tables S1 and S2). Soil organic matter content, ammonium concentration, total N and N:P ratio, and litter total C, C:P ratio and N:P were higher in late- than in early- and mid-successional stages, while pH, total soil P and total litter P were lower (Supporting Information Tables S1 and S2). Soil moisture and nitrate were highest in late- and lowest in mid-successional stages (Supporting Information Tables S1 and S2). Total soil C and C:N ratio, and litter total N and C:N ratio did not differ between successional stages (Supporting Information Tables S1 and S2).

### 3.2 | Fungal community composition

Soil source, litter type, litter sterilization and their interactions explained >35% of the variation in litter fungal community composition at the end of the litter incubation period (Figure 1, Table 1). The factor that explained the highest proportion of variation in fungal

**FIGURE 1** NMDS plot of fungal community composition in unsterilized litter (left panel) and sterilized litter (right panel) at the end of the experiment. Litter source is indicated by different colours, and soil source is indicated by different shapes. Different letters at the symbols indicate that treatments are significantly different from each other at  $p < 0.05$  as tested in pairwise PERMANOVAs analyses



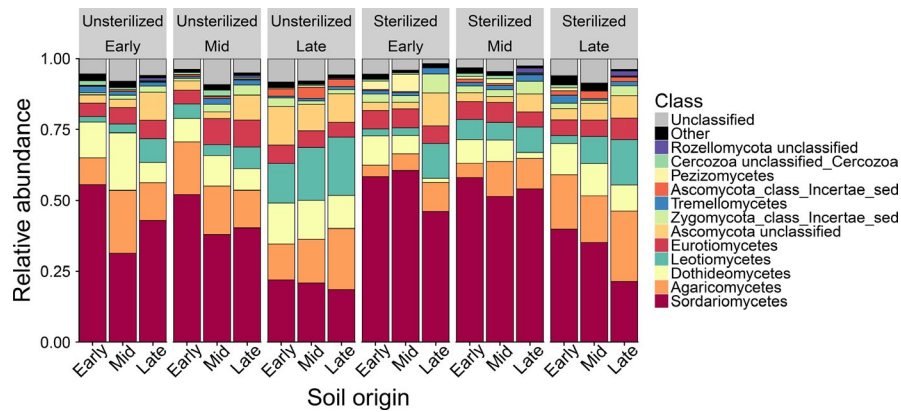
**TABLE 1** The influence of soil source (early-, mid-, late-successional stage), litter type (early-, mid-, late-successional stage), sterilization (no, yes) and their interactions on fungal community composition in the litter as tested using a PERMANOVA. Values in bold represent significant effects with  $p < 0.05$

Factor	<i>df</i>	<i>F</i>	<i>p</i>	% explained variation
Replicate transect	5, 61	2.6520	<0.001	11.5
Litter type	2, 61	4.1988	<0.001	7.3
Soil source	2, 61	5.8501	<0.001	10.1
Sterilization	1, 61	6.2662	<0.001	5.4
Litter type × soil source	4, 61	1.2619	0.048	4.4
Litter type × sterilization	2, 61	1.7286	0.003	3.0
Soil source × sterilization	2, 61	1.7023	0.002	2.9
3-way interaction	4, 61	0.7509	0.990	2.6

composition was soil source (Table 1). On the first NMDS axis, fungal composition on litter incubated in late-successional soils was different from that on litter incubated in early- and mid-successional soils (Figure 1). A significant two-way interaction between litter sterilization and soil source indicated that the impact of soil source was stronger for sterilized than for unsterilized litter on the first NMDS axis (Table 1, Figure 1). Litter type explained the second-highest proportion of total variation in fungal composition (Table 1). On the second NMDS axis, fungal communities on late-successional litter differed from those on early- and mid-successional litters (Figure 1). This effect was more pronounced in unsterilized than in sterilized litter, as revealed by a two-way interaction between litter type and litter sterilization. Litter sterilization explained a smaller proportion of total variation in litter fungal community composition on leaf litter, but its effect was still highly statistically significant (Table 1;

Figure 1). Results were the same when we performed our analyses on rarefied data, used different dissimilarity indices or used data at the genus level.

The average number of reads per sample was  $1,235 \pm 55$  (mean  $\pm$  SE, with a maximum 3,013 of and a minimum of 232) and did not differ between treatments (Supporting Information Table S3). The average number of OTUs was  $148 \pm 5.5$  (with a maximum of 291 and a minimum of 46) and was lower on sterilized ( $131 \pm 6.6$ ) than on unsterilized ( $167 \pm 7.9$ ) litter (Supporting Information Table S3). The average Shannon diversity index was lower for sterilized ( $3.59 \pm 0.07$ ) than for unsterilized ( $3.94 \pm 0.08$ ) litter (Supporting Information Table S3). The fungal classes with the highest relative abundance across all samples belonged to the classes Sordariomycetes (Ascomycota) and Agaricomycetes (Basidiomycota), which together made up 35%–70% of the total fungal community depending on treatment combination (Figure 2, Supporting Information Table S4). The relative abundance of several fungal classes was affected by soil source, litter type and sterilization treatment; however, these differences were not always reflected in post hoc analyses (Supporting Information Table S4). Sordariomycetes were higher in early- than late-successional soils and lowest on late-successional litters, and were higher on sterilized than unsterilized litter (Figure 2, Supporting Information Table S4). Dothideomycetes (Ascomycota) were higher in mid- than in late-successional soils, were generally higher on late- than on early- and mid-successional litters, and were lower on sterilized than on unsterilized litter. Leotiomycetes (Ascomycota) were highest in late-successional soils and on late-successional litters, particularly when unsterilized. The relative abundance of an unclassified taxon of Ascomycota was higher on late-successional litters than on early- and mid-successional litters, but only in early- and mid-successional soils, as was indicated by the significant interaction between soil source and litter type. Also, on late-successional litters the unclassified Ascomycota were more abundant on unsterilized than on sterilized litter, but this was not the case for the other litter types, as was indicated by the interaction between litter type and sterilization. The abundance of Zygomycota was higher on sterilized than unsterilized litter and was lower on



