

Direct and indirect effects of climate on decomposition in native ecosystems from central Argentina

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Abstract Climate affects litter decomposition directly through temperature and moisture, determining the ecosystem potential decomposition, and indirectly through its effect on plant community composition and litter quality, determining litter potential decomposition. It would be expected that both the direct and indirect effects of climate on decomposition act in the same direction along gradients of actual evapotranspiration (AET). However, studies from semiarid ecosystems challenge this idea, suggesting that the climatic conditions that favour decomposition activity, and the consequent ecosystem potential decomposition, do not necessarily lead to litter being easier to decompose. We explored the decomposition patterns of four arid to subhumid native ecosystems with different AET in central-western Argentina and we analysed if ecosystem potential decomposition (climatic direct effect), nutrient availability and leaf litter potential decomposition (climatic indirect effect) all increased with AET. In general, the direct effect of climate (AET) on decomposition (i.e. ecosystem potential decomposition), showed a similar pattern to nutrient availability in soils (higher for xerophytic and mountain woodlands and lower for the other ecosystems), but different from the pattern of leaf litter potential decomposition. However, the range of variation in the ecosystem potential decomposition was much higher than the range of variation in litter potential decomposition, indicating that the direct effect of climate on decomposition was far stronger than the indirect effect through litter quality. Our results provide additional experimental evidence supporting the direct control of climate over decomposition, and therefore nutrient cycling. For the ecosystems considered, those with the highest AET are the ecosystems with the highest potential decomposition. But what is more interesting is that our results suggest that the indirect control of climate over decomposition through vegetation characteristics and decomposability does not follow the same trend as the direct effect of climate. This finding has important implications in the prediction of the effects of climate change on semiarid ecosystems.

Key words: actual evapotranspiration, arid ecosystems, gradients, ionic exchange resins, litter quality, nutrients.

INTRODUCTION

Decomposition of litter is a fundamental control of nutrient and carbon cycling in terrestrial ecosystems (Swift *et al.* 1979; Schlesinger 2000). This process is influenced by climate, the decomposer community, and litter quality (Meentemeyer 1978; Swift *et al.* 1979; Cadisch & Giller 1997). Climate affects litter decomposition directly through the effects of temperature and moisture (Swift *et al.* 1979; Aerts 1997; Liski *et al.* 2003), thus determining the ecosystem potential decomposition. Climate can also affect decomposition indirectly, through its effect on plant community composition and thus litter quality. Litter quality, in turn, determines litter potential decomposition in a given

ecosystem (Swift & Anderson 1989; Aerts 1997; Dorrepaal *et al.* 2005). According to Aerts (1997), it would be expected that both direct and indirect effects of climate on decomposition act in the same direction along gradients of actual evapotranspiration (AET). In this way, a warm and moist climate (i.e. with high AET) would exert a double positive effect on decomposition, both by promoting the activity of decomposers (direct effect) and by allowing more decomposable species to dominate the plant community (indirect effect), as happens in tropical forests compared with drier and/or colder ecosystems (Aerts 1997; Aerts & Chapin 2000). These more decomposable species would enhance nutrient availability, thus exerting a positive feedback on decomposition. Accordingly, in regions with lower AET, such as arid and semiarid ecosystems, with seasonally or permanently low water availability, it is expected that climate limits both the

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activity of decomposers and the quality of the dominant plant species. In accordance with these ideas, some studies have shown a decrease of plant quality along gradients of increasing water shortage (Cunningham *et al.* 1999; Fonseca *et al.* 2000; Adler *et al.* 2004).

However, other studies have shown that plant quality, litter quality and/or potential decomposition of litter can remain constant (Oesterheld *et al.* 1999) or even decline with increasing precipitation or AET (Wedin 1995; Murphy *et al.* 1998; Murphy *et al.* 2002), particularly when semiarid ecosystems are considered. These studies challenge the idea that the direct (temperature and precipitation over decomposers activity) and the indirect (litter quality over decomposability) climatic control on decomposition always act in the same direction along gradients of AET. This suggests that the climatic conditions that favour decomposer activity, and in consequence ecosystem potential decomposition, do not necessarily lead to litter being easier to decompose. Consequently, the overall climatic effects on nutrient availability would be difficult to predict.

