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Water provisioning services in a seasonally dry subtropical mountain: Identifying priority landscapes for conservation

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SUMMARY

The influence of landscape characteristics on dry season baseflow in mountain areas with a long dry season depends on a complex array of factors which need to be identified in order to prioritize landscapes for conservation of water provisioning services. Our objective was to detect which landscapes, as combinations of land cover types and topographical features are better suited to provide water during the dry season. We evaluated dry season water discharge (mm day^{-1}) and rainfall during three years in 16 small headwater catchments (1.1–3.5 km^2) in the mountains of central Argentina. For each catchment we estimated landscape variables as the proportion of five land-cover units and eight topographic properties. We analyzed water discharge as a function of landscape variables using regressions. Both rainfall and water discharge declined from years 1 to 3, but differences in water discharge among catchments were larger than differences among years, and consistent throughout time. Dry season water discharge was always higher in catchments located in rugged landscapes, with a high proportion of deep valleys and rock outcrops as compared to catchments in gentle landscapes with a high proportion of plains and covered with grasslands. We conclude that conservation priorities toward rugged landscapes would optimize water provisioning services. Reducing present rates of soil loss in deep valleys and controlling their incipient invasion by woody aliens is especially important. In coincidence, rugged landscapes host a higher diversity of various taxa.

1. Introduction

The provision of water is a growing worldwide challenge due to increased demand and progressive ecosystem degradation (Mark and Dickinson, 2008). In particular, mountain ecosystems are of significant hydrological importance because they have higher precipitation and lower evapotranspiration compared with surrounding lowlands (Messerli et al., 2004; Viviroli et al., 2007). Mountains store part of the water to be released later, allowing the maintenance of perennial streams, even in regions with seasonal precipitations. Thus, the adequate management of mountain landscapes to optimize water provision is a societal need in seasonally dry regions, particularly if water for human consumption is directly obtained from the rivers, or if dams used as reservoirs are small (Smakhtin, 2001; Bruijnzeel, 2004). Complemented with other actions, the management of mountain landscapes to improve water provision can contribute to prevent water shortages and associated social conflicts during the dry season when the streamflows reach minimum values (Brauman et al., 2007; Berardo, 2014).

The success of ecosystem management depends on sound knowledge about the influence of landscape properties on key water fluxes (Ponette-González et al., 2014). One of the most important fluxes in seasonally dry social-ecological systems is the dry season baseflow. This flow depends on various factors, besides precipitation. After rainfall events, much of the water is rapidly lost through runoff, while only a portion is stored as groundwater, which can later be incorporated into the streams, if not consumed by plants or evaporated (Wittenberg and Sivapalan, 1999; Smakhtin, 2001; Laaha et al., 2013). The proportion of water which is lost by surface or sub-surface runoff, and

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hence not available for the dry season, depends on vegetation, topography, soil infiltration and soil storage capacity (Lewis et al., 2000; Bruijnzeel, 2004; Brauman et al., 2007; Yokoo et al., 2008). A good vegetation cover can favor water storage and later discharge in the dry season by protecting the soil and improving infiltration (Roa-García et al., 2011). Additionally in some ecosystems plants increase water gains to soil by catching fog (Ingraham and Mark, 2000; Ponette-González et al., 2014). But vegetation also produces water losses, by reducing throughfall inputs and consuming the water stored in the soil, partially or completely counterbalancing the gains (Bruijnzeel, 1989, 2004; Brown et al., 2005; Laaha et al., 2013; Ponette-González et al., 2014). Adding to this complexity, variations in vegetation type and abundance are often associated with other landscape features, such as topography or soil type and depth (e.g. Anchorena and Cingolani, 2002), which have their own influence on the soil water dynamics (Jobbágy et al., 2013; Laaha et al., 2013).

The complexity involved in the relationships between vegetation structure, topography, and soil with water discharge means that dry season baseflow is not easy to predict *a priori* from simple landscape proxies, nor from mechanistic models based in cause and effect relationships (Smakhtin, 2001; Yokoo et al., 2008; Blöschl et al., 2013; Wagener et al., 2013; Ponette-González et al., 2014). In this sense, measuring the local hydrological fluxes of interest and searching for relationships with land-cover and biophysical characteristics is of paramount importance (Ponette-González et al., 2014). Therefore, a comparative approach (Wagener et al., 2013), detecting empirically which landscapes, as combinations of topography and land-cover features, discharge more water per unit area in the dry season could be a good starting point. A comparative approach will give a strong basis for prediction and mapping dry season baseflow in ungauged basins (Laaha et al., 2013). This would facilitate the detection of water conservation priority areas, analyse synergies or trade-offs with other conservation needs, and design landscape management strategies, including the implementation of payments for water services (Juniper, 2013; Ponette-González et al., 2014). Also, this kind of approach will help to formulate more mechanistic hypothesis about the processes underlying the water provisioning services in areas under strong seasonal precipitation regimes.

An example of a markedly seasonal rainfall regime is the subtropical region of central Argentina where mountains play an important role as water providers. The headwater catchments located in the upper belt of the mountains (>1700 m a.s.l.) bring water to perennial rivers which supply this valuable resource to about three million people in the lowlands. The mountains of central Argentina are also important for biodiversity conservation as they harbor many endemic species and sub-species (Acosta, 1993; Nores, 1995; DiTada et al., 1996; Cabido et al., 1998). Livestock production is the main economic activity and has been traditionally managed through fires to clear woodlands and induce grass regrowth. Four centuries of these practices have produced significant soil erosion and woodland retraction, processes which are still active in large portions of the area (Cingolani et al., 2008, 2013, 2014; Renison et al., 2010). Concomitant with the degradation of the upper mountains, human occupation of the lowland area is experiencing an exponential growth (INDEC, 2012), with increases in the water demand and ever increasing water shortages during the dry season (Berardo, 2014; Dasso et al., 2014). Pine afforestation and the expansion of alien woody invaders constitute additional threats to the water supply service (Giorgis et al., 2011; Jobbágy et al., 2013; Zeballos et al., 2014). This scenario has raised human conflict together with public interest in the protection of mountain ecosystems (Berardo, 2014); but few studies exists, in this or in other seasonally dry subtropical areas, about the landscape factors controlling spatial variability in dry season baseflow (Bruijnzeel, 2004; Ponette-González et al., 2014).