**FIGURE 2** Relative abundance for the most abundant fungal classes in all different litters at the end of the experiment. Different panels represent different litter types, indicated by the names on the panels: Unsterilized = unsterilized litter, Sterilized = sterilized litter, Early = early-successional litter, Mid = mid-successional litter and Late = Late-successional litter. Within each panel, individual bars represent different soil sources: Early = early-successional soil, Mid = mid-successional soil and late = Late-successional soil

late-than on early- and mid-successional litters, but both effects occurred in late-successional soils only (Supporting Information Table S4).

Similar to the response of fungal classes, the relative abundance of fungal genera was determined by litter type, soil source, sterilization and their interactions (Supporting Information Table S5). For unsterilized litters, network analyses indicated that specific clusters of fungi occurred on early- and mid-successional litter types, while others occurred on late-successional litters (Supporting Information Figures S1 and S2; for details on clusters and genera see Supporting Information Table S5). For sterilized litters, there were fewer connections than for unsterilized litters, and specific clusters of fungi occurred in early- and mid-successional soils, while others occurred mostly in late-successional soils (Supporting Information Figures S3 and S4, for details on clusters and genera see Supporting Information Table S5). On average, 68% of our OTUs could be assigned to a fungal guild. This percentage was higher for sterilized (74%) than for unsterilized (62%) litter ( $F_{1,63} = 25.11, p < 0.001$ ), but did not differ between soil sources ( $F_{2,63} = 0.57, p = 0.570$ ) or litter types ( $F_{2,62} = 2.33, p = 0.106$ ). On average, 52% of the identified fungi were potential saprotrophs. When correcting for the percentage of fungi that could be identified, we found that the relative abundance of potential saprotrophs was affected by soil source, litter type, sterilization and their interactions (Supporting Information Figure S5). The relative abundance of saprotrophs was higher in sterilized than in unsterilized litter ( $F_{1,64} = 19.00, p < 0.001$ ). In addition, the relative abundance of saprotrophs was affected by a three-way interaction between soil source, litter type and sterilization ( $F_{4,62} = 2.56, p = 0.047$ ) and a two-way interaction between soil source and litter type ( $F_{4,63} = 6.18, p < 0.001$ ). These interactions emerged because the relative abundance of saprotrophs on mid-successional litter in late-successional soils was higher than on early-successional litter, while there was generally no difference between litter types for the other soil sources. Further, the relative abundance of saprotrophs was higher on sterilized late-successional litter in mid-successional soil than on sterilized mid-successional soils, while there was no

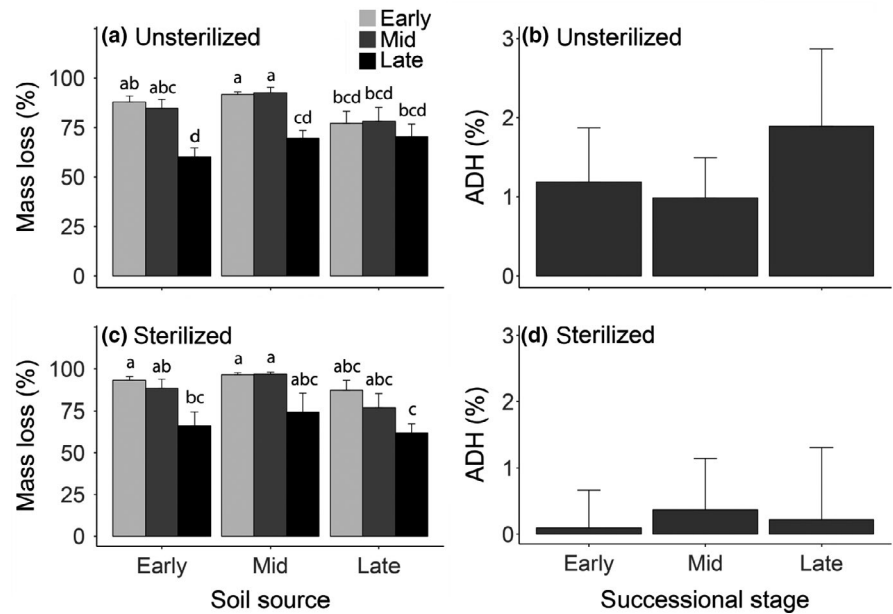
difference between litter types in mid-successional soils for unsterilized litters (Supporting Information Figure S5).

### 3.3 | Litter mass loss and home-field advantage

Litter mass loss was affected by the main effects of litter type and soil source. Litter from late-successional stages decomposed slower than litter from mid- and early-successional stages, and litter decomposed slower in soils from late-successional stages than in soils from early- and mid-successional stages (Supporting Information Table S6, Figure 3a,c). Sterilization did not affect litter mass loss, and there were no interactive effects between sterilization, litter source and soil source (Supporting Information Table S6, Figure 3a,c). There were significant relationships between litter mass loss and sample scores on the first ( $F_{1,81} = 19.40, p < 0.001, R^2 = 0.19$ ) and second NMDS axis ( $F_{1,81} = 5.45, p = 0.022, R^2 = 0.05$ ), indicating that litter mass loss rates were related to fungal community composition after 12 months of decomposition. Multiple regression analyses with the nine most dominant fungal taxa showed that there was a negative relationship of litter mass loss with the relative abundance of an unclassified Ascomycota ( $t_{71} = -3.91, p < 0.001, R^2 = 0.29$ ), but not with any of the other dominant taxa or groups of taxa.