In this context, our aims were (i) to determine if ecosystem potential decomposition (climatic direct effect) and litter potential decomposition (climatic indirect effect) both increased with AET across four arid to subhumid ecosystems in central-western Argentina; and (ii) to determine the patterns of nutrient availability across ecosystems and their relation to AET, potential ecosystem decomposition and potential litter decomposition.

METHODS

Study area

We studied four native vegetation ecosystems exposed to different climatic conditions in central-western Argentina (Díaz *et al.* 1998). These ecosystems include, in order of increasing AET (Turc formula), mountain woodlands, xerophytic woodlands, mountain grasslands, and xerophytic shrublands (Fig. 1, Table 1). These vegetation units belong to the Chaco (the first three) and the Monte (the last one) Phytogeographical Provinces (Cabrera 1976) and have been described by Luti *et al.* (1979), Cabido *et al.* (1993), and Zak and Cabido (2002). Within each ecosystem we looked for areas that preserved the characteristics of the native vegetation and we chose one. Within each of these areas (approximately 300 × 300 m) we selected five sites as replicates (4 × 4 m). Each replicate was a vegetation patch representative of the vegetation, topography and soil characteristic of the selected area within the ecosystem, and was as little

disturbed and as internally homogenous as possible. Although we have not true replicates of the ecosystems (because only one area was selected per ecosystem) we consider that the areas were representative enough to establish realistic conclusions.

The study area that was representative of mountain woodlands located in the lower slopes of the Córdoba Mountains was dominated by trees (*Acacia praecox* and *Lithraea Molleoides*). Appendix I gives further detail on species names, families and functional types. The area representative of the xerophytic woodlands was dominated by shrubs (*Larrea divaricata*), trees (*Prosopis flexuosa*) and herbaceous species (*Justicia squarrosa*), with a smaller proportion of succulents and aphyllous woody species. The study area in the mountain grasslands was dominated by grasses (*Deyeuxia hieronymi*) and herbaceous dicots (*Alchemilla pinnata*) in similar proportions, and a small proportion of shrubs. The area representative of the western xerophytic shrublands was dominated by shrubs (*Larrea cuneifolia*) followed by succulents (*Opuntia sulphurea* and *Tephrocactus articulatus*). Soils of the four areas are sandy-loam (mollisols in woodlands, xerophytic woodland and mountain grasslands, and aridisols in xerophytic shrublands). The main parameters of the upper horizon (measured from a 10-cm deep compound sample of six cores at each replicate) are summarized in Table 1. It should be noted that the two systems with lower AET are limited by different factors. In the case of mountain grasslands AET is limited by low temperatures, while in xerophytic shrublands AET is limited by low rainfalls.

Ecosystem potential decomposition

In order to assess ecosystem potential decomposition (i.e. that attributed to the macro- and micro-environmental conditions together, and independent of litter quality) we incubated two different standard materials in each ecosystem type. The standard materials used were filter paper (cellulose) and soft wood sticks. Although these materials are somewhat artificial, they can provide an index of micro- and macro-environmental controls on decomposition (Binkley 1984; Harmon *et al.* 1999) avoiding differences in decomposition due to initial differences in litter quality. Even when the standard substrates may not be present in an ecosystem in the exact form in which we incubate them, their chemical constituents are present and so are the microorganisms that can degrade them (Orwin *et al.* 2006).

At each replicate site we incubated four samples of each of the two different standard materials (filter paper and soft wood sticks). Two samples of each material were incubated over 11 weeks, and the other two over 25 weeks. Decomposition was estimated