In the present study, we aimed at detecting which landscapes, as combinations of topographical features and land cover types, are better suited to provide water during the dry season in a subtropical mountain of central Argentina. To meet this goal we evaluated the relationship between water discharge in the dry season and landscape variables in 16 small headwater catchments in the upper belt of the mountains.

2. Methods

2.1. Study area

The upper belt of the Sierras Grandes, in central Argentina (Córdoba Province), has a North–South orientation and the tectonic structure of a horst (Beltramone et al., 2002). The highest peak is at 2789 m a.s.l. but most of the area has an altitude between 2000 and 2300 m a.s.l. and consists of a dissected plateau, the "Pampa de Achala", remnant of an ancient crystalline peneplain (Cabido et al., 1987). Hills and plains are combined with gentle and deep valleys forming a very heterogeneous landscape. Most soils are Mollisols derived from the weathering of the granite substrate and fine-textured eolian deposits. Soils have high organic matter content, and in gentle valley floors are frequently waterlogged (Cabido et al., 1987; Cingolani et al., 2003). At upper topographic levels, soils tend to be shallow (<50 cm), with a sandy loam texture. At lower topographic levels soils are deeper (up to various meters, in deep valleys) and texture is finer, grading from silt loam to silty clay loam (Cabido et al., 1987). The landscape consists of a mosaic of *Polylepis australis* Bitter woodlands, tall tussock grasslands dominated by *Poa stuckertii* (Hack.), *Deyeuxia hieronymi* (Hack.) and/or *Festuca* spp., various kinds of short grazing lawns, rocky outcrops, and exposed rock surfaces due to recent soil erosion (Cingolani et al., 2004). Cover types are the result of long-term impact of livestock and fire interacting with topographic and physiographic features (Cingolani et al., 2003, 2008; Renison et al., 2015; Alinari et al., 2015). Woody aliens are expanding upwards from the lowlands and some plants or small groups can already be found above 2100 m a.s.l. (Giorgis et al., 2011 and A.M. Cingolani, pers. obs.).

Mean temperatures of the coldest and warmest months at 2200 m a.s.l. are 5.0 and 11.4 °C respectively, with no frost-free months (Colladon, 2004). The rainfall regime is monzonic, with mean annual precipitation around 900 mm concentrated in the warmest months (Table 1, Fig. 1; Colladon, 2014). From mean

Table 1
Altitudinal position and rainfall at four meteorological stations and 16 rain gauges located in or close to the study area. Rainfall for the three years of interest (1–3) is indicated in all cases, and the long term average, together with the measurement period, is also indicated for the four meteorological stations.

	Station 1	Station 2	Station 3	Station 4	Rain-gauges[a]
Altitude (m a.s.l.)	2380	2249	2200	1700	2017–2273
Rainfall (mm)					
Year 1 (2008–2009)[b]	1216	1051	868	898	–
Year 2 (2009–2010)[b]	873	749	799	830	587–768
Year 3 (2010–2011)[b]	755	752	730	714	492–809
Long term average	948	839	889	880	–
Years included in the long term average	1992–2012	2006–2012	1992–2012	1992–2012	2009–2011

[a] Data indicate the range of values.
[b] Each hydrological year is computed from September to August.

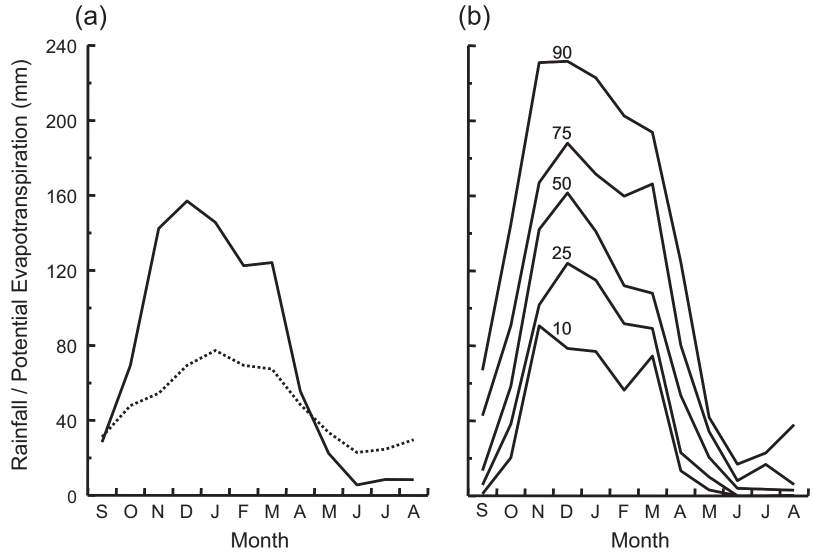

Fig. 1. Monthly climatic characteristics of the study area. (a) Average rainfall (solid line) and estimated potential evapotranspiration (dotted line). Rainfall (1992–2012) and temperature (1992–2003) data were obtained from Meteorological Station 3. (b) Monthly rainfall percentiles (indicated by a number above each line) from the same data set.

monthly temperature and light hours we estimated mean annual potential evapotranspiration as 576 mm (Thornthwaite, 1948; Colladon, 2004; Fig. 1). Snows occasionally occur in winter, covering the surface for only a few days. The driest months are usually June, July and August, but the start of the rain season can be delayed until November during extreme years (Fig. 1b). Summer intense precipitations trigger large stormflow events (Colladon and Vélez, 2011). Peak discharge in those events can be more than two orders of magnitude higher than average annual discharges, which were reported as 189–410 mm year^{-1} (*i.e.* 6–13 dm^3 s^{-1} km^{-2}) for perennial rivers originated in the upper mountain belt (Pasquini et al., 2004; Weber et al., 2005; Dasso et al., 2014; Supplementary data Fig. S1). Often, water in the downstream dams is preventively released, to avoid the flooding of the urbanized surroundings at the reservoirs and downstream. During the dry season, rivers are mainly supplied by groundwater stored in the soils, since the granite bedrock limits deep percolation of water (Beltramone et al., 2002).