Overall, home-field effects were positive ( $t_{33} = 2.44, p = 0.020$ ). This indicates a general advantage for the different litter types of being decomposed in their home soil compared to the different litter types decomposed in away soils (Figure 3b,d). There was a marginally non-significant effect of litter sterilization, but home-field effects tended to be lower for sterilized than for unsterilized litters ( $F_{1,23} = 3.87; p = 0.061$ ; Figure 3b,d). However, home-field effects did not differ between successional stages ( $F_{2,23} = 2.67, p = 0.768$ ; Figure 3b,d) and there was no interaction between successional stage and sterilization treatment ( $F_{2,23} = 0.28, p = 0.761$ ). There was no significant relationship between the magnitude of home-field effects and the Bray-Curtis dissimilarity in fungal composition ( $F_{1,24} < 0.01, p = 0.960$ ; Figure 4a), irrespective of litter sterilization treatment ( $F_{1,24} = 0.34, p = 0.566$ ; Figure 4a). Multiple regression

**FIGURE 3** Mean litter mass loss (%)  $\pm$ SE and additional decomposition at home (ADH; %)  $\pm$ SE for unsterilized litter (a, b) and sterilized litter (c, d). In panels (a) and (c), bars with different types of grey represent different litter types: Early = early-successional litter, Mid = mid-successional litter and Late = late-successional litter. Bars topped by different letters are significantly different at  $p < 0.05$  as according to a Tukey post hoc test. Litter mass loss data were tested using a general linear model with soil source, litter type and sterilization as fixed factors (Table S3). ADH was tested using a general linear model with successional stage and sterilization as a fixed factor



analyses showed that dissimilarity in the relative abundance of only one taxon could explain the magnitude of home-field effects, that is Sordariomycetes (Ascomycota), which was the most abundant taxon in the whole community.

## 4 | DISCUSSION

The aim of our study was to understand how soil source, litter type and historical contingencies of the litter microflora affected fungal community composition on decomposed leaf litter and whether differences in fungal composition between litters could explain the magnitude of home-field advantage (HFA) for litter decomposition. By analysing fungal community composition on decomposed litter, we attempted to obtain an integrated value corresponding with our integrated analysis of HFA during that same period. We found that the impact of soil source was stronger on sterilized than on unsterilized litter, while the impact of litter type on fungal composition was stronger for unsterilized than for sterilized litter. We found that not the dissimilarity in fungal composition, but the abundance of the most dominant fungal taxon corresponded with greater HFA.

### 4.1 | The impact of soil source, litter type and sterilization on litter fungal composition

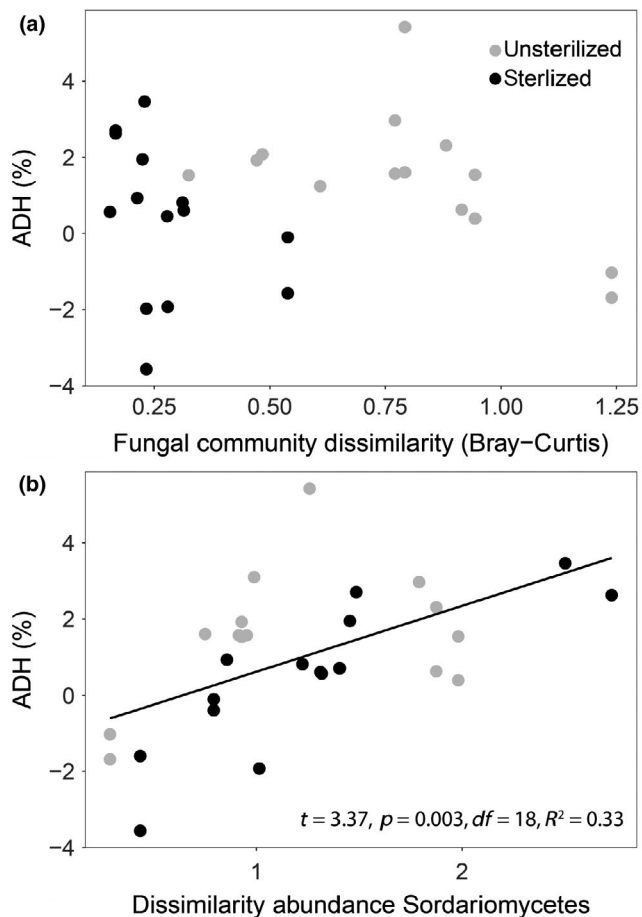
In line with our first hypothesis, we found that soil source, litter type, sterilization and their interactions all explained some of the total variation in fungal community composition on the decomposed litter. Soil source, and therefore the species pool in the soil, emerged as the most important driver of fungal composition on leaf litter. Large changes in soil fungal composition are known to occur as succession proceeds in this chronosequence (Hannula et al., 2017). However, litter type also had important effects on fungal composition, in that

early-, mid- and late-successional decomposed litter generally had different fungal communities. This may be the result of differences in chemical quality of the litter, as can be expected for old-field succession (Cortez, Garnier, Pérez-Harguindeguy, Debussche, & Gillon, 2007; Milcu & Manning, 2011; Wardle, 2002), which may act as a filter for fungal colonization (Kraft et al., 2015). Therefore, in line with knowledge on the rhizosphere composition (Berg & Smalla, 2009; Mommer et al., 2018; Raaijmakers et al., 2009) litter types can accumulate a unique microbiome (Aneja et al., 2006; Conn & Dighton, 2000; Fanin, Hättenschwiler, & Fromin, 2014).