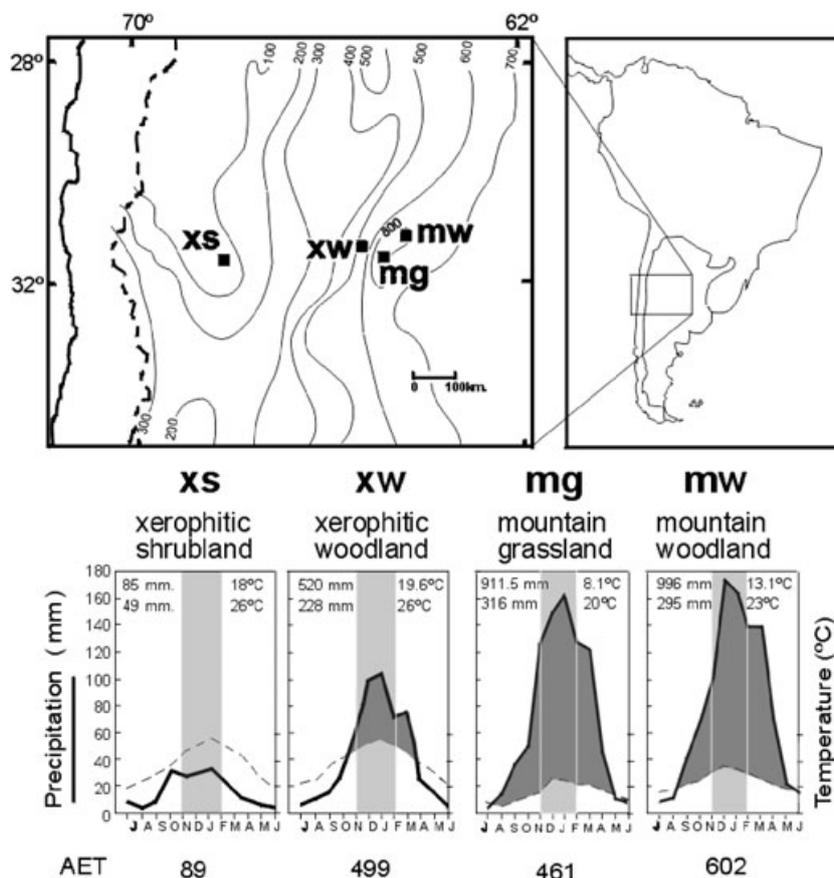


Fig. 1. Location of the study areas (indicated with the initials of each ecosystem) and climodiagrams of each area, taken from the closest meteorological station and based on 50-year averages (Servicio Meteorológico Argentino, Ferrocarriles Argentinos, Palacios & Zamar 1986). In the central-western Argentina map, lines represent isohyets. Within each climodiagram (representing mean monthly temperatures and precipitation), total annual precipitation and mean annual temperature are indicated in the first line and precipitation and mean temperature during the summer months (when incubation was carried out, within the grey box in the climodiagram) are indicated in the second line. Below each climodiagram the actual evapotranspiration (AET) calculated according to the Turc formula is indicated.

through the litter-bag technique (Bocock & Gilber 1957). Each sample consisted of a 0.3 mm mesh bag with 1 g of air-dried standard material. For each material, true initial dry mass was calculated from the water content of a subsample that was weighed at the same time as the samples, oven-dried until it was a constant weight at 50°C and then reweighed. All samples were buried (at about 3 cm depth) at each sampling site at the beginning of the rainy season (November 1997). On the basis of previous work, we carried out the first retrieval after 11 weeks of incubation (Pérez-Harguindeguy *et al.* 2000; Vaieretti *et al.* 2005). The second retrieval date was at the end of the rainy season (May 1998), after 25 weeks of incubation. Measurements of the two samples of each material and over each incubation period were average for each site before carrying out statistical analyses. Therefore, there were five mass loss values per area (representative of each ecosystem) per period per standard material. Unfortunately, during the 25-week

period the differences in decomposition rates among the two fastest decomposing ecosystems were lost due to mass losses approaching 100% in filter paper samples. For this reason, we reported only the 11-week mass loss values.

Litter potential decomposition

Potential leaf litter decomposition was obtained from previous work where all species from all sites were incubated together under standard conditions (decomposition beds, Pérez-Harguindeguy *et al.* 2000; Giorgis 2004; Vaieretti *et al.* 2005; Appendix I). Because of this the measured value of potential decomposition can be different from real. However, these experiments were not intended to simulate *in situ* decomposition in the habitats of origin of each species; rather they should be seen as a test of the role of different leaf litter quality (coming from different

