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Afforestations and wetlands, are they a good combination? Study of water fluxes in two cases of Patagonian wetlands

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ABSTRACT

Wetlands are ecosystems that play a key role in maintaining biodiversity and associated economies (e.g. agriculture and livestock) because they are reservoirs of water and carbon. Globally, these environments have been largely deteriorated, so their sustainable use takes on special importance. This study examined the effect on water recharge of the replacement of natural grassland by pine afforestation on hillsides adjacent to two Patagonian wetlands with contrasting rainfall. Results showed that independently of the rainfall at the site considered, canopies with high coverage values (90%) intercept 40% of the precipitation. This percentage is significantly reduced when canopy coverage is near 70%. Considering forested and grassland hillsides, groundwater drainage showed different patterns, consisting of only a few millimetres in the seasons recorded. Surface runoff was not a significant source of recharge for these systems in spring and summer, showing similar values in forested and grassland hillsides. Differences found in the recharge water variables between afforestation and grassland conditions cannot be directly associated with the vegetation in areas surrounding the wetlands. These systems showed high complexity, requiring site-specific analysis to determine what the impact of afforestation will be on their hydrology. Copyright © 2014 John Wiley & Sons, Ltd.

KEY WORDS wetlands; sustainable use; land use changes

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INTRODUCTION

Wetlands cover 6% of Earth's surface and contain about 12% of global carbon reserves, thus forming extremely important systems globally (Ferrati *et al*., 2005). The response of wetlands to the current climate change scenarios is considered one of the biggest questions, given their role in the dynamics of elements and material flows (IPCC, 2008). About half of global wetlands have been lost as a result of activities that have brought about deterioration, including factors such as highland clearing, urban settlement, deforestation and loss of land near floodplains and their margins by conversion to agriculture. All these land use changes have dried margins and altered drainage or dammed rivers that feed wetlands (Kundzewicz, 2003).

Specifically in Patagonia, there is a type of wetland called *mallín* (which in aboriginal Mapuche language means 'wet meadow'; Raffaele, 1999). These systems are azonal areas characterized by constant soil moisture throughout most of the year, hydrophilic herbaceous vegetation and meadow physiognomy. Regarding water availability, their flat-concave relief and relatively low position in landscapes, they receive semi-permanent or permanent water inputs via surface and/or subsurface (Ciari, 2009). The combined effects of precipitation, evapotranspiration and water interactions result in a distinctive surface and ground water dynamics pattern. Patagonian meadows (*mallines*) are distributed throughout the west-east gradient from the Andes hillside (with 2000-mm average annual precipitation) to the ecotone and steppe zones (with 300-mm average annual precipitation; Raffaele, 1999). Specifically, more than half the water that recharges these environments comes from glaciers and high mountaintops; water infiltrates deep into the underground aquifers and then surfaces in low areas at great distances (Ciari, 2009). At the east end of their distribution, *mallines* are important water reservoirs, as much for animal consumption as for herbaceous vegetation productivity, having 10 and 20 times greater forage productivity than the surrounding steppe and thus high environmental and economic regional importance (Burgos *et al*., 1996).

The conservation and sustainable use of *mallines* are of particular importance considering that they occupy only 1·5% of the Patagonian surface and historically have suffered varying degrees of deterioration. Additionally, inadequate management of these systems has caused an

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increase of their vulnerability as a result of changes in the original floristic composition and physical and chemical characteristics of soils (Perotti *et al*., 2005).

In this type of semi-arid region, an increase in vegetation cover can cause a decrease in water recharge of environments located at lower altitudes (Llorens *et al*., 2003; Farley *et al*., 2005). Thus, changes in land use can disrupt or alter the hydrological cycle at a given site, altering the rainfall and evapotranspiration balance through changes in the levels of interception, deep drainage and transpiration by vegetation, as well as runoff levels. Although changes in the dynamics of such systems have been studied for some regions, the existing information for other regions of the world is scarce or absent.

Globally, there is consensus that forests use more water than grasslands do (e.g. Vertessy *et al*., 2002; Farley *et al*., 2005). Thus, marked changes have been documented in different environments in the various components of water balance and dynamics, depending on the size of the afforestation area and the species planted (Ponton *et al*., 2006), climatic characteristics of each site (Huber *et al*., 2008; Little *et al*., 2009) and leaf area developed by each species (Fahey and Watson, 1991). In addition to changes in water flow, several studies show changes in soil properties (e.g. increased water retention capacity and macro-porosity, which promotes the infiltration and redistribution of water in the soil; e.g. Joffre and Rambal, 1988; Zheng, 2006).

The impact of afforestation on water flows in the NW Patagonia, Argentina, is a fairly well-studied process. This region has approximately 80 000 forested hectares of which 80% is ponderosa pine (*Pinus ponderosa* Doug. ExP., And Laws). Areas that are feasible for future afforestation extend to the east of the Patagonian region (steppe environments), where water recharge of wetlands is particularly important because they act as the summer water reservoir and are highly productive systems from the forage standpoint. To date, in northwestern Patagonia, studies have shown increased water consumption by plantations of *P. ponderosa* compared with surrounding grasslands (8–26% higher consumption depending on the climatic conditions of the growing season and planting density; Gyenge *et al*., 2002). Considering the increase in the rate of afforestation with this fast-growing species, the ecological and productive role of the *mallines* systems and the effects of afforestation found elsewhere in the world (e.g. Diaz and Rebori, 2002; Gyenge *et al*., 2002, 2009; Jobbágy and Jackson, 2004), in this study, it was hypothesized that (i) the replacement of natural grasslands by afforestation on hillsides surrounding *mallines* will increase the amount of rain intercepted by canopies decreasing both, superficial and deep drainage and (ii) the negative effect of hillside afforestation on water recharge of the *mallines* will be greater at xeric sites than udic sites, mainly due to differences in the amount of water inputs.

Thus, our aim was to quantify water recharge and supplying components of the *mallines* (precipitation, interception, deep drainage and runoff) within a basin with a *mallín* surrounded by hillsides with *P. ponderosa* afforestation and without afforestation (natural grassland), to determine the effect of afforestation on water recharge and storage, in two *mallines* of contrasting rainfall.

MATERIALS AND METHODS

Study area

The study area is located in the pre-mountain area in northwest Patagonia (Argentina). This region is characterized by a Mediterranean-type climate, with cold, humid winters (4 °C, 700-mm precipitation; De Fina, 1972) and hot, dry summers (16 °C, 150-mm precipitation; De Fina, 1972). Within this region, we selected *mallines* that have surrounded hillsides with *P. ponderosa* plantation (forested hillsides) as much native grassland hillsides. The two hillside conditions (forested and non-forested) had similar gradients and exposures to sun and wind.