The effect of soil source was stronger for sterilized than for unsterilized litter, even after the 12 months of litter incubation in our experiment. This indicates that the phyllosphere microbiome on unsterilized decomposed litter may have provided resistance against invasion by fungi from the species pool in the soil (Mallon, Elsas, & Salles, 2015). Thus, under natural conditions, where plant litters are unsterilized, the leaf microbiome exerts a priority effect by influencing colonization of the decomposing litter by fungi from the soil (Fukami, 2015; Fukami et al., 2010; van der Wal et al., 2015) and can drive fungal community composition (Lin et al., 2015; Voříšková & Baldrian, 2013). Many previous controlled experiments studying HFA have used sterilized litter because elimination of litter communities allows for testing the local adaptation of the soil community (Palozzi & Lindo, 2018). Our comparison of sterilized and unsterilized litter suggests that specific decomposers may be imported together with the litter itself, at least to some extent.

Our findings are consistent with recent work showing that endophytes in phyllosphere can cause priority effects and play a key role in shaping decomposer communities on leaf litter (Lin et al., 2015; Voříšková & Baldrian, 2013). As such, it has been shown that the phyllosphere microbiome (including both endo- and epiphytes) can differ between plant species (Persoh, 2013) and genotypes (Wagner et al., 2016) and can depend on plant functional traits (Kembel et





**FIGURE 4** (a) Fungal community dissimilarity (Bray–Curtis) and (b) the dissimilarity in abundance of Sordariomycetes (calculated as the logarithm of ratio of the relative abundance home and away) plotted against the percentage of additional decomposition at home (ADH). Points represent individual litter samples. We only included data from litter samples incubated in soils collected from the same successional stage (“home combinations”), because ADH is calculated per successional stage, and not for each individual litter transplant. Relationships are tested using linear regression analyses

al., 2014) and environmental conditions (Kraft et al., 2015). Hence, shifts in plant composition and trait spectra along the successional gradient in our study (Morriën et al., 2017) may shape differences in phyllosphere microbiome and thus the decomposer community on leaf litter. Future work could further advance this topic by focusing on the variation in phyllosphere communities at litter fall and the role of these communities in litter breakdown and HFA (Austin et al., 2014).

#### 4.2 | Relationship between fungal dissimilarity and home-field advantage

We found that litter mass loss differed between the litter types from the three successional stages and between the soils in which the litters were incubated, but this was not reflected in shifts in HFA between the successional stages. Overall, the litters in our experiment

experienced HFA and the decomposed litters had unique fungal communities (Figure 1). This is in line with a recent study (Lin et al., 2019); however, in contrast to our second hypothesis, we found that the magnitude of HFA effects did not relate to the dissimilarities between litters of the overall fungal community. Instead, variation in litter mass loss related to variation in fungal composition of the decomposed litter. It could be that in our work HFAs were weak because we sourced litter from mixed plant communities (Veen, Freschet, et al., 2015; Veen, Sundqvist, et al., 2015), and hence that relationships between fungal composition and HFA are weak as well, so that they cannot be detected easily. Alternatively, biotic and abiotic environmental factors other than those that we have measured might play a role in determining HFA (Palozzi & Lindo, 2018). For example, larger soil fauna (Milcu & Manning, 2011) or soil bacteria could also be important drivers of HFA, but these were not included in our greenhouse study or molecular analyses, respectively. Alternatively, variation in soil abiotic conditions, differences in litter quality or the presence of other soil organisms can interact with fungal decomposer communities to drive decomposition processes. This could modify the effect of specific interactions (due to specific decomposer communities or specific soil abiotic conditions) between litter and soil sources and therefore may have contributed to explain HFA effects (Austin et al., 2014; Veen et al., 2018). Therefore, there is a need to further elucidate the role of variation in soil physiochemical conditions and biotic interactions in the soil for driving HFA effects (Palozzi & Lindo, 2018).

Although there were no clear relationships between variation in fungal composition and HFA, we found an almost significant reduction in HFA on sterilized litter (Figure 3). This finding suggests that the removal of the phyllosphere community present at the time of litter collection eliminated specialists that otherwise cause HFA (Austin et al., 2014). Also, our results showed that increasing dissimilarity in the most abundant fungal taxon, that is, Sordariomycetes, between home and away soils corresponded with stronger HFA. This class of fungi contains species that perform a wide variety of functions in the soil, ranging from plant parasitism to decomposition of recalcitrant organic material (Nguyen et al., 2016). It was the most abundant fungal class in our samples, suggesting that fungi may play a role in HFA if they are common enough. To further unravel the impact of individual fungal taxa on HFA will require disentangling the fungal community in more detail and performing transplant experiments with targeted fungal taxa or entire fungal groups.