The area is formally protected by a national park and reserve (Quebrada del Condorito) created in 1997, and a provincial water reserve (Pampa de Achala) created in 1999. The national park consists in 26,000 ha under state ownership (Fig. 2), while the national and provincial reserves in 12,600 and 117,500 ha respectively, under private ownership. Woodlands are expanding and soil erosion is decreasing in portions of the national park where livestock was excluded, and in several small woodland restoration sites within the provincial water reserve (Renison et al., 2013; Cingolani et al., 2013, 2014). In other sections of the national park, where livestock was maintained at low stocking rates to avoid biodiversity losses and prevent wildfires, moderate soil erosion still persists, particularly at upper topographic levels (APN, 2007; Vaieretti et al., 2013; Cingolani et al., 2013). In most of the privately owned protected areas, still managed in the traditional way with high stocking rates and the use of fire, woodlands are retracting and bare rock is in rapid expansion due to soil erosion (APN, 2007; Cingolani et al., 2013, 2014).

2.2. Study design

We selected 16 small headwater catchments (1.1–3.5 km^2) in the upper belt of the Sierras Grandes. Catchments were located between 2000 and 2200 m a.s.l. in the upper dissected plateau, distributed in an area of approximately 300 km^2 (central point of the study area at ca. 31°38′57″S, 64°47′51″W, Fig. 2). We searched for catchments with different land cover and topographic characteristics to represent as best as possible the whole range of landscape variation in the upper dissected plateau. To select the sites and initially mark the catchments divides, we used Google Earth, Landsat ETM images, and topographic and land cover maps included in a Geographic Information System (Cingolani et al., 2008). Finally, we corroborated at field the catchments' divides, and corrected them when necessary, using a Global Positioning System.

2.3. Rainfall data

We used three long-term series of monthly precipitation data and one shorter series from four meteorological stations close to the northern and northeastern part of the study area (Supplementary data Figs. S1–S4 and 2, Colladon, 2014). From each series, we calculated the total annual long-term average and the annual rainfall for three hydrological years of interest (2008–2009, 2009–2010 and 2010–2011, hereafter first, second and third hydrologic years), computed from September to August (Table 1). We were interested in these years because they were anterior to streamflow measurements (see Section 2.4).

Additionally, in September 2009 we installed a rain-gauge within each of the 16 catchments (Fig. 2). We used cooking oil to prevent water evaporation and visited rain-gauges periodically from the beginning of September 2009 to the end of August 2011. From these data, we obtained annual rainfall for the second and third hydrological years (2009–2010 and 2010–2011), but we lacked data for the first (2008–2009).

2.4. Streamflow measurements

At each catchment outlet, we selected one adequate segment in the channel to measure streamflow through the salt dilution method (Moore, 2004). We searched for segments with at least 10 m of massive rock floor without deep pools, and a narrow sector in the downstream point. The salt dilution method consists in the injection of a saline solution at a constant rate into the upper point

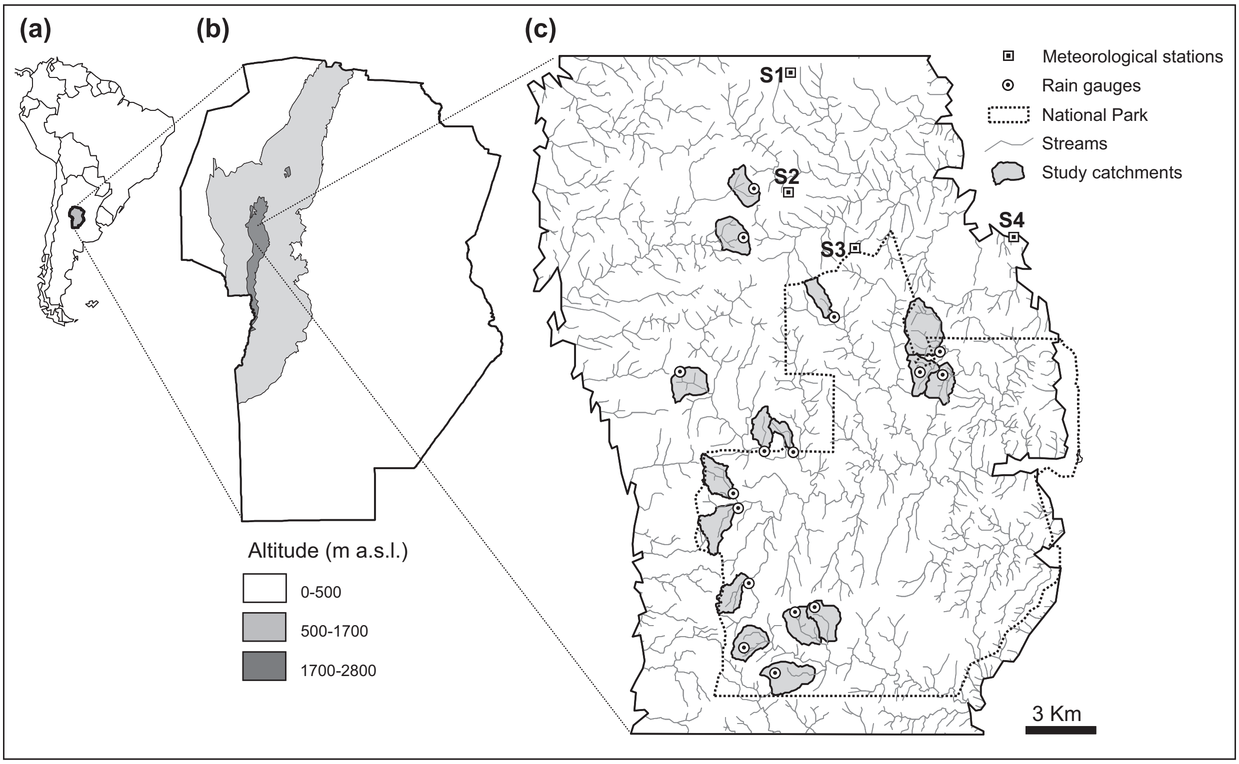

Fig. 2. Location of the study sites at (a) South America; (b) Córdoba province; and (c) The mountains above 1700 m a.s.l. where we indicate the location of the study catchments, rain gauges, meteorological stations and limits of the national park and surrounding national and provincial reserves under private ownership (see Google Earth).