Grassland areas were characterized by herbaceous and grasses species (*Festuca pallescens*; *Hordeum* spp.; *Deschampsia* spp.; *Stipa speciosa*, *Bromus* spp., *Azorrella* spp., *Fragaria chiloensis*, *Anemone multifida*, *Agrostis* spp. and *Puccinelia* spp.) and, to a lesser extent, shrub vegetation (*Mulinum spinosum*, *Acaena splendens* and *Rosa eglanterea* only present at the Udic site, see the following discussion), with characteristics of steppe environments with a high percentage of bare soil (20–68% in the Xeric and Udic site, respectively, both described in what follows). Forested hillsides with dense plantings (2 × 2 m) were selected, with low presence of understory herbaceous and shrub. The main herbaceous species in both centres *mallín* were *Juncus oleraceus*, *Carex* spp., *Ranunculus* spp. and *Trifolium repens*.

To assess the different impact of pine plantations with contrasting rainfall regimes on *mallines*, two environments of this type were selected: 'Udic site' (El Porvenir farm; 40° 06′ 50″S, 71° 09′ 56″W; annual average precipitation of 1294 mm) and 'Xeric site' (La Veranada farm; 41° 13′ 53″S, 71° 11′ 40″W; annual average precipitation of 800 mm) (Figure 1 and Table I). The sites were selected on the basis of historical regimes of average rainfall. In both cases, precipitation was principally distributed during autumn and winter. In winter, these environments are flooded, with the water table at surface. As spring and summer pass, the mallines are drying, remaining water only in the central area.

Both sites had a history of intensive livestock use and low current stocking (Table I), showing a high level of soil erosion on hillsides (grassland hillsides) and *mallín* edge area, which was more noticeable on the grassland hillside

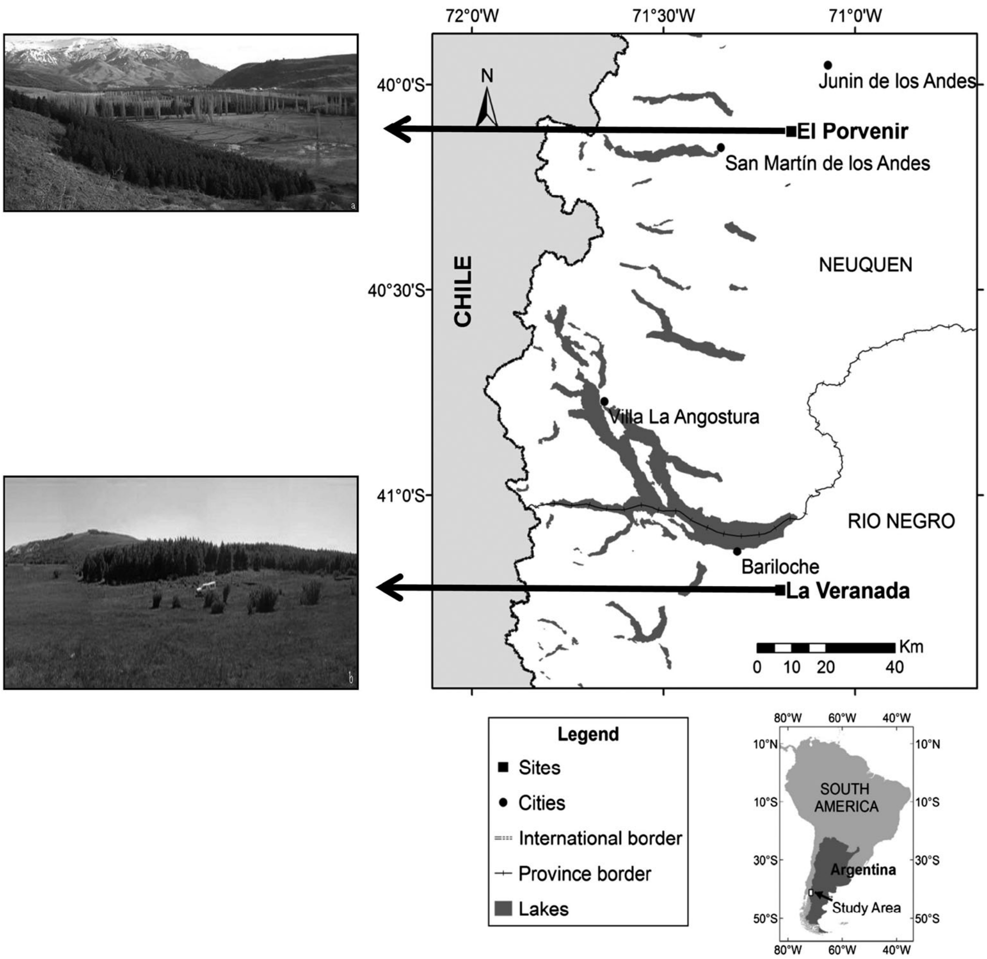

Figure 1. Location and photographs of the two selected sites (mallines). The Udic site was located in El Porvenir farm near the town of San Martin de los Andes and the Xeric site was located at La Veranada farm near the city of San Carlos de Bariloche.

Table I. Site and afforestation characteristics at the Udic and Xeric sites.

Summary of site features	Udic site	Xeric site
General characteristics		
Annual average precipitation (mm)	1294	800
Summer precipitation (mm)	330	269
Altitude (m)	861	1022
Afforestation characteristics		
Afforestation age (years)	23	23
Forested area (ha)	50	30
Afforestation density (tree/ha)	1620	909
Diameter at breast height (cm)	20	22
Height (m)	11	11
Site quality *	regular–bad	regular–bad
Type and cattle stocking in *mallín* and grassland conditions		
Beef cattle	1 individual/10 ha	—
Ovine cattle	—	1 individual/ha

**sensu* Andenmatten and Letourneau (1997).

at the Udic and on the forested hillside at the Xeric site. Winter rains erode the soil, generating grooves for water runoff and dragging fine material that is deposited downstream of the hillsides. The currently forested zone was previously subject to strong grazing pressure, leading to desertification processes due to overgrazing, commonly observed in other areas of Patagonia (Ares, 2007). Within the classification of Bonvissuto and Somlo (1998), studied *mallines* are within the so-called ‘Prairies of White Coirón (*F. pallescens*)’, rich in forage production in wet years, with plenty of water. Under fair and bad conditions, these systems have between 90% and 100% of ground coverage with dominance of forage species and a high dry matter production per year (1000–2000 kg ha^{-1} $year^{-1}$). Under poor conditions (due to overuse and water erosion) that increase the percentage of bare soil and reduce forage productivity, this variable ranges from 500 to 1500 kg ha^{-1} $year^{-1}$. Dominant species in these systems (*F. pallescens*) represents about 30–60% of vegetation cover under fair and poor conditions (corresponding to the conditions registered at Udic and Xeric site). From a topographical point of view, the Udic site in particular was characterized as a typical foothills *mallín* of NW Patagonia, and the Xeric site as a typical ecotone *mallín* (*sensu* Bonvissuto and Somlo, 1998).