In our experiment, we measured litter mass loss and fungal composition after 12 months of litter incubation. This informed on the changes that occurred in the fungal community integrated over the experimental period, both when we removed and did not remove the microbes from the phyllosphere at the start of the experiment. However, fungal composition and the strength of litter selection effects may vary during the process of litter breakdown, because fungal composition will be subject to succession during decomposition (Aneja et al., 2006; van der Wal et al., 2015; Voříšková & Baldrian, 2013). Therefore, we recognize that measuring the fungal community throughout the full course of the 12 months might help

us to increase understanding of the role of microbes in driving litter mass loss and HFA effects at the stages where most of the litter breakdown occurs. Future work should perform multiple destructive harvests and measure the temporal dynamics of HFA (Fanin, Fromin, & Bertrand, 2016) and litter microbiomes (Aneja et al., 2006; Voříšková & Baldrian, 2013). Also, sampling fungal communities in the phyllosphere before or at the start of litter breakdown would allow to identify to what extent phyllosphere communities drive the priority effects that we observed (Fukami et al., 2010; van der Wal et al., 2015) and contribute to explain HFA effects. Finally, integrating measurements on fungal biomass, fungal activity and species composition (see, e.g. Lin et al., 2019; Morriën et al., 2017) may be essential to obtain a comprehensive understanding of the role of litter microbiomes for driving HFA (Props et al., 2017; Zhang et al., 2017).

## 5 | CONCLUSIONS

We show that leaf litters at the end of a 12-month incubation period have unique decomposer communities, but particularly on unsterilized litter. This implies a priority effect, as fungi from the soil appeared more successful in colonizing sterilized than unsterilized litter. Our findings suggest that leaf litters select for fungal communities early on in the decomposition process or in their phyllosphere and those communities prevent invasion from the soil. Dissimilarity in the entire litter microbiome of the decomposed litter could not explain observed HFA effects. Instead, we found that the most abundant fungal taxon correlated with the magnitude of HFA. Therefore, we suggest that not the entire microbiome, but specific fungal species or taxa may be important for generating HFA effects. Our results show that decomposed plant litters can build up unique decomposer communities, which may have functional consequences for litter breakdown and may play a role in driving HFA. This finding helps to unravel the mechanisms that can drive HFA (Austin et al., 2014; Palozzi & Lindo, 2018). In addition, our study suggests that the presence of specific decomposer taxa could feed back to plant growth and performance (Wardle et al., 2004) via accelerated litter breakdown and thus soil nutrient cycling. It will therefore be important to integrate the role of specific decomposers for litter breakdown into research on plant-soil feedback to further disentangle functional consequences of HFA (Kardol, Veen, Teste, & Perring, 2015; Zhang, Van der Putten, & Veen, 2016).

## ACKNOWLEDGEMENTS

We are grateful to "Vereniging Natuurmonumenten" for permission to collect litter and soil samples. We thank Annemieke van der Wal for advice on molecular analyses. We thank Naomi te Braak, Raquel Luján Soto, Rebecca Pas and Eefje Sanders for assistance in the field and greenhouse. Iris Chardon, Carolin Weser and Margareta Söderström performed the chemical analyses and Naomi te Braak assisted with the molecular work. We thank Mattias den Hollander and Fleur Gahwens-Bruning for their assistance with bioinformatics.

G.F.V. was supported by a VENI grant (no. 863.14.013) from the Netherlands Organisation for Scientific Research (NWO). The contribution of D.A.W. was supported by a Wallenberg Scholars award. This is NIOO publication 6716.

## AUTHORS' CONTRIBUTIONS

G.F.V., D.A.W. and W.H.P. designed the experiment. G.F.V. carried out the experiment. G.F.V. and T.B.-S. performed the molecular work. G.F.V. and B.L.S. analysed the data. G.F.V. led the writing of the manuscript. All authors contributed to writing and revising of the manuscript and gave final approval for publication.

## DATA AVAILABILITY

Litter mass loss and environmental data are deposited in the Dryad Digital Repository <https://doi.org/10.5061/dryad.527d3f9> (Veen, Snoek, Bakx-Schotman, Wardle, & Putten, 2019). Sequencing data have been submitted to the European Nucleotide Archive (accession number: PRJEB24897; study name: ena-STUDY-NIOO-KNA W-09-02-2018-13:45:33:313-478). Our R code is available upon request.

## ORCID

G. F. (Ciska) Veen  <https://orcid.org/0000-0001-7736-9998>

David A. Wardle  <https://orcid.org/0000-0002-0476-7335>

## REFERENCES

- Aneja, M. K., Sharma, S., Fleischmann, F., Stich, S., Heller, W., Bahnweg, G., ... Schloter, M. (2006). Microbial colonization of beech and spruce litter—Influence of decomposition site and plant litter species on the diversity of microbial community. *Microbial Ecology*, *52*, 127–135. <https://doi.org/10.1007/s00248-006-9006-3>
- 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 Phytologist*, *204*, 307–314. <https://doi.org/10.1111/nph.12959>
- Ayres, E., Steltzer, H., Simmons, B. L., Simpson, R. T., Steinweg, J. M., Wallenstein, M. D., ... Wall, D. H. (2009). Home-field advantage accelerates leaf litter decomposition in forests. *Soil Biology & Biochemistry*, *41*, 606–610. <https://doi.org/10.1016/j.soilbio.2008.12.022>
- Benjamini, Y., & Hochberg, Y. (1995). Controlling the false discovery rate: A practical and powerful approach to multiple testing. *Journal of the Royal Statistical Society. Series B, (Methodological)*, *57*(1), 289–300. <https://doi.org/10.1111/j.2517-6161.1995.tb02031.x>
- Berg, G., & Smalla, K. (2009). Plant species and soil type cooperatively shape the structure and function of microbial communities in the rhizosphere. *FEMS Microbiology Ecology*, *68*, 1–13. <https://doi.org/10.1111/j.1574-6941.2009.00654.x>
- Bezemer, T. M., Fountain, M. T., Barea, J. M., Christensen, S., Dekker, S. C., Duyts, H., ... van der Putten, W. H. (2010). Divergent composition but similar function of soil food webs beneath individual plants: Plant species and community effects. *Ecology*, *91*, 3027–3036. <https://doi.org/10.1890/09-2198.1>