of the stream segment, and measuring throughout time the electric conductivity at the downstream point until measurements stabilize at a high plateau. We calculated streamflow ($dm^3\ s^{-1}$) using data of the electric conductivity of the stream before injection, at the plateau, and of the saline solution, in combination with the injection rate value. This method has been recommended for small streams in mountain areas and has an error of about ±7% (Moore, 2004). It has been successfully used for similar streams in the study region (Jobbágy et al., 2013). We measured dry season streamflow four times at each catchment: in August 2009, August 2010, September 2010 and September 2011. Due to logistical reasons the measurements of the 16 catchments took various days at each date (21, 20, 7 and 8 days, respectively), but we changed the order in which we made the measurements to minimize systematic biases due to the advance of the dry season. For all dates, measurements were done before the onset of the wet season, and no rain events were registered during the measuring periods. For each date and catchment (4×16), we transformed the absolute streamflow values ($dm^3\ s^{-1}$) into streamflow per unit area ($mm\ day^{-1}$; $1\ mm = 1000\ m^3\ km^{-2}$), hereafter "water discharge".

2.5. Landscape and climatic variables

From a land cover map of the area (Cingolani et al., 2004) we calculated the proportion (%) of five land cover types at each catchment: rock outcrops, rock exposed by soil erosion (hereafter "exposed rock"), short lawns, tussock grasslands and woodlands.

From a Digital Elevation Model (25 m Aster Global DEM, ASTER GDEM, 2009) we obtained eight topographic variables for each catchment: average altitude (m a.s.l.), altitude range (maximum–minimum, m), average slope gradient (%), a roughness index (m), and the proportion (%) of plains, deep valleys, gentle valleys and hills. Average altitude and altitude range were obtained directly from the DEM. Average slope gradient and the proportion of plains were obtained from a slope gradient layer (%) calculated from the DEM. Pixels with less than 5% slope were considered as plains. Average roughness, the proportion of deep and gentle valleys, and the proportion of hills were calculated from a topographic position layer (m). The topographic position for each pixel in the layer was calculated as a negative or positive value which indicates its position below or above the surrounding landscape, respectively. For the calculation of this layer we used a circular kernel of 7 pixels diameter (175 m), and obtained the difference between two values: the vertical distance between the focal pixel and the lowest altitude of the kernel, and the vertical distance between the focal pixel and the highest altitude of the kernel. Thus, if the distance from the focal pixel to the lowest altitude was shorter than the distance from the focal pixel to the highest altitude, the value was negative, while a positive value was obtained in the opposite situation. The roughness index was defined from that layer as the standard deviation of the topographic position values of all pixels within the catchment. The proportion of deep and gentle valleys were defined as the proportion of pixels with topographic position values less than −9 m, and between −9 m and −3 m, respectively. The proportion of hills was defined as the proportion of pixels with values greater than 9 m.

We also calculated two rainfall layers for the second and the third hydrologic years (2009–2010 and 2010–2011), by interpolation from the 20 available recording points (16 rain gauges plus four stations), weighting by the inverse of the Euclidean distance. From these layers we estimated an average value of precipitation for each catchment and hydrologic year. For the first hydrologic year (anterior to the streamflow measures of August 2009) we only had data for the four meteorological stations, so we did not calculate a precipitation data layer nor a precipitation average value for each catchment.

2.6. Data analyses

2.6.1. Temporal and spatial patterns of rainfall and discharge variation

To analyze temporal patterns, we compared rainfall among the three hydrologic years through pair-wise paired *t*-tests using data from the four meteorological stations. Then we repeated the analysis for the second and third years using data from the 20 recording points. We also used paired *t*-tests to compare water discharge among the four dates (August 2009, August 2010, September 2010 and September 2011) across the 16 catchments.

To analyze if spatial patterns of rainfall were congruent among the three years, we performed pair-wise Pearson correlations across the four meteorological stations. Then we repeated the analysis for the second and third year across the 20 recording points. To analyze if spatial patterns of water discharge were congruent among dates, we performed pair-wise Pearson correlations across the 16 catchments. Since the water discharge was strongly correlated among dates (see Section 3.1), we calculated the average of the four dates as a synthetic variable.

Then, we analyzed if spatial patterns of rainfall were associated with spatial patterns of water discharge through Pearson correlations. For all dates except the first, we correlated water discharge with the rainfall of the anterior hydrologic year. We considered two alternative measures of rainfall: the one of the closest rain-gauge and that obtained from the interpolated layers. The first date was not analyzed because we had no detailed rainfall records per catchment.

2.6.2. Landscape characteristics and water discharge

We described and summarized landscape variability across catchments through Principal Component Analysis (PCA) using a correlation matrix (Afifi and Clark, 1984). We used a data matrix of 16 catchments × 13 variables (five cover types plus eight topographic variables). We considered PCA axes 1 and 2 as synthetic variables describing the main trends of landscape variation across space.

We searched for key landscape controls on water discharge through multiple regressions across catchments. We tested different models using average water discharge as the dependent variable. In a first approach, we included PCA Axis 1 and 2 as independent variables. In a second approach we tested all the possible combinations up to two individual landscape variables not correlated among them (Afifi and Clark, 1984). We did not test combinations of more than two variables to prevent overfitting. We then selected the three models with the highest r^2 out of all the models in which both variables were significant ($P \leqslant 0.05$). For the model using PCA axes and for the three selected models using individual variables, we tested the normality of residuals using Kolmogorov–Smirnov tests, and their Euclidean spatial autocorrelation using Moran index based in the inverse of the Euclidean distance (Diniz-Filho et al., 2003). We repeated all the analyses considering each date separately, and for all dates except the first, we incorporated the rainfall of the anterior year as an additional possible explanatory variable.

3. Results

3.1. Temporal and spatial patterns of rainfall and discharge variation

Annual rainfall decreased from the first to the third year, and differences between years were significant in almost all cases, either considering the four meteorological stations or the 20 recording points (Fig 3a). Water discharge differences between dates also were significant in almost all cases and consistent with the inter-annual variation of rainfall, and with stream water depletion due to the advance of the dry season (Fig. 3b). The highest and lowest discharge values were recorded in the first and the last date respectively, with an average within catchment reduction of 38 ± 8% (mean ± SE) between both dates, which in absolute terms represented 0.073 ± 0.014 mm day^{-1}.