Table 1. Ecosystem characterization in terms of vegetation type, location, climate and soil characteristics

| Vegetation types | Mountain woodlands | Xerophytic woodlands | Mountain grasslands | Xerophytic shrublands |
|------------------------------------|--------------------------|--------------------------|--------------------------|--------------------------|
| Location | 31°14'25"S 64°19'26"O | 30°47'04"S 64°53'42"O | 31°36'49"S 64°43'06"O | 31°41'49"S 68°09'18"O |
| Altitude | 700 m a.s.l. | 620 m a.s.l. | 1900 m a.s.l. | 600 m a.s.l. |
| Climate | | | | |
| Temperature (°C) | | | | |
| Annual | 13 | 20 | 8 | 18 |
| Summer | 23 | 26 | 20 | 26 |
| Precipitation (mm) | | | | |
| Annual | 996 | 520 | 912 | 85 |
| Summer | 295 | 228 | 316 | 49 |
| AET (annual) | 602 | 499 | 461 | 89 |
| Soil characterization | Mean ± SE | Mean ± SE | Mean ± SE | Mean ± SE |
| SOM (%) | 20.8 ± 1.7 | 5.4 ± 0.7 | 6.4 ± 0.7 | 0.6 ± 0.1 |
| pH | 6.0 ± 0.1 | 6.9 ± 0.2 | 6.71 ± 0.2 | 7.8 ± 0.2 |
| Texture (% sand/loam/clay) | 68/23/9 | 80/15/5 | 56/25/19 | 66/24/10 |
| CEC (Me/100 g) | 34.1 ± 2.1 | 17.8 ± 0.2 | 31.0 ± 3.3 | 12.0 ± 0.7 |
| Bulk density (gr/cm ³) | 0.5 ± 0.1 | 1.0 ± 0.03 | 1.1 ± 0.1 | 1.3 ± 0.03 |

AET, actual evapotranspiration; CEC, cation exchange capacity; SOM, soil organic matter.

ecosystems) under a standard environment (Cornelissen *et al.* 1999; Pérez-Harguindeguy *et al.* 2000).

Accordingly, potential leaf litter decomposition for each site (five per ecosystem) was calculated from the potential leaf litter decomposition of the different species present (Appendix I). To obtain a unique value representative of the site, we calculated a community-aggregated value based on the biomass-ratio hypothesis by averaging the values for all species at each site, weighting each by its abundance at the site (Grime 1998; Díaz *et al.* 1999; Garnier *et al.* 2004). The abundance of each species was estimated through a floristic survey, where cover was estimated in categories of 5%.

For all decomposition beds leaf litter was collected from at least eight individuals for each species from the site where each species was more abundant. After processing, all species from the different sites were incubated together under standard conditions (see Cornelissen 1996 and Cornelissen *et al.* 1999 for details of collection, processing and incubation). Although the absolute values of percentage mass loss may change according to the external conditions, the ranking of species is consistent, as was demonstrated by strong correlation coefficients of rank order when the same sets of species were incubated under contrasting conditions ($r = 0.91$ for a set of 14 British species, and $r = 0.96$ for a set of 16 Argentine species, Cornelissen *et al.* 1999).

Nutrient availability in soil

To measure the accumulation of the main available soil nutrients for plants (NH_4^+ , NO_3^- , and PO_4^{3-}), we placed

three 5 g ion exchange resin bags (Amberlite IRN-150), packed in 0.3 mm double nylon mesh within each sampling site (Sibbeson 1977; Binkley & Hart 1989; Stark 2000). Each set of three resin bags was buried under 5 cm of the soil at each replicate site in each ecosystems, and was incubated during the same period as the litter bags (25 weeks). During this period ionic resins absorbed available nutrients in the same way as a plant root, and thus resins may realistically reflect nutrient availability for plants. After the incubation, resin bags were collected and washed free of soil in de-ionized water. Nutrients (NH_4^+ , NO_3^- , and PO_4^{3-}) were extracted in ClNa 2 N and measured colourimetrically with an Autoanalyser (Lajtha *et al.* 1999; Robertson *et al.* 1999).

Statistical analysis of data

Since data were normally distributed (Sokal & Rohlf 1995) and the dispersion plots of residuals indicated that errors were independent and variance was homogeneous (Infostat, Di Rienzo *et al.* 2002) we used ANOVA to test for differences in the ecosystem potential decomposition (estimated as percentage dry weight loss) and litter potential decomposition, among the four areas representative of the ecosystems, and, in the case of ecosystem potential decomposition, also between standard materials. For ecosystem potential decomposition, the dry weight loss at the two different dates were analysed separately. We then carried out posthoc comparisons with Fisher least significant difference tests (Infostat, Di Rienzo *et al.* 2002). To analyse associations between ecosystem potential

decomposition, litter potential decomposition and nutrients and with AET, we carried out Pearson correlations among variables, across the set of four ecosystems. For these analyses, we used an average value per variable for each ecosystem. Since we had so few points, this analysis had little statistical power, but we used it as an indicator of the degree of congruence between patterns of variability of the different variables across ecosystems.