- 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. <https://doi.org/10.1007/s10021-015-9880-y>
- Conn, C., & Dighton, J. (2000). Litter quality influences on decomposition, ectomycorrhizal community structure and mycorrhizal root surface acid phosphatase activity. *Soil Biology and Biochemistry*, *32*, 489–496. [https://doi.org/10.1016/S0038-0717\(99\)00178-9](https://doi.org/10.1016/S0038-0717(99)00178-9)
- Cortez, J., Garnier, E., Pérez-Harguindeguy, N., Debussche, M., & Gillon, D. (2007). Plant traits, litter quality and decomposition in a Mediterranean old-field succession. *Plant and Soil*, *296*, 19–34. <https://doi.org/10.1007/s11104-007-9285-6>
- Csardi, G., & Nepusz, T. (2006). *The igraph software package for complex network research* (p. 1695). Complex Systems: InterJournal.
- de Boer, W., Folman, L. B., Summerbell, R. C., & Boddy, L. (2005). Living in a fungal world: Impact of fungi on soil bacterial niche development. *FEMS Microbiology Reviews*, *29*, 795–811. <https://doi.org/10.1016/j.femsre.2004.11.005>
- Edgar, R. C., Haas, B. J., Clemente, J. C., Quince, C., & Knight, R. (2011). UCHIME improves sensitivity and speed of chimera detection. *Bioinformatics*, *27*, 2194–2200. <https://doi.org/10.1093/bioinformatics/btr381>
- Fanin, N., Fromin, N., & Bertrand, I. (2016). Functional breadth and home-field advantage generate functional differences among soil microbial decomposers. *Ecology*, *97*, 1023–1037.
- Fanin, N., Hättenschwiler, S., & Fromin, N. (2014). Litter fingerprint on microbial biomass, activity, and community structure in the underlying soil. *Plant and Soil*, *379*, 79–91. <https://doi.org/10.1007/s11104-014-2051-7>
- Farell, E. M., & Alexandre, G. (2012). Bovine serum albumin further enhances the effects of organic solvents on increased yield of polymerase chain reaction of GC-rich templates. *BMC Research Notes*, *5*, 257–257. <https://doi.org/10.1186/1756-0500-5-257>
- Fierer, N., & Jackson, R. B. (2006). The diversity and biogeography of soil bacterial communities. *Proceedings of the National Academy of Sciences of the United States of America*, *103*, 626–631. <https://doi.org/10.1073/pnas.0507535103>
- 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. *Journal of Ecology*, *100*, 619–630. <https://doi.org/10.1111/j.1365-2745.2011.01943.x>
- Fukami, T. (2015). Historical contingency in community assembly: Integrating niches, species pools, and priority effects. *Annual Review of Ecology, Evolution, and Systematics*, *46*, 1–23. <https://doi.org/10.1146/annurev-ecolsys-110411-160340>
- Fukami, T., Dickie, I. A., Wilkie, J. P., Paulus, B. C., Park, D., Roberts, A., ... Allen, R. B. (2010). Assembly history dictates ecosystem functioning: Evidence from wood decomposer communities. *Ecology Letters*, *13*, 675–684. <https://doi.org/10.1111/j.1461-0248.2010.01465.x>
- 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. *Global Change Biology*, *6*, 751–765. <https://doi.org/10.1046/j.1365-2486.2000.00349.x>
- Grime, J. P. (2006). *Plant strategies, vegetation processes, and ecosystem properties*. John Wiley & Sons.
- Hannula, S. E., Morriën, E., de Hollander, M., van der Putten, W. H., van Veen, J. A., & de Boer, W. (2017). Shifts in rhizosphere fungal community during secondary succession following abandonment from agriculture. *The ISME Journal*, *11*, 2294. <https://doi.org/10.1038/ismej.2017.90>
- Hunt, H. W., Ingham, E. R., Coleman, D. C., Elliott, E. T., & Reid, C. P. P. (1988). Nitrogen limitation of production and decomposition in prairie, mountain meadow and pine forest. *Ecology*, *69*, 1009–1016. <https://doi.org/10.2307/1941256>
- Ihrmark, K., Bodeker, I. T. M., Cruz-Martinez, K., Friberg, H., Kubartova, A., Schenck, J., ... Lindahl, B. D. (2012). New primers to amplify the fungal ITS2 region – evaluation by 454-sequencing of artificial and natural communities. *FEMS Microbiology Ecology*, *82*, 666–677. <https://doi.org/10.1111/j.1574-6941.2012.01437.x>