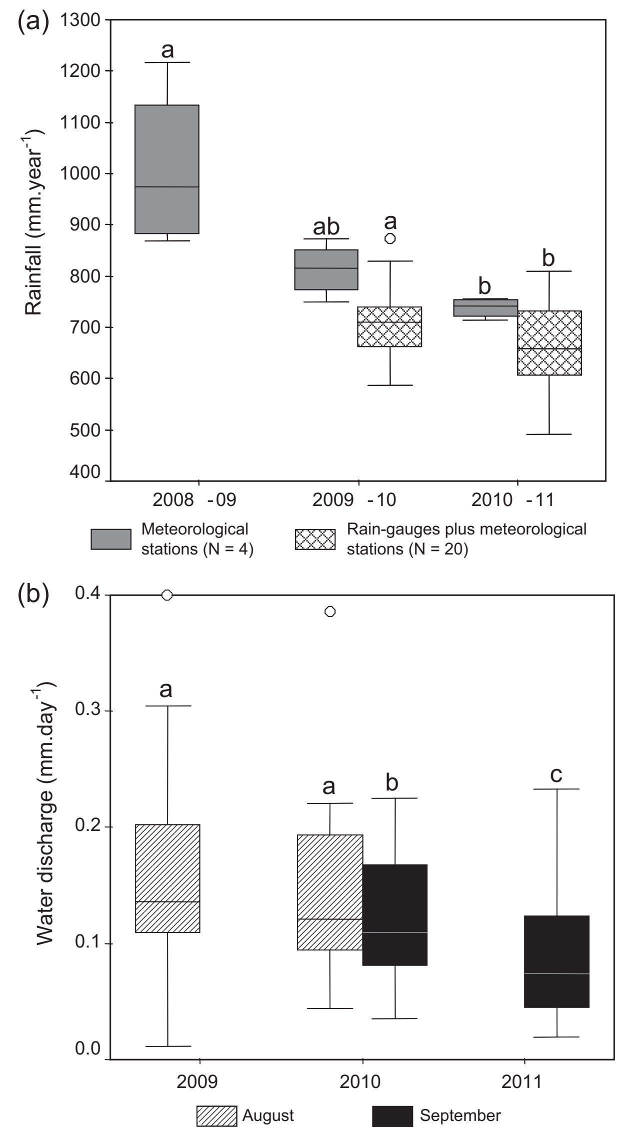

Fig. 3. (a) Box plots of the rainfall for the three hydrologic years of interest considering the four meteorological stations, and for the second and third years considering the 20 recording points (16 rain gauges plus four meteorological stations). Different letters indicate significant differences between hydrological years (paired *t*-tests, $P \leqslant 0.05$). For each data set, differences between the first and the second year, and between the second and the third using the four stations were close to significance, with $P < 0.08$ in both cases. (b) Box plots of the water discharge at four dates for the 16 catchments. Different letters indicate significant differences between dates (paired *t*-tests, $P \leqslant 0.05$).

Spatial patterns of rainfall were not congruent among years, indicating that no meteorological station or rain gauge consistently recorded more precipitations than the others (correlations across the four stations: $R = 0.39$, $P = 0.61$ for the first versus the second year; $R = 0.85$, $P = 0.15$ for the first versus the third year; $R = -0.06$, $P = 0.94$ for the second versus the third year; and across the 20 points: $R = 0.42$, $P = 0.07$ for the second versus the third year). In contrast, the spatial patterns of water discharge were congruent among dates, indicating that some catchments consistently discharged more water in the dry season than others (see correlations in Table 2). The within date differences between the

Table 2
Pair-wise Pearson correlations of water discharge among the four dates, across the 16 catchments. We also included the correlation between each date and the across-dates average.

	August 09	August 10	September 10	September 11
		Pearson R^a		
August 10	0.77	–	–	–
September 10	0.83	0.90	–	–
September 11	0.82	0.88	0.86	–
Average	0.92	0.94	0.95	0.94

[a] All Pearson correlations were statistically significant ($P \leqslant 0.05$).

catchments with highest and lowest water discharge were of 97%, 88%, 84% and 91% for the first to the last date, respectively (average 90 ± 3%), which in absolute terms represented 0.39, 0.34, 0.19 and 0.21 mm day^{-1} (average 0.28 ± 0.05 mm day^{-1}, Fig 3b). The spatial variation in water discharge was not correlated with the spatial variation in annual rainfall of the anterior hydrologic year in neither of the tested cases (R varying between −0.42 and 0.18, $P > 0.05$ in all cases).

The values reported in the previous paragraphs indicate that the temporal variation in water discharge within catchments (38 ± 8%) is lower than the spatial variation between catchments (90 ± 3%). In both cases, variations were larger than the measurement error of the salt dilution method (7%).

3.2. Landscape characteristics and water discharge

The landscape variability across catchments was summarized in PCA Axis 1, which explained 51.5% of the variance, and PCA Axis 2, which explained an additional 17.6%. PCA Axis 1 described a gradient from gentle to rugged landscapes (Supplementary data Figs. S2 and S3). Catchments with a large proportion of plains, gentle valleys and short lawns were located in the negative side; while steep and rugged catchments with a large proportion of deep valleys, hills and rocky outcrops were located in the positive side. PCA Axis 2 described a gradient from catchments with high proportion of exposed rock in the negative side toward catchments with high proportion of tussock grasslands in the positive side (Fig. 4).

Average water discharge in the dry season was highest in catchments positioned in the positive side of PCA Axis 1, dominated by steep and rugged landscapes; and lowest in catchments positioned in the negative side of PCA Axis 1, dominated by gentle landscapes (Table 3; Fig. 5a). Additionally, average water discharge showed a negative trend ($P = 0.06$) with PCA Axis 2. Though not significant, this suggest that, topographic factors being similar, water discharge in the dry season was higher in catchments with more proportion of exposed rock, in comparison with catchments with more tussock grasslands (Fig. 5b).