Water recharge variables considered and quantified

The hydrology of a basin is determined by various factors that influence the water 'input' and 'output' processes in the system. The water balance is a one-dimensional model that describes the dynamics of soil moisture based on stochastic weather events regarding rainfall water partition from a long-term quantitative relationship (Eagleson, 1978). This combination of events is presented as

$$P = I + \Delta S + D + R + EP \qquad (1)$$

where P is precipitation, I is interception by vegetation, ΔS corresponds to the variation in soil moisture, D is deep drainage, R corresponds to runoff (surface and subsurface) and EP corresponds to evapotranspiration occurred during a given period. These variables were recorded fortnightly or monthly (depending on seasonal weather conditions) throughout spring, summer and autumn (September to April) of seasons 2005–2006 to 2008–2009.

In order to study the water fluxes in two cases of Patagonian wetlands, at each hillside (grassland and afforestation hillsides) within each *site*, precipitation (P) was recorded using rain gauges ($n = 3$, Figure 2). Interception (I) of rainfall was estimated from the difference in precipitation values recorded by the rain gauges installed in the grassland ($n = 3$) and afforestation hillside bellow tree canopy ($n = 9$) at each site (Figure 2). In order to establish the relationship between tree coverage values and the interception percentage, vertical canopy coverage was determined on each rain gauge by digital photographs obtained using a Nikon Coolpix 5400 camera with a Delta-T SCL8 fish eye lens (Delta-T Devices Ltd, Cambridge, UK). The camera was mounted on a tripod to ensure the horizontal position of the lens at 1 m above ground level and oriented towards the magnetic north. Photographs were taken in the early morning or late afternoon to avoid the influence of direct sun casting shadows inside the canopy and therefore errors in the estimates. Photo analysis was performed with the program Clear Light Analyzer v.2.0 (Frazer *et al*., 1999) in order to obtain the percentage of coverage and canopy openings. Before analysing hemispheric photos, some parameters were set. Image parameters are (i) magnetic north correction by calculating magnetic declination and (ii) adjusting the projection distortion as provided by the lens manufacturer. *A priori*, it was considered that the interception by shrubs and grasses corresponded to a very low percentage in relation to the tree interception, given its low coverage and the high percentage of bare soil (mainly at the Udic site).

Deep drainage (D) was estimated from the variation in the soil water content (θ) determined by the gravimetric method, according to the technique proposed by Rambal (1984) ($n = 2$ grassland hillside, $n = 2$ forested hillside; Figure 2). Soil samples were taken every 20 cm down to 1.2 m total depth using an edaphological auger at the beginning of the spring, time in which soil reached their field capacity because of the winter rains and snowfall. A surface of 9 m^2 was then covered with a 100-micron transparent polyethylene sheet, placing a barrier upstream of the covered ground perpendicular to the slope (using a 3×2 m piece of galvanized sheet iron). Thus, by inhibiting the direct input of precipitation, runoff or water loss by evaporation, deep drainage could be quantified. This procedure was repeated in each successive sampling season until the soil profile was found to be dry.

Surface runoff (R) was measured by installing permanent 1 m^2 plots ($n = 9$ on each forested and grassland hillside; Figure 2), surrounded by barriers built with wood and galvanized steel, following Kothyari *et al*. (2004). Surface drained water in each plot was collected in a graduated container. At each sampling point (plot), the slope of the terrain was measured using a hand clinometer (Suunto MM, Finland).

In order characterize the soils of each hillside, composite soil samples from the first 40 cm on each hillside at each site were collected and characterized according to structure, texture and chemical parameters (bulk density, % organic matter and total C, N, C:N ratio). For chemical and physicochemical characterization of soils, the methodology

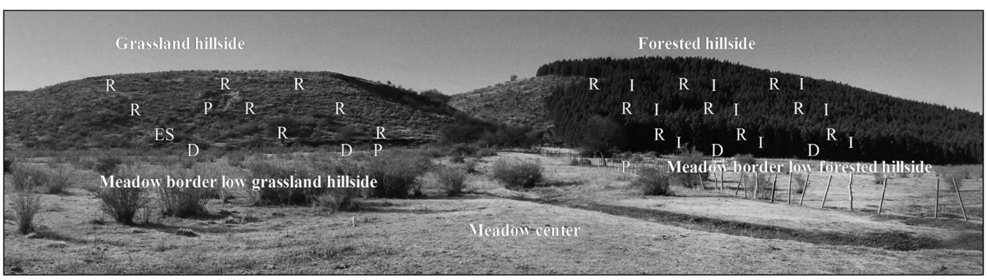

Figure 2. Location of the measuring devices in forested and grassland hillsides (sampling design). References: P = precipitation; I = intercept; D = drainage; R = runoff.

described by Sparks *et al.* (1996) was used, determined on air-dried samples sieved through 0·5-mm mesh, organic C (Walkley-Black) and total N (semi technical -micro Kjeldahl).

Statistical analysis

The average rainfall recorded during each successive season was compared with the historical average precipitation using Student's *t*-test for statistical comparison of means between two independent groups. The historical average precipitation values per month were estimated from data from the National Weather Service (Chapelco and Bariloche airports, and Udic and Xeric sites, respectively). In cases where the assumptions of the model could not be tested (normality and homogeneity of variance), the nonparametric Mann-Whitney test was used.

The rainfall intercept values for the two most representative tree cover percentages (70% and 90%) were compared by fitting the database to logarithmic models (Equation 2) using the program Table Curve 2D (Jandel Scientific Software AISN).

$$I = a + b^{*}\ln(P) \tag{2}$$

where I is interception, a is maximum interception with respect to precipitation, b is slope intercept function versus precipitation and P is the precipitation value recorded on each of the gauges. Subsequently, the models for each level of coverage at each site were compared using Fisher's *F*-test (Neter and Wasserman, 1974). In cases where the difference between models tested, the estimated parameters for each of the models were compared using Student's *t*-test (Sokal and Rohlf, 1995).

