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Effects of afforestation on groundwater recharge and water budgets in the western region of Uruguay

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Abstract:

Uruguay has stimulated the development of its forest sector since the promulgation of Forest Law N° 15 939 in December of 1987. Nevertheless, the substitution of natural grasslands with forest plantations for industrial use has raised concerns regarding hydrological processes of groundwater recharge and water consumption involving evapotranspiration. The purpose of this study is to assess the effects of this substitution approach on water resources. Input data were collected from two small experimental watersheds of roughly 100–200 hectares located in western Uruguay. The watersheds are characterized by *Eucalyptus Globulus ssp. Maidenni* and natural grasslands for cattle use. Total rainfall, stream discharge, rainfall redistribution, soil water content and groundwater level data were collected. Groundwater recharge was estimated from water table fluctuations and from groundwater contributions to base flows. Seasonal and annual water budgets were computed from October of 2006 to September of 2014 to evaluate changes in the hydrological processes. The data show a decrease in annual specific discharge of roughly 17% for mean hydrological years and no conclusive effects on annual groundwater recharge in the forested watershed relative to the reference pasture watershed. Reduced annual specific discharge is equivalent to the mean annual interception. The computed actual annual evapotranspiration is consistent with international catchment measurements. Reduction rates vary seasonally and according to accumulated rainfall and its temporary distribution. The degree of specific discharge decline is particularly high for drier autumns and winters (32 to 28%) when the corresponding rainfall varies from 275 to 400 mm. These results are of relevance for water resources management efforts, as water uses downstream can be affected. These findings, based on a study period dominated by anomalous wet springs and summers and by dry autumns and winters, oppose earlier results based on 34 years of rainfall and discharge data drawn from Uruguayan large basins. Copyright © 2016 John Wiley & Sons, Ltd.

KEY WORDS *Eucalyptus* plantations; water budget; groundwater recharge; experimental watersheds

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INTRODUCTION

Uruguay is a country located within the temperate zone of South America and is characterized by a humid subtropical climate (Silveira and Alonso, 2009). The country's landscape is gently undulating with very smooth slopes. Traditional land uses employed since the XVIII century include natural grasslands for livestock. However, Forest Law No. 15 939 enacted in 1987 has encouraged the conversion of natural grasslands with less productive soils classified as 'lands with forest priority' into *Eucalyptus* and *Pinus* plantations. As a consequence, the country's plantation area has expanded from some 45 000 ha at the end of the 1980s to roughly 1 150 000 ha today. Foreign companies have constructed two pulp mills on the banks of the Uruguay and de la Plata rivers in western Uruguay, and the government is discussing the eventual construction of a third pulp mill with foreign companies. Therefore, the measured effects of forest plantations on water yields presented in this paper are expected to remain the same in the near future or to even increase because of a potential expansion of planted areas.

The effects of land-use changes, e.g. replacing natural prairies with forest plantations, on the hydrological cycle have been a cause for concern and have motivated extensive research based on experimental watersheds (Bosch and Hewlett, 1982; Bruijnzeel, 1990; Calder, 1992, 2005; Best *et al.*, 2003; Andréassian, 2004; Genereux *et al.,* 2005; Birkinshaw *et al*., 2011). Two methods are usually applied to understand these changes: paired watersheds or long time series from single watersheds (Birkinshaw *et al*., 2014). Afforestation reduces surface water availability across a broad range of climates (Farley *et al.*, 2005). Forest plantations