The best model using individual independent variables showed that catchments with the largest proportion of deep valleys discharged more water in the dry season (Table 3; Fig. 5c). An additional but smaller proportion of variance was explained by exposed rock, indicating that, deep valleys being equal, catchments with more exposed rock discharged more water (Fig. 5d). The second and third models using individual variables were very similar to the first, but including roughness and slope gradient, respectively, in the place of deep valleys. These three variables (deep valleys, roughness and slope gradient) were strongly correlated among them and with PCA Axis 1 ($R > 0.93$, $P < 0.001$ in all cases, Fig. 4a), indicating that the four alternative models described the same pattern of water discharge variation in relation to landscape type.

Residuals of the four models reported in Table 3 were normally distributed (in all cases Kolmogorov–Smirnov $Z > 0.62$; $P > 0.05$) and not autocorrelated (Z scores = −0.99, −0.60, −0.37 and −0.66, for the four models respectively, $P > 0.05$ in all cases). This indicates that results are not flawed due to spatial dependence (Diniz-Filho et al., 2003).

When analyses were repeated using individual dates, results were very similar, and rainfall of the anterior hydrologic year was not significant (results not shown). This indicates that at the scale of our study, spatial variation in rainfall did not substantially affect the spatial pattern of water discharge.

Table 3
Regression models for average water discharge (mm day^{-1}). First we showed the model run with PCA Axes 1 and 2 as independent variables, and then the three best models which combine two landscape variables not correlated among then.

	Coefficient	P	Explained Variance (%)	r^2
Intercept	0.128	<0.001	–	
PCA Axis 1	0.044	0.004	41	
PCA Axis 2	−0.026	0.061	15	0.557
Intercept	−0.037	0.398	–	
Deep valleys (%)	0.010	0.008	39	
Exposed rock (%)	0.004	0.046	17	0.560
Intercept	−0.121	0.094	–	
Roughness (m)	0.024	0.008	37	
Exposed rock (%)	0.004	0.049	16	0.535
Intercept	−0.117	0.100	–	
Slope gradient (%)	0.013	0.011	33	
Exposed rock (%)	0.004	0.033	20	0.534

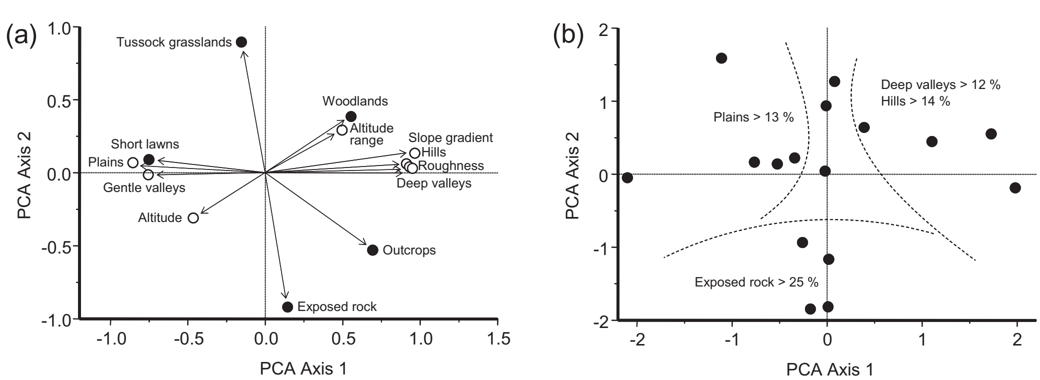

Fig. 4. (a) Contribution of land-cover (black circles) and topographic (open circles) variables to PCA Axis 1 and 2. (b) Location of catchments (black circles) along PCA Axis 1 and 2. With dashed lines we separated catchments with largest values of some landscape variables to aid in the interpretation of the plots.

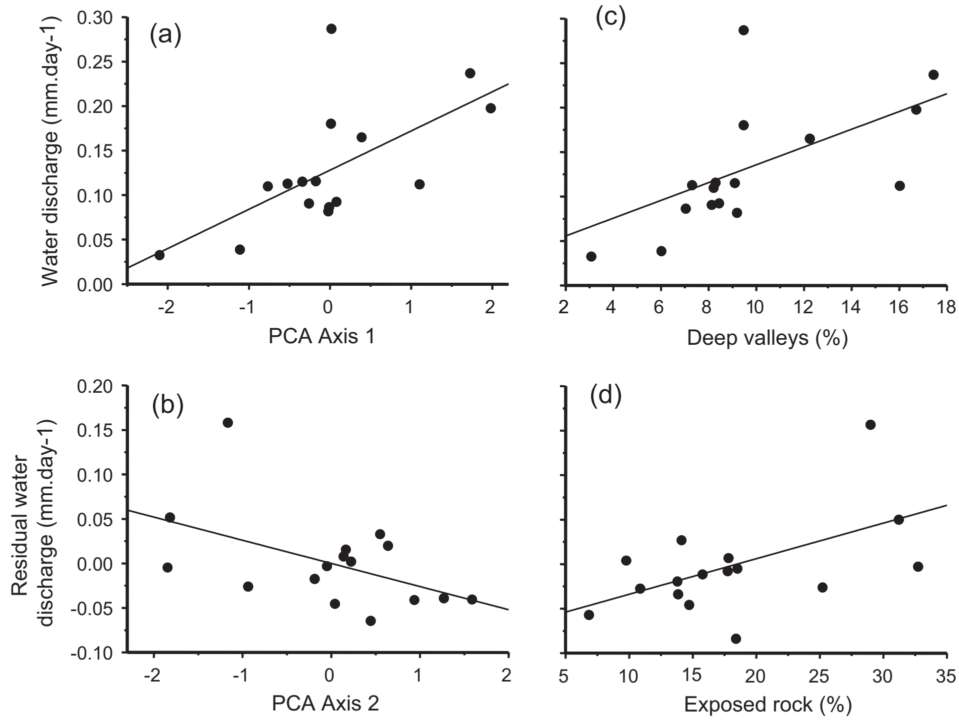

Fig. 5. (a) Average water discharge per catchment as a function of PCA Axis 1 and (b) residual water discharge (*i.e.* not explained by PCA Axis 1) as a function of PCA Axis 2. (c) Average water discharge per catchment as a function of the proportion of deep valleys and (d) residual water discharge (*i.e.* not explained by the proportion of deep valleys) as a function of the proportion of eroded rock. The corresponding models are reported in Table 3.