Additionally, in order to achieve a simplified interpretation and detect rainfall thresholds at which intercept values tended to be approximately constant, the results were adjusted to segmented linear models ('segmental linear regression', Draper and Smith, 1998), using the Prism5 program (GraphPad, San Diego, CA). This model was used as a supplement to the logarithmic models described. The structure of this model was

$$y_1 = \text{intercept}_1 + \text{slope}_1{}^{*}x \tag{3}$$

$$y \text{ when } x_0 = \text{slope}_1{}^{*}x_0 + \text{intercept}_1$$

$$y_2 = y \text{ when } x_0 + \text{slope}_2{}^{*}(x - x_0)$$

$$y = \text{if } (x < x_0, y_1, y_2)$$

where intercept_1 is the y value (that represents I) where the first line segment intersects the y-axis, slope_1 is the slope of the first linear function expressed in units of y divided by units x (that represent p-values), intercept_2 is the value of y where the second linear function (right end portion) intercepts the y-axis, slope_2 is the slope of the second line segment expressed in units of y divided by units x and x_0 is the value of x where the two line segments intersect. The first linear function defines the first line segment from a given slope intercept. The second linear equation computes the y-value of the right end portion of the first regression when $x = x_0$. The third linear function computes the second regression segment.

Segmented linear regression models by level of coverage at each site were compared by the maximum likelihood method using the Akaike information criterion. In each case where there were differences between the models for each slope zone, parameters of the models were compared using the global fit method (Motulsky and Christopoulos, 2004).

In order to estimate water millimetre drained in the soil over time, moisture content was analysed throughout the analysed seasons. Rambals (1984) function was used to relate soil water content over time:

$$\theta = a\, e^{-bt} \tag{4}$$

where a is the intercept of the relationship between soil moisture versus time (t) and b the slope thereof, the time derivative of this equation is

$$d\theta/dt = -abe^{-bt} \tag{5}$$

substituting t from Equations 4, 5 drainage was obtained according to the soil water content. Data were fitted to the Rambal function (1984) using the Prism5 program (GraphPad, San Diego CA), comparing models between hillside conditions (forested and grassland) by the *F*-test at each site. When there were differences between the models for each slope condition, parameters of the exponential function at each site were performed by the global fit method (Motulsky and Christopoulos, 2004).

For the comparisons of the runoff values between hillsides (forested and grassland) dataset of each site in relation to net precipitation (P–I) was adjusted to linear models using the Prism5 program (GraphPad, San Diego CA), and compared using *F*-test. When there were differences between the models for each condition, parameters of the lineal function at each site were performed by the global fit method (Motulsky and Christopoulos, 2004).

One-way ANOVA was carried out to test the differences in soil texture (percentages of clay, silt and sand) between hillside conditions (forested and grassland) at each site.

All tests listed were carried out using Statistica 7.0 (Stat Soft, Inc., Tulsa, USA) and Sigma Stat 3.5 for Windows (Systat Software Inc., Germany).

RESULTS

The values for accumulated rainfall per month in the 2005–2006 and 2006–2007 seasons were similar to historical values for the region (Udic site: 1200-mm annual precipitation, precipitation sampling season from September to April 330 mm; Xeric site: annual precipitation 800 mm, precipitation sampling season from September to April 269 mm, Figure 3). In the following two seasons, 2007–2008 and 2008–2009, the values were below historical averages. At the Udic site, precipitation was 34·8% and 64·5% of historical average rainfall for the seasons –2008 and 2008–2009, respectively, the difference being statistically significant only in the first case ($p = 0{\cdot}008$ and $p = 0{\cdot}104$; Figure 3). At the Xeric site, precipitation was 77·3% and 31·2% of historical average rainfall for the seasons 2007–2008 and 2008–2009, respectively, without statistically significant differences ($p = 0{\cdot}457$ and $p = 0{\cdot}064$ respectively; Figure 3).

The average values for total interception during the sampling seasons were 49·5 ± 35·2% for afforestation at the Udic site and 52·5 ± 44·8. % at the Xeric site (Table II). The dataset for tree coverage percentages for each site was subdivided into two subgroups of dataset by values close to 70% coverage (40% of the values reached that subgroup) and values close to 90% (60% of the values reached that subgroup). There were differences in the intercept values according to precipitation (in mm) between the two selected ranges of tree coverage percentages (Figure 4). Differences between models were due to an interaction between parameters (*a* and *b*) for both coverages within the same site ($p = 0{\cdot}0351$ at the Udic Site and $p = 0{\cdot}0348$ at the Xeric site, Figure 4). On the other hand, comparisons of segmented linear models between 70 and 90% coverage per site did not show statistically significant differences ($p = 0{\cdot}1900$ and $p = 0{\cdot}3400$ for Udic and Xeric sites respectively). For this reason, a single model was adjusted for each dataset per site, allowing us to set the threshold at which an increase in rainfall event generate a lower rate in the values of interception by the foliage (88 and 16 mm of precipitation in the Udic and Xeric site, respectively; parameter Xo; Figure 5).

Analysis of the four sampling periods shows that the average number of millimetre drained per day represents only a small proportion of precipitation over that time (Table II). The relationship between drainage and the soil water content showed a linear trend. At both sites, the differences between the slopes of these functions were significant for grassland and forested hillsides ($p < 0{\cdot}0001$), so it was not possible to compare the ordinates at the origin between the two hillsides. Drainage at the Udic site for the same soil moisture value was higher for the grassland hillside than the forested hillside (steeper slope in the linear function),

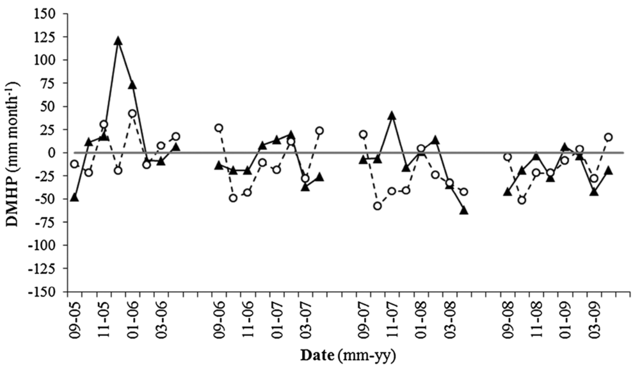

Figure 3. Difference between registered mean precipitation and historical mean precipitation per month (DMHP; mm month^{-1}) per season and site. White circles and segmented line show the precipitation difference at the Xeric site. Black triangles and full line show the precipitation difference at the Udic site. The grey line in the graph shows the value in which differences between registered mean precipitation and historical mean precipitation turns from deficit (negative values) to excess (positive values).