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intercept incident rainfall and tend to lose more water through transpiration than grasses mainly because of their extended root systems (Bosch and Hewlett, 1982; Andréassian, 2004; Brown *et al*., 2005; Chang, 2006; Birkinshaw *et al*., 2011). However, such affects appear to fluctuate depending on climatic patterns and tree species present. In regards to groundwater recharge, some researchers have found a decreasing pattern (Finch, 1998; Le Maitre *et al.*, 1999; Allen and Chapman, 2001; Calder, 2007; Birkinshaw *et al.*, 2011; Fan *et al.*, 2014). However, García Préchac *et al.* (2001) found less soil water retention at 10 kPa (field capacity) in different Uruguayan experimental plots of *Eucalyptus* plantations compared to retention levels found for the same soils covered by natural pastures. Cerdà *et al.* (1998) and Ferreira *et al.* (2015) observed changes in soil properties associated with drought periods, during which rainfall shortages cause organic surfaces in soils to become hydrophobic and water repellent. The severity of soil water repellence is higher under tree canopies and decreases in bare areas (Zavala *et al*., 2014). Diamantopoulos *et al.* (2013) found that water repellence enhances preferential flows in unsaturated zones. Then, water tends to percolate through macropores that develop around the root system and bypass upper soil horizons. Krishnaswamy *et al.* (2013) studied three ecosystems: evergreen forests, former evergreen forests converted into tree savannahs (i.e. degraded forest), and exotic Acacia plantations. The results of the three different water balance approaches suggest a consistent ranking where groundwater recharge is higher in forests followed by Acacia plantations and tree savannahs. Weingartner *et al.* (2003) showed that forests increase base flows, and the FAO (2008) notes that while forests consume water, they also improve infiltration and recharge rates.

In spite of scientific research conducted on the effects of forest plantations on the hydrological cycle, studies must be extended to climate and forest regimes beyond those examined in the literature (Birkinshaw *et al.*, 2011). Best *et al.* (2003) and Brown *et al.* (2005) also highlight a lack of information reported in the literature regarding effects of vegetation changes on seasonal yields and flow regimes. The majority of previous work has an emphasis on annual or mean annual water yields. This study focuses on understanding effects of the replacement of natural grasslands with *Eucalyptus* plantations in temperate zones with small topographical slopes and presenting interannual and intraseasonal variability in local rainfall (Diaz *et al*., 1998; Cazes-Boezio *et al.*, 2003). The effects of *Eucalyptus* plantations on evapotranspiration and groundwater recharge were monitored in two experimental watersheds located in western Uruguay. The aim of this paper is to quantify the effects of replacing natural grasslands with forest plantations on hydrological processes. This is performed through an analysis of computed annual and seasonal water balances, and particularly for years involving drought stress where effects on seasonal yields are important, and especially for downstream water users.

MATERIAL AND METHODS

Study area

The study area is located in western Uruguay in the department of Paysandú. Two experimental watersheds with different vegetation features were analysed. Don Tomas (hereinafter DT) is a 2.12-km^2 watershed with 56% of its surface covered with an adult plantation of *Eucalyptus Globulus ssp. Maidenni* planted in the fall and spring of 1998. During the study period, the Diameter at Breast Height increased from 0.14 to 0.16 m, the average height increased from 14.9 to 16.8 m, plantation density remained at 895 trees/ha and the IAF reached 2.81 by the end of the period. The remaining area includes firewall and riparian areas covered with natural pastures and thickets. The DT basin outlet is located at 32°14′55.5″ South Latitude and 57°38′48.7″ West Longitude. The reference watershed, La Cantera (hereinafter LC), is a 1.20-km^2 watershed covered with grasses (Baccharis *sp*., Bromus *sp*., Paspalum *sp*., and Stipas *sp*.). The basin outlet is situated at 32°16′37.8″ South Latitude and 57°36′14.3″ West Longitude. The distance between the centres of gravity of the experimental watersheds is roughly 5 km. It was not possible to find a closer catchment that has remained under pasture without being converted into a forest plantation for the whole study period. Nevertheless, this distance is acceptable, as Uruguay is characterized by minor topographic variations between its southern and northern areas, producing mild cumulative spatial rainfall variations across the whole country on a semi-annual and annual basis. In the study area, the mean historical annual rainfall is 1208 mm. Both main streams are located along the left bank of the Capilla Vieja stream as shown in Figure 1. The latter is a tributary of the Queguay River, which discharges into the Uruguay River.

