



Short communication

Is litter decomposition influenced by forest size and invertebrate detritivores during the dry season in semiarid Chaco Serrano?

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ABSTRACT

Forest loss can affect ecosystem processes such as litter decomposition. In semi-arid areas, where forest loss is increasing, soil fauna can play a particularly important role on litter decomposition. However, few studies have addressed the effect of soil fauna on litter decomposition in fragmented semi-arid forests, and none within the dry season in which most litter is shed. In this study, we employed litterbags filled with a common substrate to assess forest size and invertebrate detritivore effects on decomposition. Our results showed an average 14% litter mass loss along 3–9 months of incubation in the dry season, with variations being independent of forest size. Although exclusion resulted in slightly lower abundance of invertebrate detritivores, litter decomposition was similar in exclusion and non-exclusion treatments. We found no significant relationships between fragment size and invertebrate abundance or richness, which in turn did not influence decomposition. Temperature or moisture limitations, and even photo-degradation, could have masked differences in decomposition rates related to forest size during the dry season in semi-arid Chaco Serrano. Additionally, harsh environmental conditions during the incubation period could constrain the impact of invertebrate detritivores on the decomposition process. The absence of clear links between forest fragmentation, decomposition and soil fauna during the dry season, when conditions might be particularly limiting for this process, and when invertebrates could be expected to play a particularly important role, opens up new questions and highlights the complexity of this fundamental ecosystem process.

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1. Introduction

Forest loss is a main driver of the current biodiversity crisis affecting not only species composition, abundance and richness of biological assemblages (Cagnolo et al., 2009) but also altering ecosystem processes such as litter decomposition (Bennet and Saunders, 2010), which determines carbon turnover and nutrient cycling. At local scale, litter decomposition is influenced mainly by litter quality, microclimatic conditions and soil organisms (Gonzalez and Seastedt, 2001), all of which can be altered by forest size in fragmented landscapes. In smaller forest fragments, the

proportion of edge habitat increases leading to overall microclimatic conditions similar to those of edge habitats, i.e. higher light incidence, higher temperature and lower humidity, in comparison with larger fragments (Saunders et al., 1991). These particular microclimatic conditions could, in turn, affect litter quality via changes in plant community composition and both, microenvironment and litter quality alterations, could influence abundance, composition and activity of soil fauna (Vasconcelos and Laurance, 2005). Moreover, decreasing fragment size may lead to reduced plant and animal population size and increased risk of local extinctions (Ewers and Didham, 2006).

In semi-arid areas, moisture limitations for microorganisms may enhance the role of soil macro and mesofauna on litter decomposition (Araujo et al., 2012). However, most studies about forest fragmentation and soil fauna effects on leaf litter decomposition have been conducted in tropical forests (e.g. Vasconcelos and Laurance, 2005; Didham, 1998), with fewer studies from arid and

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semi-arid regions (Gonzalez and Seastedt, 2001; Xin et al., 2012). To our knowledge this is the first study from a seasonally dry forest focusing on the relationship between decomposition and forest area mediated by soil fauna within the dry season when most litter reaches the soil and when climatic conditions differ most from those of tropical and temperate forests.

Chaco Serrano is a seasonally dry subtropical forest from Central Argentina which lost over 90% of its original cover in recent years (Zak et al., 2004). In this ecosystem, fragmentation has resulted in biodiversity impoverishment, including plant and insect communities (Cagnolo et al., 2009), as well as alterations in ecological processes such as herbivory, parasitism (Valladares et al., 2006) and litter decomposition in the wet season (Moreno et al., 2014). In this study, we assessed the effects of fragment size and invertebrate detritivores on litter decomposition during the dry season in semi-arid Chaco Serrano. We expected that: (1) litter decomposition would be greater in larger than in smaller fragments; (2) abundance and richness of invertebrate detritivores would decrease with fragment size, showing changes in taxonomic composition due to differential vulnerability; (3) if invertebrate detritivores have a relevant impact on litter decomposition, their exclusion would significantly decrease litter decomposition rates, and such rates would be related to detritivore abundance, richness and/or taxonomic composition.

2. Materials and methods

The study was conducted in a fragmented area of Chaco Serrano in central Argentina (31°10' to 31°30'S and 64° 00' to 64°30'W). The region has a temperate climate with markedly scarce precipitations in winter. During the 9-month duration of the study, monthly mean temperatures ranged from 9.4 °C to 22.9 °C, with an accumulated rainfall of 362 mm and with water stress conditions along most of the incubation period (Fig. 1).

Native vegetation is currently restricted to isolated forest fragments within an intensely managed agricultural and urban matrix (Zak et al., 2004). We selected 12 of those fragments ranging from 0.57 ha to more than 1000 ha, keeping isolation between them (342.30 ± 90.56 m average distance to the nearest larger fragment) and matrix characteristics (soybean crop) as uniform as possible.