RESULTS

Ecosystems potential decomposition

Values of dry weight loss (%) of standard materials in the different ecosystems were highly correlated between 11 and 25 weeks of burial ($r = 0.93$, $P < 0.001$ for filter paper, and $r = 0.91$, $P < 0.001$ for wood sticks: Pearson Correlation Coefficient). During the 11-week period half of the filter paper samples lost more than 95% of their mass (Fig. 2).

We found significant main effects and interactions between ecosystem type and material on ecosystem potential decomposition ($P < 0.0001$, Table 2, Fig. 2). In general, and as expected, filter paper decomposed significantly faster than wood sticks in all ecosystems except in the xerophytic shrubland, where decomposition of both materials was not significantly different (Fig. 2). Out of the four ecosystems compared, the xerophytic woodland and the mountain woodlands showed the highest decomposition rates for both materials, and the xerophytic shrublands showed the lowest. Mountain grasslands showed intermediate ecosystem decomposition, but in the case of filter paper

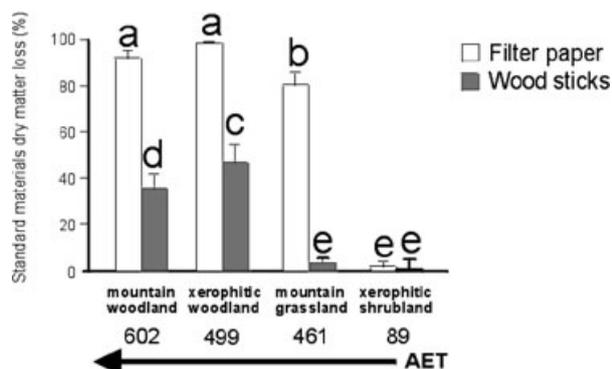


Fig. 2. Ecosystem potential decomposition: mass loss of filter paper samples (white bars), and wood stick samples (black bars), during 11 weeks under the summer climatic conditions of the main native ecosystems of central Argentina. Different letters indicate significant differences (Fisher least significant difference test, $P < 0.05$). Below the bars of each ecosystem the actual evapotranspiration (AET) calculated according to the Turc formula is indicated.

decomposition was relatively fast, while in the case of wood sticks decomposition did not differ from that observed in the xerophytic shrubland ($P > 0.05$, Fig. 2). Decomposition of filter paper was significantly correlated with AET values ($r = 0.97$, $P = 0.03$, $n = 4$), but not with other climatic variables or soil characteristics. Overall, decomposition was maximum at systems with higher AET (xerophytic woodlands and mountain woodlands), lower under intermediate AET (mountain woodlands, the coldest ecosystem), and minimum at sites with low AET (xerophytic shrublands, the driest ecosystem).

Soil nutrient availability

In general, soil nutrient availability was higher in the systems with highest AET (woodland ecosystems), and decreased in systems with lowest AET (xerophytic shrublands and mountain grasslands, Fig. 3). Nitrogen availability, both for NO_3^- and NH_4^+ , was markedly higher in the mountain woodlands (Fig. 3a,b), whereas PO_4^- availability was similar in both woodlands (Fig. 3c). Although they generally followed the same trend (filter paper: $r = 0.46$ for NH_4^+ , $r = 0.53$ for NO_3^- , $r = 0.40$ for PO_4^- ; wood sticks: $r = 0.60$ for NH_4^+ , $r = 0.70$ for NO_3^- , $r = 0.90$ for PO_4^-) or with AET ($r = 0.64$ for NH_4^+ , $r = 0.68$ for NO_3^- , $r = 0.42$ for PO_4^-), no nutrient variable was significantly correlated with patterns of decomposition in standard materials ($P > 0.05$).

Litter potential decomposition

Litter potential decomposition rates (community-aggregated values) estimated for the xerophytic shrubland and xerophytic woodland species assemblages were significantly faster than those estimated for the

Table 2. ANOVA model for mass loss of standard materials (filter paper and wood sticks) incubated for 11 weeks during the rainy season under the environmental conditions of four native ecosystems from central to western Argentina, and a *posteriori* test of the least significant differences for (a) ecosystem and (b) material

| Source of variation | d.f. | F | Significance of F |
|----------------------|------|-----------|-------------------|
| Main effects | 4 | 178.72 | *** |
| Ecosystem | 3 | 114.11 | *** |
| Substrate | 1 | 359.74 | *** |
| Two-way interactions | | | |
| Ecosystem* substrate | 3 | 4609.53 | *** |
| Explained | 7 | 20 684.55 | *** |

Asterisk indicate significance of factors: *** $P < 0.0005$.