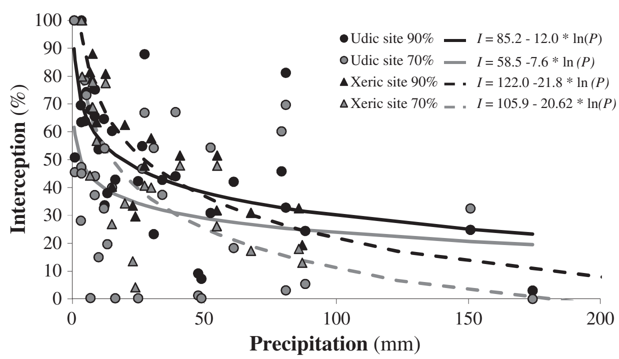

Figure 4. Relationship between the percentage of interception (*I*) and precipitation (*P*) at both sites fall under two foliar coverage percentages (90% and 70%). The equation that fitted the data was $I = a + b * \ln(P)$. Goodness of fit of the models were 0.15 and 0.64 for the 70% range, 0.44 and 0.68 for the 90% range at the Udic and Xeric site respectively.

Table II. Summary of the results obtained from the analysis of the water recharge variables registered at both study sites and hillside conditions.

	Udic site		Xeric site	
Summary of results	Afforestation	Grassland	Afforestation	Grassland
Precipitation (mm day^{-1})	1·46 ± 1·67		1·39 ± 1·46	
Interception (mm day^{-1})	0·72 ± 1·03	—	0·73 ± 1·18	—
Drainage (mm day^{-1})	0·04 ± 0·06	0·04 ± 0·08	0·02 ± 0·02	0·02 ± 0·02
Runoff (mm day^{-1})	0·04 ± 0·09	0·07 ± 0·14	0·03± 0·06	0·04 ± 0·09
Slope (°)	14·7 ± 6·6	17·5 ± 5·8	18·6 ± 6·6	8·1 ± 1·6

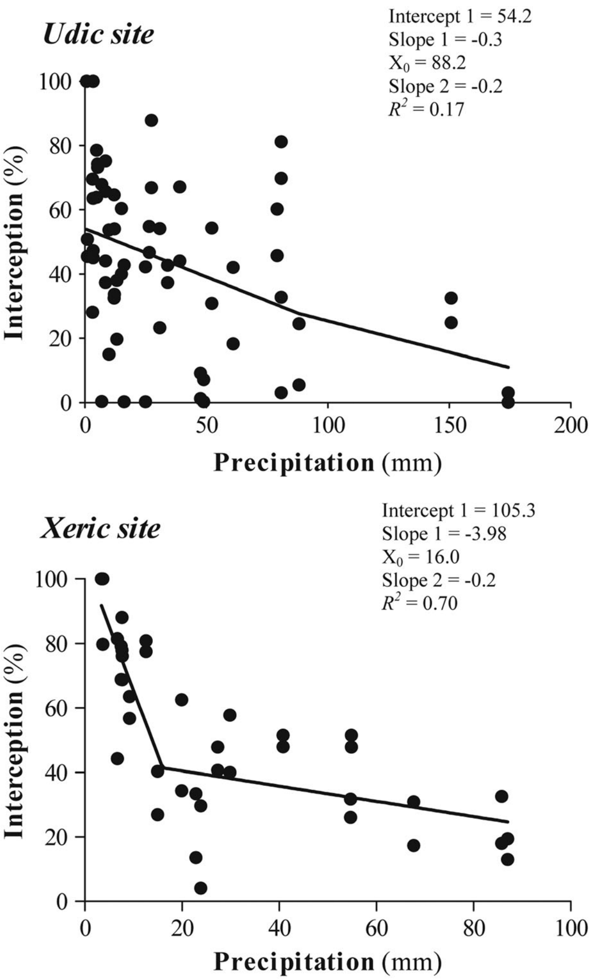

Figure 5. Simplified relationship between the percentage of interception and precipitation (mm) at Udic and Xeric sites using a single model segmented linear regression.

whereas for the Xeric site, the forested hillside had a greater slope than grassland hillside (Figure 6).

Surface runoff recorded throughout the spring and summer of successive studied seasons was low and highly variable; the overall average of the four seasons was $0{\cdot}07 \pm 0{\cdot}13\,\text{mm day}^{-1}$ at the Udic site (representing $3{\cdot}7 \pm 4{\cdot}9\%$ of precipitation) and $0{\cdot}03 \pm 0{\cdot}08\,\text{mm day}^{-1}$ at the Xeric site (representing $1{\cdot}8 \pm 2{\cdot}8\%$ of precipitation, Table II). Differences between slopes of lineal models were found at the Udic site between hillsides ($R = 0{\cdot}02120 * P + 0{\cdot}02918$; $r^2 = 0{\cdot}07$ for forested hillside and $R = 0{\cdot}05521 * P + 0{\cdot}01019$; $r^2 = 0{\cdot}37$ for grassland hillside; $p < 0{\cdot}0001$). Because the slopes differ so much, it is not possible to test whether the intercepts differ significantly.

In the Xeric site, no differences between slopes of the lineal models were found between hillside condition ($R = 0{\cdot}03476 * P + 0{\cdot}01507$ for forested hillside and $R = 0{\cdot}01907 * P + 0{\cdot}00929$ for grassland hillside; $p = 0{\cdot}1208$). Because the slopes are not significantly different, it is possible to calculate one slope for all the data (0·02099).