4. Discussion

4.1. Temporal and spatial discharge variation

Our dry season water discharge records are comparable with those obtained in other studies undertaken in the same mountain range (Table 4). The large variation observed within and between studies illustrates the large spatial and temporal heterogeneity in dry season baseflow. Understanding the factors controlling this heterogeneity is of paramount importance for water supply planning and design (Smakhtin, 2001; Laaha et al., 2013).

The differences among the four analyzed dates were consistent with expectations according to precipitations of the anterior hydrologic year and the gradual soil water depletion along the dry season, as described in Smakhtin (2001). These patterns illustrate that when recharge is reduced due to low precipitations in the rainy season, the water provisioning service in the next dry season is threatened, particularly if dry season extends toward the end of September or October. Most rivers originating in the study mountains discharge water in dams, but the water holding capacity of those reservoirs is not enough to cope with events of extreme low flows, and their water quality is significantly lowered (Fernandez et al., 2012; Dasso et al., 2014; Berardo, 2014). In addition, demand upstream of the reservoirs is rapidly increasing due to urban development (Berardo, 2014). Under this scenario, the water discharged into the channels during the dry season becomes highly valuable. Understanding its spatial variability and controlling factors is necessary to manage the landscape for water production and eventually, implement payments for water services (Ponette-González et al., 2014).

The spatial variation in water discharge across catchments was the main focus of our study. It was stronger than the temporal variation across dates, and consistent along the four dates analyzed. We are aware that the average, for a given catchment, calculated on the basis of only four measurements may not be an accurate estimator of the absolute August–September average discharge. However, the congruence among dates implies that the across-dates average may be a reliable relative estimator. Using this estimator, we were able to detect that rugged landscapes with steep slopes and a high proportion of deep valleys consistently discharge more water in the dry season than gentle landscapes. Since dry season baseflow depends on groundwater drainage into channels, our results mean that rugged landscapes have higher water percolation into deep soil layers, higher groundwater storage capacity and/or lower water losses through evapotranspiration (Wittenberg and Sivapalan, 1999; Smakhtin, 2001; Bruijnzeel, 2004).

Table 4
Range of water discharge values recorded in this and other studies through instantaneous measures in August and/or September months before the onset of the wet season. In each case, we indicated the number of catchments, the total number of records, and the range of recording dates (months and years). We also indicated the size and the altitudinal range of the catchments.

	This study	Gigantes[a]	Calamuchita[b]	S. Antonio[c]
Water discharge (m^3 km^{-2} day^{-1})[d]	20–400	0–273	27–1088	54–121
N° Catchments	16	2	8	1
N° Records	64	32	20	5
Months	August–September	August–September	August	August–September
Years	2009–2011	2010–2013	2004–2006	2009–2011
Size of catchments (km^2)	1.1–3.5	0.22–0.27	0.42–1.42	514
Altitudinal range (m a.s.l.)	1981–2286	2216–2386	1075–1983	650–2386

[a] Renison et al. (unpublished data).
[b] Jobbágy et al. (2013).
[c] Fernandez R. (pers. comm.).
[d] 1000 m^3 km^{-2} = 1 mm.

4.2. The role of topography and land cover

Our results suggest that deep valleys, more abundant in rugged and steep landscapes, play an important role as water reservoirs, while gentle valleys may be less important reservoirs. At upper topographic levels, either in gentle or rugged landscapes, soils rarely exceed the meter in depth (Cingolani et al., 2003, 2013), and remain saturated only a few days after rains, thus they may not play a role as reservoirs to maintain streamflow in the dry season (M. Poca unpublished data).

The importance of deep valleys is probably associated to their soils being several meters deep, as was observed at gullies or in soil profiles near stream banks. In contrast, soils in gentle valleys are often shallower, and have the water table close to the soil surface due to the entire saturation of the soil profile (Cabido et al., 1987; Cingolani et al., 2003, 2013). These soils get rapidly waterlogged in the wet season, and water losses are produced through saturation overland flow and high evapotranspiration, since plant roots reach the water table even in the dry season (Cabido et al., 1987; Bruijnzeel, 1989; Wittenberg and Sivapalan, 1999; Cingolani et al., 2003, 2004, 2008; Yokoo et al., 2008; Laaha et al., 2013). More detailed studies, analyzing the streamflow recession curves, would be necessary to better depict the mechanisms by which deep and gentle valleys recharge, store and discharge water (Wittenberg and Sivapalan, 1999; Smakhtin, 2001).

Water storage in the reservoirs of rugged landscapes could be further benefited by an extra water input from the surrounding uplands, which are steep and hilly (Nyberg, 1996; Gómez-Plaza et al., 2001; Qiu et al., 2001). We hypothesize that these properties, together with a high proportion of rocky outcrops, favor stormflow runoff into the deep valleys acting as reservoirs, where water may have good infiltration opportunities due to a lower long term impact of livestock and fire (Cingolani et al., 2008; Renison et al., 2015; M. Poca unpublished data). In contrast, uplands in gentle landscapes are less steep and have less rock outcrops, characteristics which promote *in situ* infiltration and later plant consumption, reducing runoff into the valleys (Qiu et al., 2001; Brauman et al., 2007).

Exposed rock in the landscape contributed slightly to increase water discharge in the dry season. This land cover is mainly associated to upper topographic positions, both in rugged and gentle landscapes (Cingolani et al., 2008). Although exposed rock is less steep than rocky outcrops, the lack of vegetation may contribute to drive some of the stormwater into the neighboring valleys (Cingolani et al., 2008, 2013). On the contrary, when uplands are less eroded and more covered by tussock grasslands (both variables were negatively correlated among them, $R = -0.82$ and Fig. 3a) more water infiltrates and eventually is consumed by plants, without reaching the reservoirs in the valleys.