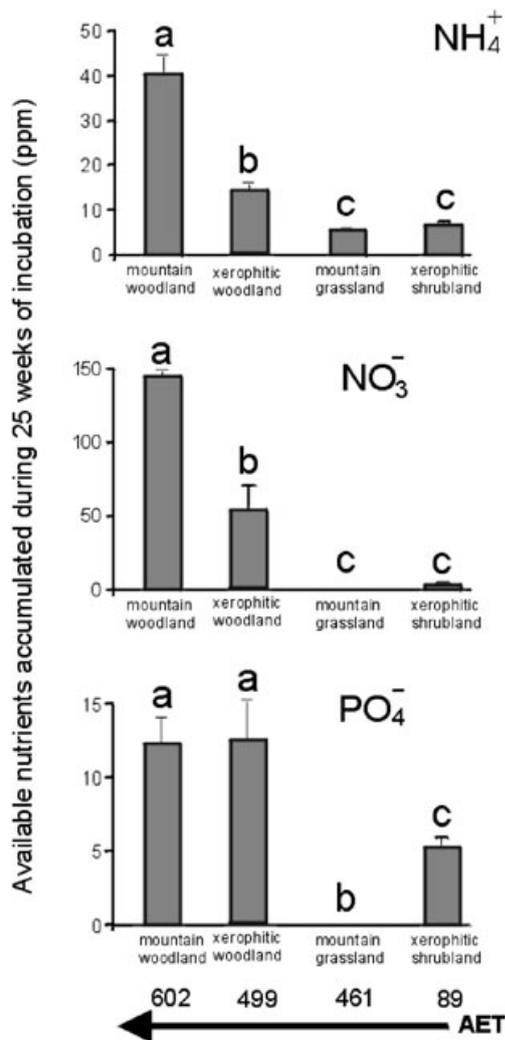


Fig. 3. Available nutrients absorbed by ion exchange resins incubated during 25 weeks (the maximum incubation period) at the same sites where incubation of standard materials was made at each system. Different letters indicate significant differences (Fisher least significant difference test, $P = 0.05$). Below the bar of each ecosystem the actual evapotranspiration (AET) calculated according to the Turc formula is indicated.

mountain grasslands and the mountain woodlands assemblages (Fig. 4). Litter potential decomposition was not significantly correlated with AET ($r = -0.48$), potential ecosystem decomposition (filter paper: $r = -0.36$; wood sticks: $r = 0.29$) or nutrient availability in the four ecosystems ($r = -0.26$ for NH_4^+ , $r = -0.18$ for NO_3^- , $r = 0.45$ for PO_4^-), nor were there any consistent trends (Figs 2–4).

DISCUSSION

In general, the direct effect of climate (AET) on decomposition, evaluated through the incubation of

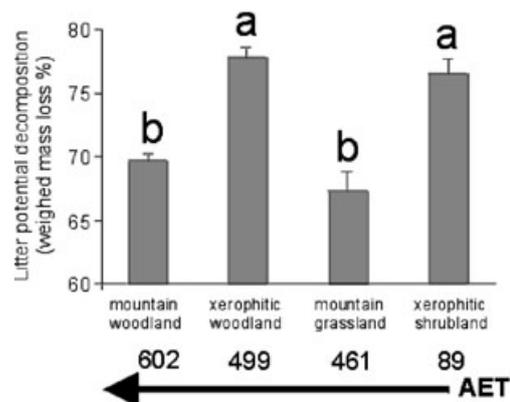


Fig. 4. Litter potential decomposition (weighted mean potential decomposition expressed as percentage mass loss) of the assemblage of species at each site. Different letters indicate significant differences (Fisher least significant difference test, $P = 0.05$). Below the bar of each ecosystem the actual evapotranspiration (AET) calculated according to the Turc formula is indicated.

standard materials (i.e. ecosystem potential decomposition), showed a similar pattern to nutrient availability in soils (higher for xerophytic and mountain woodlands and lower for the other ecosystems), but quite different from the pattern of litter potential decomposition. However, the range of variation in the ecosystem potential decomposition was much higher than the range of variation in litter potential decomposition, indicating that the direct effect of climate on decomposition was far stronger than the indirect effect through litter quality.