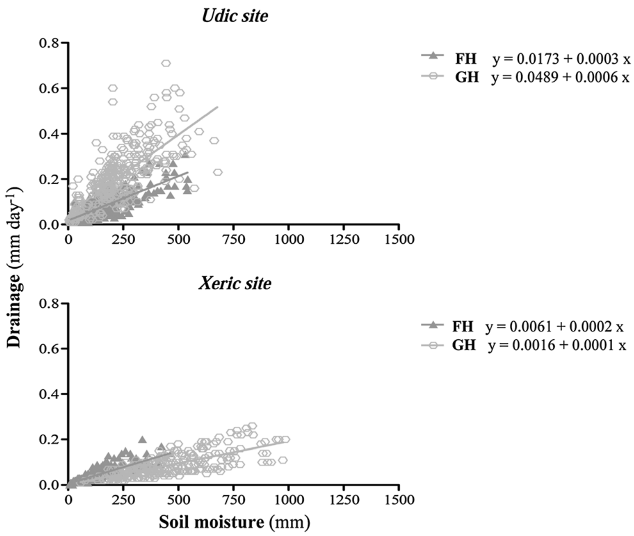

Figure 6. Relationship between drainage values (mm day^{-1}, estimated from the derivative of the water content in soil over time) and the soil water content (mm) recorded for each time point and sampling, in both, Udic and Xeric, site. References: FH = forested hillside; GH = grassland hillside.

Additionally, differences between the elevations of the lineal models were not quite significant. Because the R intercepts were not significantly different ($p = 0{\cdot}0632$), calculation of one R intercept for all the data was possible (0·01394).

Both sites had sandy loam soils, with predominantly sand percentages (20–25%) and silt (22–19%). Soil texture and nutrient availability were qualitatively different between hillsides per site, as well as between sites (Figure 7, Table III). At the Udic site, a greater proportion of sand was qualitatively observed in the soil of the grassland hillside. Only the percentage of silt showed statistically significant difference between hillside conditions, being higher in the forested hillside ($p = 0{\cdot}0320$). Organic matter, total nitrogen content and carbon-nitrogen ratio were qualitatively similar between hillsides (forested and grassland). On the other hand, at the Xeric site, qualitative differences in soil texture and nutrient composition between hillside conditions were observed. The

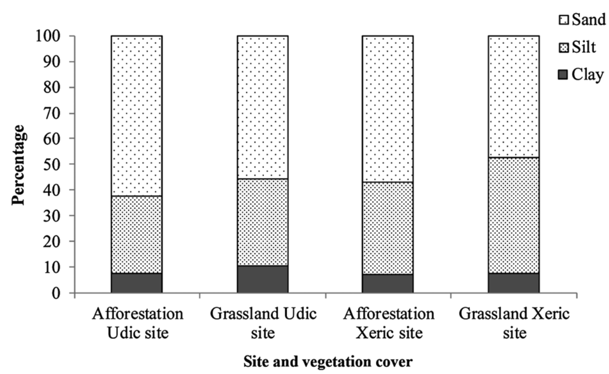

Figure 7. Percentages of soil textural components on Udic and Xeric sites of forested and grassland hillsides.

Table III. Characterization of forested and grassland hillsides soils of the Udic and Xeric site (0–40 cm).

	Udic site		Xeric site	
Soil characteristics	Afforestation	Grassland	Afforestation	Grassland
% OM	6·5	7·7	7·1	15·1
% N	0·2	0·3	0·3	0·5
C/N	14	14	14	15

References: % OM, organic matter content; % N, nitrogen percentage; C/N, carbon-nitrogen ratio.

proportion of sand was higher in the forested than in grassland hillside (Figure 7) without statistically significant differences between them. Organic matter and nitrogen content in grassland hillside was twice as high as in the forested hillside (Table III).

DISCUSSION

Canopy interception could represent a significant fraction of precipitation, which evaporates directly from it without reaching the soil, leading to a decrease in the water supply (Bryant *et al*., 2005; Echeverria *et al*., 2007). In this regard, it was noted that a 20% difference in canopy coverage (90–70%) significantly reduces the interception percentages, allowing increased input of rainfall to the soil of these systems. Our results suggest that the relationship between rainfall and interception was modulated by the level of tree cover, with the percentage of interception decreasing more in the Xeric site than in the Udic site considering similar rainfall events (Figure 5). Additionally, between-site differences were observed in precipitation event values (mm) from which the percentage of intercepted rain became constant (88 and 16 mm for Udic and Xeric site, respectively; Figure 6). Differences in these values may be due to differences in the characteristics of either forestation between sites, such as leaf surface area, or number of branches and type of precipitation event. Water saturation point of the crowns of *P. ponderosa* forest plantations located in the valley of Meliquina (Neuquén Province, with 800-mm annual precipitation) was close to 30 mm, after which water starts dripping from the canopy (Licata *et al*., 2010). In the present study, rainfall events greater than 30 mm usually comprise more than two consecutive days of rain and provide for partial drying of foliage and soaking (Licata *et al*., 2010), being rare in the summer months (growing season). In contrast, events below this value are characteristic in the region throughout the summer, and constitute the supply of water in the growing season (in addition to water stored in the soil, Gyenge *et al*., 2002). Thus, forest management to reduce coverage and leaf area (up to certain thresholds) would allow more input to the ground water without affecting tree growth.

Within the watersheds considered and upon analysing both hillside conditions (forested and grassland), drained millimetres over the seasons analysed were low, representing a small fraction of the water supply for recharging these systems at this time of the year. The same trend was observed in other *mallines* analysed in the Chubut Province, where small contribution of drainage was recorded in summer months; however, drainage is an important factor in the water balance during winter months (Ciari, 2009). Beyond the concrete contribution during the summer, drainage values between hillside conditions showed significant differences. At the Udic site, grassland hillside showed higher drainage values than forested hillside for the same soil moisture content. Conversely, in the Xeric site, higher drainage values were measured on the forested hillside. In both cases, the condition of soil texture at the Udic site (forested hillside) and Xeric site (grassland hillside) may favour retention (trend to higher content of clay and silt) and redistribution of water that enters the soil (as suggested by Fernandez and Trillo (2005)). Particularly at the Xeric site, the highest organic matter content recorded (higher water retention) explains that forested hillside soils, with a slightly gritty texture, have the highest drainage values. Although the difference in soil texture and composition between hillsides was not statistically significant, it may be enough to generate a significant difference in the drained millimetres per day. It is to be expected that the presence of trees would produce changes in soil properties in the medium and long term. Trees should increase the organic matter content and generate greater soil macro-porosity, making it more likely that water infiltration and drainage would reach the groundwater (Joffre and Rambal, 1988, La Manna *et al*., 2013). However, this was not observed in forested hillsides at the sites we analysed possibly because of the young age of afforestation due to which there has not yet been an increase in organic matter, and thus, the soil condition prior to planting would be more decisive on drainage.