- Kardol, P., Bezemer, T. M., & van der Putten, W. H. (2006). Temporal variation in plant-soil feedback controls succession. *Ecology Letters*, *9*, 1080–1088. <https://doi.org/10.1111/j.1461-0248.2006.00953.x>
- Kardol, P., Veen, G. F., Teste, F. P., & Perring, M. P. (2015). Peeking into the black box: A trait-based approach to predicting plant-soil feedback. *New Phytologist*, *206*, 1–4. <https://doi.org/10.1111/nph.13283>
- Keiser, A. D., Strickland, M. S., Fierer, N., & Bradford, M. A. (2011). The effect of resource history on the functioning of soil microbial communities is maintained across time. *Biogeosciences*, *8*, 1477–1486. <https://doi.org/10.5194/bg-8-1477-2011>
- Kembel, S. W., O'Connor, T. K., Arnold, H. K., Hubbell, S. P., Wright, S. J., & Green, J. L. (2014). Relationships between phyllosphere bacterial communities and plant functional traits in a neotropical forest. *Proceedings of the National Academy of Sciences of the United States of America*, *111*, 13715–13720. <https://doi.org/10.1073/pnas.1216057111>
- Köster, J., & Rahmann, S. (2012). Snakemake--a scalable bioinformatics workflow engine. *Bioinformatics*, *28*(19), 2520–2522. <https://doi.org/10.1093/bioinformatics/bts480>
- Kraft, N. J. B., Adler, P. B., Godoy, O., James, E. C., Fuller, S., & Levine, J. M. (2015). Community assembly, coexistence and the environmental filtering metaphor. *Functional Ecology*, *29*, 592–599. <https://doi.org/10.1111/1365-2435.12345>
- Kuznetsova, A., Brockhoff, P. B., & Christensen, R. H. B. (2017). lmerTest Package: Tests in Linear Mixed Effects Models. *Journal of Statistical Software*, *82*
- Li, Y. B., Li, Q., Yang, J. J., Lü, X. T., Liang, W. J., Han, X. G., ... Sayer, E. (2017). Home-field advantages of litter decomposition increase with increasing N deposition rates: A litter and soil perspective. *Functional Ecology*, *31*, 1792–1801. <https://doi.org/10.1111/1365-2435.12863>
- Lin, D., Pang, M., Fanin, N., Wang, H., Qian, S., Zhao, L., ... Ma, K. J. P. (2019). Fungi participate in driving home-field advantage of litter decomposition in a subtropical forest. *Plant and Soil*, *434*, 467–480.
- Lin, Y. H., He, X. B., Ma, T. W., Han, G. M., & Xiang, C. G. (2015). Priority colonization of *Cinnamomum camphora* litter by endophytes affects decomposition rate, fungal community and microbial activities under field conditions. *Pedobiologia*, *58*, 177–185. <https://doi.org/10.1016/j.pedobi.2015.09.001>
- Mallon, C. A., Elsas, J. D. V., & Salles, J. F. (2015). Microbial invasions: The process, patterns, and mechanisms. *Trends in Microbiology*, *23*, 719–729. <https://doi.org/10.1016/j.tim.2015.07.013>
- McDonald, D., Clemente, J. C., Kuczynski, J., Rideout, J. R., Stombaugh, J., Wendel, D., ... Caporaso, J. G. (2012). The Biological Observation Matrix (BIOM) format or: How I learned to stop worrying and love the ome-ome. *GigaScience*, *1*, 7. <https://doi.org/10.1186/2047-217X-1-7>
- 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. <https://doi.org/10.1111/j.1600-0706.2010.19418.x>
- Mommer, L., Cotton, T. E. A., Raaijmakers, J. M., Termorshuizen, A. J., Ruijven, J., Hendriks, M., ... Dumbrell, A. J. (2018). Lost in diversity: The interactions between soil-borne fungi, biodiversity and plant productivity. *New Phytologist*, *218*, 542–553. <https://doi.org/10.1111/nph.15036>
- Morriën, E., Hannula, S. E., Snoek, L. B., Helmsing, N. R., Zweers, H., de Hollander, M., ... van der Putten, W. H. (2017). Soil networks become more connected and take up more carbon as nature restoration progresses. *Nature Communications*, *8*, 14349. <https://doi.org/10.1038/ncomms14349>