In relation to the direct effect of climate on decomposition, we found that ecosystem potential decomposition decreased with AET across the ecosystems studied as predicted by the general models (Meentemeyer 1978; Aerts 1997). Decomposition of standard substrates was highest in ecosystems with intermediate values of precipitation and temperature, since AET is limited by temperature in cold and moist ecosystems, and limited by precipitation in warm and dry ecosystems. These results are in broad agreement with those from Gallardo and Merino (1993) who found that decomposition shifted from being temperature-limited in areas close to the Atlantic Ocean (moist and cold) to being mainly moisture-limited in the southern part of the Mediterranean (drier and warmer). The pattern of soil nutrient availability, following a similar trend as the pattern of ecosystem decomposition (i.e. decomposition of standard materials), suggests the prevalence of a direct climatic control on nutrient availability mediated by decomposition in the ecosystems considered in this study.

In relation to the indirect control of climate, we found that litter potential decomposability did not follow the same pattern as ecosystems potential

decomposition. The pattern in this case was coincident with temperature and precipitation gradients but not with AET, as species coming from warmer and more xeric ecosystems tended to decompose faster, while species coming from colder and moister environments tended to decompose slower. In other words, climate direct control of decomposition was not associated with its indirect effect through vegetation quality and hence decomposability.

Our results are consistent with studies reporting a decrease in litter quality with increasing precipitation for grassland ecosystems (Wedin 1995; Murphy *et al.* 2002), but they are in partial contradiction with the findings of Aerts (1997), Aerts and Chapin (2000) and Chapin *et al.* (2002) for temperate and tropical ecosystems. In our study, the different patterns of potential ecosystem and litter decomposition were particularly evident from the results obtained for the xerophytic shrublands and mountain woodlands, where the climatically most unfavourable system in terms of AET and available nutrients (xerophytic shrubland), showed high potential litter decomposability, while the climatically most favourable environment, with high AET and high nutrient content (mountain woodlands), showed low potential litter decomposability. Our results do not allow us to provide an alternative mechanism to that proposed by Aerts (1997) to explain the patterns observed. However, Killingbeck and Whitford (1996) pointed out that the usually high quality of desert shrub leaves, and consequently its decomposability, could be related to the high spatial and temporal heterogeneity common to arid ecosystems (Cadisch & Giller 1997; Cornelissen *et al.* 1999). Consequently, although average nutrient availability is low in these systems, this availability may be comparatively high under shrubs, which in turn may be not nutrient limited. Additionally, Murphy *et al.* (2002), who found higher litter quality in arid grasslands with less than 300 mm rainfall than in highly productive tall-grasslands, argued that increasing water availability allows plants to use nutrients more efficiently. This should lead to more carbon fixed per unit of nitrogen, producing leaves richer in structural material such as lignin, which should in turn confer advantages in canopy dominance.

In conclusion, our results provide additional experimental evidence supporting the direct control of climate over decomposition, and therefore nutrient cycling. For the ecosystems considered, those ecosystems with the highest AET have the highest potential decomposition. But, more interesting; our results suggest that the indirect control of climate over decomposition through vegetation characteristics and decomposability does not follow the same trend as the direct effect of climate. When thinking of the possible effects of these contrary controls it should be noted that indirect control through vegetation characteristics appears to be considerably less important than direct

control, as shown by the magnitude of the differences between ecosystems in potential decomposition and litter potential decomposition. Thus, under the climate change scenarios for the next decades (McCarthy *et al.* 2001), changes in litter potential decomposition can either enhance or partially revert changes in ecosystem potential decomposition depending on the ecosystems considered.