On the other hand, runoff values recorded throughout spring and summer were very low, which allows us to infer that it is not important in recharging water in the *mallines* located downhill during summer. Upon analysing each particular site, we found that in the Udic site, runoff values were higher on grassland hillsides than on forested hillsides. The grassland hillside has a large percentage of bare soil and steeper slope than the forested hillside, which may explain the aforementioned pattern. In relation to this, several authors report that as the vegetation becomes denser, runoff values are lower because denser vegetation increases retention of water from precipitation by promoting infiltration (Farmer *et al*., 2003; Pizarro Tapia *et al*., 2006).

During the winter months, the time of the year when most of the precipitation occurs (approximately 70% of total precipitation), the foliage is saturated and interception is negligible. Water that reaches the soil exceeds field capacity generating a significant contribution of surface and subsurface runoff and drainage in hillside areas leading to a rise of the water table in the *mallines* leaving these systems flooded (personal observations; Ciari, 2009). Therefore, if there is an impact of the afforestations on water recharge of those wetlands, it was expected that it could be detected at the time of the year with water shortage (during austral spring and summer) and with a more marked effect over the Xeric site.

In relation to the location of the *mallines* in the precipitation gradient, there was no a tendency confirming a greater and differential effect by afforestation at the Xeric site. Intercept values were high at both sites, showing similar tendencies. Contrasting results for drainage and runoff at both sites were consistent with the characteristics of vegetation density, organic matter content and soil texture, which appears to be the most determining factor on this variable (Johnston *et al*., 2001).

Although this study analyses the input variables at surface, water in *mallines* drains slowly and sub-superficially during the growing season, revealing the importance of hydrological studies in this kind of systems and their relationship to the surrounding environments as areas of contribution (Ciari, 2009; Jobbagy *et al*., 2011). This suggests that environments surrounding the *mallines* have a high capacity to capture water resources, allowing infiltration into the soil and transferring it to *mallines*. Subsurface water that recharges aquifers from early autumn to early spring can move slowly and reaches low relief areas where it rises throughout the year in *mallín* systems together with the temporary surface pathways (Lanciotti *et al*., 1992). In this regard, studies carried out in *mallines* analysed show that afforestation does not have a direct effect on groundwater levels (Weigandt *et al*., 2011).

CONCLUSIONS

Despite the high interception registered, surface runoff did not decrease under the forested hillsides with relation to grasslands hillside, and accounted for only a small contribution to the water recharge of *mallines* located downstream to the hillside in summer. Deep drainage, such as surface runoff, represents a small percentage of water recharge in *mallines*, possibly dependent on the soil texture and organic matter content. This study highlights the importance of analysing the hydrological behaviour of each sector of these environments for each particular site.

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REFERENCES

Andenmatten E, Letourneau F. 1997. Curvas de índice de Sitio para Pinus Ponderosa (Dougl.) Law de aplicación en la región Andino Patagónica de Chubut y Río Negro, Argentina. *Bosques* **18**(2): 13–18.

Ares J. 2007. Systems valuing of natural capital and investment in extensive pastoral systems: Lessons from the Patagonian case. *Ecological Economics* **62**: 162–173.

Bonvissuto G, Somlo R. 1998. Guías de condición para los campos naturales de Precordillera y Sierras y Mesetas de Patagonia. Prodesar INTA-GTZ, 24.

Bryant M, Bhat S, Jacobs J. 2005. Measurements and modeling of throughfall variability for five forest communities in the southeaster US. *Journal of Hydrology* **312**: 95–108.

Burgos A, Lanciotti M, Bonvisutto G. 1996. Evaluación de Balance Hídrico en un Mallín precordillerano. In *Acta XV Congreso Argentino de la Ciencia del Suelo*. Santa Rosa, La Pampa.

Ciari G. 2009. Transferencia y Exportación de Recursos en una Cuenca Serrana de Patagonia Extra Andina. Tesis para optar por el grado de Magíster, Facultad de Ciencias Agrarias y Forestales-Universidad Nacional de La Plata.

De Fina A. 1972. El clima de la región de los Bosques Andino-Patagónicos. *La región de los Bosques Andino-Patagónicos, Sinopsis General*. Dimitri M. (Ed.). Instituto Nacional de Tecnología Agropecuaria: Buenos Aires; 35–58.

Díaz R, Rebori G. 2002. Redistribución de las lluvias y balance de agua en una plantación de Eucalyptus dunni en el sur de Santa Fé (Segunda Parte). *SAGPyA Forestal* **24**: 14–17.

Draper NR, Smith H. 1998. *Applied Regression Analysis*. 3rd edn, John Wiley & Sons, Inc.: New York.

Eagleson, PS. 1978. Climate, soil, and vegetation: 1. Introduction to water balance dynamics. *Water Resour. Res.* **14**(5): 705–712. DOI: 10.1029/WR014i005p00705

Echeverría C, Hubber A, Taberlet F. 2007. Estudio comparativo de los componentes del balance hídrico en un bosque nativo y una pradera en el sur de Chile. *Bosque* **28**(3): 271–280.

Fahey B, Watson A. 1991. Hydrological impacts of converting tussock grassland to Pine plantation, Otago, *New Zealand. Journal of Hydrology* **30**(1): 1–15.

Farley K, Jobbágy E, Jackson R. 2005. Effects of afforestation on water yield: a global synthesis with implications for policy. *Global Change Biology* **11**: 1565–1576. DOI: 10.1111/j.1365-2486.2005.01011.x

Farmer D, Sivapalan M, Jothityangkoon C. 2003. Climate, soil, and vegetation controls upon the variability of water balance in temperate and semiarid landscapes: Downward approach to water balance analysis. *Water Resources Research* **39**(2): 1–21.

Fernández RJ, Trillo N. 2005. La textura del suelo como fuente de heterogeneidad; sus efectos sobre la oferta de agua para las plantas. In *La heterogeneidad de la vegetación de los agroecosistemas. Un* *homenaje a Rolando León*, Oesterheld M, Aguiar MR, Ghersa CM, Paruelo JM (eds). Editorial Facultad de Agronomía: Buenos Aires, 171–192.

Ferrati R, Canziani GA, Moreno DR. 2005. Estero del Ibera: hydrometeorological and hydrological characterization. *Ecological Modelling* **186**: 3–15.

Frazer GW, Canham CD, Lertzman KP. 1999. Gap Light Analyzer (GLA), Version 2.0: imaging software to extract canopy structure and gap light transmission indices from true-colour fisheye photographs, users manual and program documentation. Copyright © 1999: Simon Fraser University, Burnaby, British Columbia, and the Institute of Ecosystem Studies, Millbrook, New York.