- Nguyen, N. H., Song, Z., Bates, S. T., Branco, S., Tedersoo, L., Menke, J., ... Kennedy, P. G. (2016). FUNGuild: An open annotation tool for parsing fungal community datasets by ecological guild. *Fungal Ecology*, 20, 241–248. <https://doi.org/10.1016/j.funeco.2015.06.006>
- Oksanen, J., Blanchet, F., Kindt, R., Legendre, P., Minchin, P., & O'Hara, R. (2018). *Vegan: Community ecology package*. R package vegan, vers. 2.2-1; 2015.
- Palozzi, J. E., & Lindo, Z. (2018). Are leaf litter and microbes team players? Interpreting home-field advantage decomposition dynamics. *Soil Biology and Biochemistry*, 124, 189–198. <https://doi.org/10.1016/j.soilbio.2018.06.018>
- Persoh, D. (2013). Factors shaping community structure of endophytic fungi—evidence from the Pinus-Viscum-system. *Fungal Diversity*, 60, 55–69. <https://doi.org/10.1007/s13225-013-0225-x>
- Props, R., Kerckhof, F.-M., Rubbens, P., De Vriese, J., Sanabria, E. H., Waegeman, W., ... Boon, N. (2017). Absolute quantification of microbial taxon abundances. *The ISME Journal*, 11, 584. <https://doi.org/10.1038/ismej.2016.117>
- Quince, C., Lanzen, A., Davenport, R. J., & Turnbaugh, P. J. (2011). Removing noise from pyrosequenced amplicons. *BMC Bioinformatics*, 12, 38. <https://doi.org/10.1186/1471-2105-12-38>
- R Development Core Team. (2013). *R: A language and environment for statistical computing*. Vienna, Austria: R Foundation for Statistical Computing.
- Raaijmakers, J. M., Paulitz, T. C., Steinberg, C., Alabouvette, C., & Moënne-Loccoz, Y. (2009). The rhizosphere: A playground and battlefield for soilborne pathogens and beneficial microorganisms. *Plant and Soil*, 321, 341–361. <https://doi.org/10.1007/s11104-008-9568-6>
- Schloss, P. D., Westcott, S. L., Ryabin, T., Hall, J. R., Hartmann, M., Hollister, E. B., ... Weber, C. F. (2009). Introducing mothur: Open-source, platform-independent, community-supported software for describing and comparing microbial communities. *Applied and Environmental Microbiology*, 75, 7537–7541. <https://doi.org/10.1128/AEM.01541-09>
- van de Voorde, T. F. J., van der Putten, W. H., & Bezemer, T. M. (2011). Intra- and interspecific plant-soil interactions, soil legacies and priority effects during old-field succession. *Journal of Ecology*, 99, 945–953. <https://doi.org/10.1111/j.1365-2745.2011.01815.x>
- van der Putten, W. H. (2012). Climate change, aboveground-belowground interactions, and species' range shifts. *Annual Review of Ecology, Evolution, and Systematics*, 43, 365–383. <https://doi.org/10.1146/annurev-ecolsys-110411-160423>
- van der Wal, A., Geydan, T. D., Kuyper, T. W., & de Boer, W. (2013). A thready affair: Linking fungal diversity and community dynamics to terrestrial decomposition processes. *FEMS Microbiology Reviews*, 37, 477–494. <https://doi.org/10.1111/1574-6976.12001>
- van der Wal, A., Ottosson, E., & de Boer, W. (2015). Neglected role of fungal community composition in explaining variation in wood decay rates. *Ecology*, 96, 124–133. <https://doi.org/10.1890/14-0242.1>
- Veen, G. F., Freschet, G. T., Ordonez, A., & Wardle, D. A. (2015). Litter quality and environmental controls of home-field advantage effects on litter decomposition. *Oikos*, 124, 187–195. <https://doi.org/10.1111/oik.01374>
- Veen, G. F., Keiser, A. D., van der Putten, W. H., & Wardle, D. A. (2018). Variation in home-field advantage and ability in leaf litter decomposition across successional gradients. *Functional Ecology*, 32, 1563–1574. <https://doi.org/10.1111/1365-2435.13107>
- Veen, G. F., Snoek, B. L., Bakx-Schotman, J. M. T., Wardle, D. A., & van der Putten, W. H. (2019). Data from: Relationships between fungal community composition in decomposing leaf litter and home-field advantage effects. *Dryad Digital Repository*, <https://doi.org/10.5061/dryad.527d3f9>
- Veen, G. F., Sundqvist, M. K., & Wardle, D. A. (2015). Environmental factors and traits that drive plant litter decomposition do not determine home-field advantage effects. *Functional Ecology*, 29, 981–991. <https://doi.org/10.1111/1365-2435.12421>
- Vivanco, L., & Austin, A. T. (2008). Tree species identity alters forest litter decomposition through long-term plant and soil interactions in Patagonia, Argentina. *Journal of Ecology*, 96, 727–736. <https://doi.org/10.1111/j.1365-2745.2008.01393.x>
- Voříšková, J., & Baldrian, P. (2013). Fungal community on decomposing leaf litter undergoes rapid successional changes. *The ISME Journal*, 7, 477. <https://doi.org/10.1038/ismej.2012.116>
- Wagner, M. R., Lundberg, D. S., del Rio, T. G., Tringe, S. G., Dangl, J. L., & Mitchell-Olds, T. (2016). Host genotype and age shape the leaf and root microbiomes of a wild perennial plant. *Nature Communications*, 7, 15. <https://doi.org/10.1038/ncomms12151>
- Wang, Q., Garrity, G. M., Tiedje, J. M., & Cole, J. R. (2007). Naive Bayesian classifier for rapid assignment of rRNA sequences into the new bacterial taxonomy. *Applied and Environmental Microbiology*, 73, 5261–5267. <https://doi.org/10.1128/AEM.00062-07>
- Wardle, D. A. (2002). *Communities and ecosystems: Linking the aboveground and belowground components*. Princeton, NJ: Princeton University Press.
- Wardle, D. A., Bardgett, R. D., Klironomos, J. N., Setälä, H., van der Putten, W. H., & Wall, D. H. (2004). Ecological linkages between aboveground and belowground biota. *Science*, 304, 1629–1633. <https://doi.org/10.1126/science.1094875>
- Zhang, N., Van der Putten, W. H., & Veen, G. F. (2016). Effects of root decomposition on plant-soil feedback of early- and mid-successional plant species. *New Phytologist*, 212, 220–231. <https://doi.org/10.1111/nph.14007>
- Zhang, Z., Qu, Y., Li, S., Feng, K., Wang, S., Cai, W., ... Deng, Y. (2017). Soil bacterial quantification approaches coupling with relative abundances reflecting the changes of taxa. *Scientific Reports*, 7, 4837. <https://doi.org/10.1038/s41598-017-05260-w>
- Zobel, M. (2016). The species pool concept as a framework for studying patterns of plant diversity. *Journal of Vegetation Science*, 27, 8–18. <https://doi.org/10.1111/jvs.12333>

## SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of the article.

**How to cite this article:** Veen GF, Snoek BL, Bakx-Schotman T, Wardle DA, van der Putten WH. Relationships between fungal community composition in decomposing leaf litter and home-field advantage effects. *Funct Ecol*. 2019;33:1524–1535. <https://doi.org/10.1111/1365-2435.13351>