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APPENDIX I

Species (name from Zuloaga *et al.* 1994; Zuloaga & Morrone 1996a,b), family, functional type (modified from Diaz & Cabido 1997), ecosystem where the species were present, and percentage potential litter mass loss (% LML).

| Species | Family | Functional types | Ecosystem | % LML |
|--|----------------|------------------|-----------|-------|
| <i>Acacia aroma</i> Gilles ex Hook. & Arn. | Fabaceae | Tree | mw | 70.3 |
| <i>Acacia caven</i> (Molina) Molina | Fabaceae | Tree | mw | 56.4 |
| <i>Acacia furcatispina</i> Burkart | Fabaceae | Tree | xw | 89.1 |
| <i>Acacia praecox</i> Griseb. | Fabaceae | Tree | mw | 71.9 |
| <i>Aspidosperma quebracho-blanco</i> Schlttdl. | Apocynaceae | Tree | xw | 61.9 |
| <i>Carex fuscata</i> d'Urv. | Cyperaceae | Graminoid | mg | 29.7 |
| <i>Celtis pallida</i> Torr. | Celtidaceae | Shrub | mw and xw | 83.1 |
| <i>Celtis tala</i> Gillies ex Planch. | Celtidaceae | Tree | mw | 81.7 |
| <i>Condalia montana</i> A. Cast. | Rhamnaceae | Shrub | mw | 60.7 |
| <i>Conyza bonaerensis</i> (L.) Cronquist | Asteraceae | Herbaceous dicot | mw | 61.6 |
| <i>Cordobia argentea</i> (Griseb.) Nied. | Malpighiaceae | Vine | xw | 99.9 |
| <i>Croton sarcopetalus</i> Müll. Arg. | Euphorbiaceae | Herbaceous dicot | mw | 93.7 |
| <i>Bromelia urbaniana</i> (Mez) L.B. Sm. | Bromeliaceae | Bromeliad | xw | 56.5 |
| <i>Deyeuxia hieronymi</i> (Hack.) Türpe | Poaceae | Graminoid | mg | 52.4 |
| <i>Eryngium agavifolium</i> Griseb. | Apiaceae | Bromeliad | mg | 77.5 |
| <i>Eupatorium viscidum</i> Hook. & Arn. | Asteraceae | Shrub | mw | 92.8 |
| <i>Geranium magellanicum</i> Hook. f. var. <i>magellanicum</i> | Geraniaceae | Herbaceous dicot | mg | 61.7 |
| <i>Juncus uruguensis</i> Griseb. | Juncaceae | Graminoid | mg | 65.2 |
| <i>Justicia squarrosa</i> Griseb. | Acanthaceae | Herbaceous dicot | xw | 93.7 |
| <i>Alchemilla pinnata</i> Ruiz & Pav | Rosaceae | Herbaceous dicot | mg | 90.8 |
| <i>Larrea cuneifolia</i> Cav. | Zygophyllaceae | Shrub | xs | 89.8 |
| <i>Larrea divaricata</i> Cav. | Zygophyllaceae | Shrub | xw | 89.8 |
| <i>Lithraea molleoides</i> (Vell.) Engl. | Anacardiaceae | Tree | mw | 57.5 |
| <i>Nothoscordum gracile</i> (Dryand. ex Aiton) Stearn | Liliaceae | Graminoid | mg | 97.3 |
| <i>Opuntia sulphurea</i> Gillies ex Salm-Dyck | Cactaceae | Succulent | xw and xs | 59.6 |
| <i>Pithecoctenium cynanchoides</i> DC | Bignoniaceae | Vine | xw and mw | 97.5 |
| <i>Poa stuckertii</i> (Hack) Parodi | Poaceae | Graminoid | mg | 63.5 |
| <i>Prosopis flexuosa</i> DC. | Fabaceae | Tree | xw | 72.9 |
| <i>Senna aphylla</i> (Cav.) H.S. Irwin & Barneby | Fabaceae | Aphyllous shrub | xs | 40.6 |
| <i>Setaria pampeana</i> Parodi ex Nicora | Poaceae | Graminoid | xw | 78.0 |
| <i>Stipa eriostachya</i> H.B.K. | Poaceae | Graminoid | mw | 38.4 |
| <i>Tephrocactus articulatus</i> (Pfeiff.) Backeb. var. <i>articulatus</i> | Poaceae | Succulent | xs | 58.9 |
| <i>Trichloris crinita</i> (Lag.) Parodi | Poaceae | Graminoid | xw | 58.6 |
| <i>Zizyphus mistol</i> Griseb. | Rhamnaceae | Tree | xw | 53.5 |

mg, mountain grassland; mw, mountain woodland; xs, xerophytic shrubland; xw, xerophytic woodland.