Gyenge JE, Fernández ME, Dalla Salda G, Schlichter TM. 2002. Silvopastoral system in Northwester Patagonia II: water balance and water potential in a stand of *Pinus ponderosa* and native grassland. *Agroforestry Systems* **55**: 47–55.

Gyenge JE, Fernández ME, Schlichter T. 2009. Effects on site water balance of conversion from native mixed forest to Douglas-fir plantation in NW Patagonia. *New Forests* **38**: 67–80.

Huber A, Iroume A, Bathurst J. 2008. Effect of *Pinus radiata* plantations on water balance in Chile. *Hydrological Processes* **22**(1): 142–148.

IPCC (Intergovernmental Panel on Climate). 2008. Climatic change and water. Intergovernmental Panel on Climate Change. IPCC Technical Paper VI. Bates B, Kundzewicz Z, Wu S, Palutikof J. WMO y UNEP.

Jobbagy EG, Jackson RB. 2004. Groundwater use and salinization with grassland afforestation. *Global Change Biology* **10**: 1299–1312.

Jobbagy EG, Nosetto MD, Villagra PE, Jackson RB. 2011. Water subsidies from mountains to deserts: their role in sustaining groundwater-fed oases in a sandy landscape. *Ecological Application* **21**(3): 678–694.

Joffre R, Rambal S. 1988. Soil water improvement by trees in the rangelands of southern Spain. *Acta Oecologica* **9**: 405–422.

Johnston CA, Bridgham SD, Schubauer-Berigan JP. 2001. Nutrient dynamics in relation to geomorphology of riverine wetlands. *Soil Science Society of America Journal* **65**: 557–577.

Kothyari BP, Verma PK, Joshi BK. 2004. Rainfall-runoff-soil and nutrient loss relationships for plot size areas of bhetagad watershed in Central Himalaya, India. *Journal of Hydrology* **293**: 137–150.

Kundzewicz ZW. 2003. Ecohydrology for sustainable wetlands under global change – data, models and management. In *Wethydro Center of Excellence in Wetland Hydrology Measurement Techniques and Data Assessment in Wetland Hydrology*, Ignar E, Nowakowsk P, Okruszko T (eds). Warsaw Agricultural University Press: Warsaw, Poland; 25–36.

La Manna L, Buduba C, Gigli A, Rostagno CM. 2013. Efecto de las plantaciones sobre la erosión hídrica potencial en suelos degradados de la Región Andino Patagónica. II Jornadas Forestales de Patagonia Sur. 2º Congreso Internacional Agroforestal Patagónico. El Calafate, Santa Cruz, Argentina.

Lanciotti ML, Bonvissuto GL, Bellati J, Somlo R. 1992. *Mallines: suelos, vegetación, fauna y nutrición animal. Recopilación bibliográfica*. IV Jornadas de Suelos de la Patagonia, Pto.: Madryn, Chubut.

Licata J, Pypker TG, Weigandt M, Unsworth MH, Gyenge JE, Fernández ME, Schlichter TM, Bond BJ. 2010. Decreased rainfall interception balances increased transpiration in exotic ponderosa pine plantations compared with native cypress stands in Patagonia, Argentina. *Ecohydrology* **4**(1): 83–93.

Little C, Lara A, Mc Phee JY, Urrutia R. 2009. Revealing the impact of forest exotic plantations on water yield in large scale watersheds in South-Central Chile. *Journal of Hydrology* **374**: 162–170.

Llorens P, Poch R, Latron J, Gallart F. 2003.Rainfall interception by a *Pinus sylvestris* forest patch overgrown in a Mediterranean mountainous abandoned area I. Monitoring design and results down to the event scale. *Journal of Hydrology* **199**: 331–345.

Motulsky HJ, Christopoulus A. 2004. *Fitting Models to Biological Data Using Linear and Non-linear Regression*. GraphPad Software Inc.: San Diego, California.

Neter J, Wasserman W. 1974. *Applied Linear Statistical Models. Regression, Analysis of Variance and Experimental Design*. R. D. Irwin Inc.: Illinois, EEUU.

Perotti M, Diéguez M, Jara F. 2005. Estado del Conocimiento de Humedales del Norte Patagónico (Argentina): Aspectos Relevantes e Importancia para la Conservación de la Biodiversidad Regional. *Revista Chilena de Historia Natural* **78**: 723–737.

Pizarro Tapia R, Tapia Cornejo M, Arellano L, Jordán Díaz C, Frías Daza C. 2006. Coeficiente de escorrentía instantánea para la cuenca del río Tutuvén, VII Región del Maule, Chile. *Bosque* **27**(2): 83–91.

Ponton S, Flanagan LB, Alstad KP, Johnson BG, Morgenstern K, Kljun N, Black TA, Barr AG. 2006. Comparison of ecosystem water-use efficiency among Douglas-fir forest, aspen forest and grassland using eddy covariance and carbon isotope techniques. *Global Change Biology* **12**(2): 94–310.

Raffaele E. 1999. Mallines: aspectos generales y problemas particulares In *Tópicos sobre humedales subtropicales y templados de Sudamérica*, Malvarez AI (ed.). UNESCO: Montevideo, Uruguay; 27–33.

Rambal S. 1984. Water balance and pattern of root water uptake by *Quercus coccifera* L. evergreen scrub. *Oecolgia* **62**: 18–25.

Sokal RR, Rohlf FJ. 1995. *Biometry: The Principles and Practice of Statistics in Biological Research*. 3rd edn, W. H. Freeman & Co.: New York, 887.

Sparks DL, Page AL, Helmek PA, Loeppert RH, Soltanpour PN, Tabatabai MA, Johnston CT, Sumner ME (eds). 1996. *Methods of Soil Analysis*. Part 3. Chemical Methods. SSSA. Book Series N° 5. SSSA, ASA: Madison, Wisconsin.

Vertessy R, Zhang L, Dawes W. 2002. Plantations, river flow and river salinity. In *Proceeding of the Prospects for Australian Forest Plantation 168 conference*, 20-21 August 2002, Gerrand A (ed.). Bureau of Rural Sciences: Canberra, Australia, 29–40.

Weigandt M, Gyenge J, Fernández ME, Varela S, Schlichter T. 2011. Is forage productivity of meadows influenced by the afforestation of upstream hillsides? A study in NW Patagonia. *Forest Systems* **20**(1): 165–175.

Zheng F. 2006. Effect of vegetation changes on soil erosion on the loess plateau. *Pedosphere* **16**(4): 420–427.