To analyze the effect of forest size on leaf litter decomposition we incubated, in all the selected sites, a common plant substrate (leaves of *Platanus acerifolia* (Ait.) Willd.) in order to control for litter quality effects (Berg and Laskowski, 2006). We collected leaf

litter in autumn, keeping it air-dried until processing. We prepared 216 litterbags (15 × 20 cm) with 0.3 mm mesh at the bottom (to avoid losing material from inside the bags) and 1 mm mesh on top, to prevent colonization by soil macroinvertebrates (e.g. ants, beetles, Diplopoda) and mesofauna (larger-bodied Acari and Collembola) while allowing microfauna access. On half of the litterbags, we made five 1-cm² perforations to facilitate soil fauna access (Vasconcelos and Laurance, 2005), in order to evaluate the effect of invertebrates on decomposition by comparing perforated and non-perforated litterbag decomposition patterns.

In February 2009, we placed nine perforated (non-exclusion treatment) and nine non-perforated (exclusion treatment) litterbags, each filled with 2 g of the common substrate, within each of the twelve forest fragments (in a 3 × 2 array with 1 m spacing, at approximately 20 m from the forest edge), covering them with local litter to simulate the natural decomposition conditions. After three, five and nine months of incubation, we randomly retrieved three perforated and three non-perforated litterbags of each fragment to determine litter decomposition in the period. Litterbags retrieved after three and five months of incubation were stored at 20 °C until processing (Vasconcelos and Laurance, 2005). Litterbags removed after nine months of incubation were first transported to the laboratory and their contents were placed in Berlese funnels (44 cm height, 20 cm diameter) under 60-W light bulbs during 7 days, for invertebrate extraction. The invertebrates were preserved in 70% ethanol solution. Finally, we cleaned and oven-dried the litter at 50 °C, during three days, to measure dry mass loss due to decomposition.

To estimate true dry mass we first calculated air-dried water content. We air-dried a subsample of the common substrate and then oven-dried it at 50 °C until constant weight was achieved (about 48 h). Water content (%) was calculated from the mass loss in samples after drying. As litterbags can be contaminated with soil, we also corrected litter mass loss as the effective loss of organic matter (litter mass loss without ashes) by burning the samples in an oven at 500 °C for 4 h (Perez Harguindeguy et al., 2013). We calculated the decomposition constant (*k*-value) as follows: $k = -\ln(M_t/M_0)/t$, where *k* = decomposition rate constant (year⁻¹), *M*₀ = mass of litter at time 0, *M*_{*t*} = mass of litter at time *t*, and *t* = duration of incubation (years) (Perez Harguindeguy et al., 2013).

All specimens extracted from litterbags were counted and identified (to Family level in the case of insects and to Order level for other invertebrates) and assigned to detritivore or other functional groups depending on the dominant feeding habit of each taxonomic group. We performed all statistical analyses using detritivores only.

We expressed litter decomposition rates as the percentage of dry mass loss per litterbag, whereas abundance and richness of invertebrate detritivores were indicated by the number of individuals or taxa per litterbag. To evaluate the effects of fragment size on (a) litter decomposition rates and (b) detritivore abundance and richness, we used (a) a linear mixed-effects model (LMM) and (b) generalized linear mixed models (GLMM) with litter decomposition rate, abundance or richness as response variables, fragment area (log+1-transformed to linearize the relationship) and exclusion treatment as fixed effects, and site (forest fragment) as random effect. In (a) we also incorporated incubation time as fixed effect and, nested within site, as random effect in order to contemplate dependence of repeated measurements over time. Litter decomposition and abundance-richness were fitted with a Gaussian and a Poisson error structure, respectively. We performed a principal component analysis (PCA) to explore changes in taxonomic composition (quantitative data) of invertebrate assemblages, using the software Infostat version 2008. We employed Spearman correlations between assemblage position on each of the two first

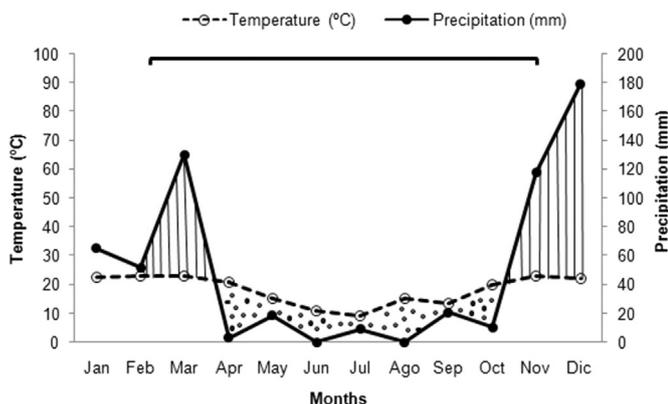


Fig. 1. Climograph of the study site, mean values January to December 2009. JAN = January, FEB = February, MAR = March, APR = April, MAY = May, JUN = June, JUL = July, AUG = August, SEP = September, OCT = October, NOV = November, DEC = December. Bracket on the top indicates the study period. The dotted areas indicate periods of water stress and the striped areas indicate periods of excess water.

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