Numerous studies have reported increases in water discharge after deforestation or reductions after afforestation, driven by a high water consumption of woody vegetation (Brown et al., 2005; Farley et al., 2005; Mark and Dickinson, 2008; Zha et al., 2010). On this basis, lower discharges would be expected in catchments with higher woodland cover, but our results were not consistent with this prediction, since woodland proportion was not significant in any regression model. As woodlands in the area tend to be better preserved in the deep valleys acting as reservoirs (Renison et al., 2006, 2015; Cingolani et al., 2008), they may be playing a role protecting soils from erosion, and improving infiltration, which could compensate a deeper exploration of the soil profile (Bruijnzeel, 2004; Neary et al., 2009; Bonell et al., 2010; Germer et al., 2010; Renison et al., 2010; M. Poca unpublished data). Nevertheless, it is possible that *P. australis* woodlands do not, in fact, consume more water than some grasslands. Greenness indices of woodlands in the growth season are similar to those of *P. stuckertii* tussock grasslands and short hydromorphic lawns, which are the grassland types dominating gentle valleys, suggesting a similar vegetation activity and water use at the community level (Cingolani et al., 2008). In line with this evidence, another study showed that *P. australis* leaf water potential in the growth season is similar or even lower (*i.e.* more negative) than that of *P. stuckertii* and some short herbaceous plants (M. Poca unpublished data), suggesting a similar or lower transpiration rate (Hacke et al., 2006). In addition, we know that *P. australis* growth is largely reduced in the dry season (Giorgis et al., 2010), thus reducing water consumption in the most limiting period.

4.3. Management implications

Management for water provision in the dry season should mainly focus on the conservation of deep valleys, which seem to be the key biophysical factor regulating dry season baseflow. The main issue in deep valleys is controlling woody invaders which to date are found at low densities but are quickly expanding upwards into the mountains. Woody aliens have an efficient water transport strategy, suggesting high water consumption, and tend to invade the valleys before other habitats (Giorgis et al., 2011; Zeballos et al., 2014). In many sectors of the mountains, including our study catchments, deep valleys are naturally protected from livestock and fire by their steep topography. This reduces the propagation of fires and livestock densities due to the difficult access in combination with increased predation, mainly by *Puma concolor* (Pia et al., 2013; Renison et al., 2015; Alinari et al., 2015). Thus, in our study sites soil erosion in deep valleys is mostly restricted to small gullies and erosion edges which can be stopped by locally excluding livestock thus stimulating the passive growth of vegetation (Renison et al., 2006; Cingolani et al., 2008, 2013).

Differently from what happens in the sector we have studied, in more populated parts of the water reserve soil erosion in deep valleys is a problem. There, *P. concolor* have been locally exterminated and fires have been ignited more frequently to allow for the maintenance of higher livestock densities. In consequence, today the landscape is highly degraded even in deep valleys, which have widespread soil erosion edges, deep gullies and patches of exposed rock (Cingolani et al., 2008, 2013, 2014). In these areas, the active restoration of the valleys might be necessary. An encouraging example is a 45 ha restoration project initiated in 1997 in one of the most degraded sectors of the Sierras Grandes. The project has been successful at reducing soil erosion by excluding livestock and actively restoring the vegetation cover in gullies and erosion edges, especially in the valleys. This was done together with the planting of 25,000 *P. australis* trees and other woody and herbaceous native species (views of the project area may be found in Google Earth at −31.415, −64.805 degrees of latitude and longitude respectively; Renison et al., 2005; Aronson et al., 2007; Landi and Renison, 2010). Also, some records indicate that dry season water discharge in this restored catchment is higher than in a neighboring and very similar, but not restored, catchment (D. Renison unpublished data). By prioritizing the conservation of rugged landscapes with abundant deep valleys, the management for water provision in the dry season will also benefit biodiversity because these landscapes harbor more richness and abundance of keystone and endemic species than gentle landscapes (García et al., 2008; Pia et al., 2013; Cingolani et al., 2010). This illustrates an interesting synergy between water provision services and conservation of biodiversity.

In uplands, the rocky surfaces contributed to increase water recharge of the reservoirs. Rocky surfaces may be natural outcrops or rock exposed by soil erosion produced by long-term livestock disturbance (Cingolani et al., 2008, 2013). The management implication from the point of view of water production, disregarding other considerations, is that soil erosion in uplands is slightly beneficial. Nevertheless, tolerating soil erosion for water production is in conflict to several other ecosystem services, such as provision of clean water, carbon capture and long term livestock production. Furthermore, the conservation of particular species which are only, or mainly, found in uplands could be jeopardized, or the large native herbivores which are being re-introduced into the area, *Lama guanacoe* (Cingolani et al., 2010; Flores et al., 2012). Thus, we suggest a management strategy where soil erosion in uplands is avoided in order to enhance alternative ecosystem services, and where soil conservation and alien control in deep valleys is prioritized for water services. Uplands, especially in gentle landscapes may be managed to sustain livestock at low densities to avoid soil loss, but maintaining low plant biomass values as compared to that with livestock exclusions. We could not evaluate the independent effect of short grazing lawns, which are low-biomass communities (Pucheta et al., 1998) since this land cover was associated with topography in our study catchments. However, according to the literature, maintaining short lawns in the uplands could favor runoff at some extent (Bruijnzeel, 2004), and the recharge of the valleys acting as reservoirs. However, this needs further research, fixing topography and experimentally changing vegetation cover.

The management implications of our study probably apply to several mountain ranges with similar topographical and climate characteristics such as the sub Andean mountains of Argentina. They might even apply to the vast high Andean *Polylepis* spp. belt, which from Venezuela to Argentina form an impressive 5400 km long patchwork of forest islands dominated by this genus (Gareca et al., 2010; Renison et al., 2013). Still, we suggest testing whether our results can truly be used in other mountain ranges by performing similar studies to ours. As all these mountain ranges are used for livestock production and soil erosion is often mentioned as a consequence of this activity, stakeholders could be stimulated to reduce or exclude livestock from deep valleys by payments for ecosystems services, thus compensating their economic loss from reduced livestock (Brauman et al., 2007; Juniper, 2013; Ponette-González et al., 2014).

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.jhydrol.2015.03.041. These data include Google maps of the most important areas described in this article (Fig. 2).

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