The examined watersheds were selected as they present similar hydrological physical characteristics in regards to main stream lengths, basin slopes and time of concentration as shown in Table I. Geomorphological and soil characteristics in both watersheds are very similar, thus validating the selected methodology. The main soils found in the highest areas or headwaters are Pachic Argidol and Typical Hapludert; loam to deep sandy loam soils identified as Typical Argidol are found below the flattened surface between main streams; and dark, texturally varied soils classified as Typical Hapludalf are found in lower areas. Soil thicknesses vary from 0.80 to 1.20 m.

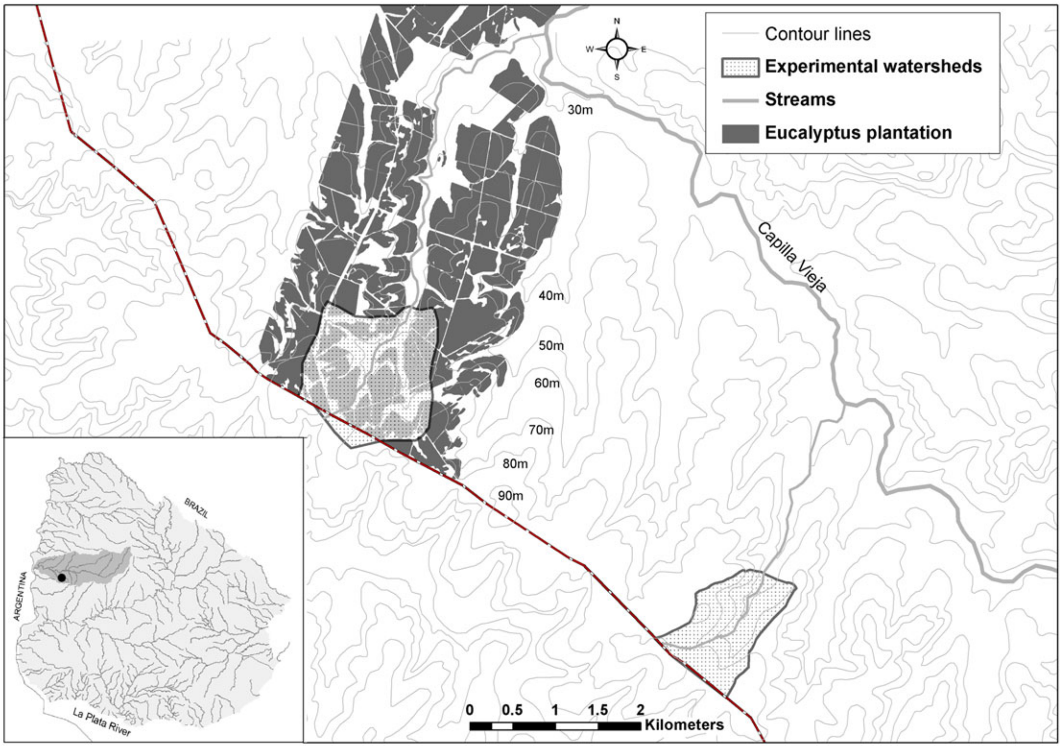

Figure 1. The Don Tomas and La Cantera watersheds

Table I. Physical characteristics of the studied watersheds.

Physical parameter	Don Tomas	La Cantera
Surface area (km^2)	2.12	1.20
Main stream length (m)	1783	2168
Main stream slope (%)	0.90	1.58
Basin slope (%)	4.68	5.92
Time of concentration (min)	39.0	36.5

Methodology

Seasonal (autumn and winter from April 1 to September 30 and spring and summer from October 1 to March 31 of the following year) and annual water budgets for the hydrological year (October 1 to September 30 of the following year) were calculated for the DT and LC watersheds.

The water budget equation for a given watershed (Best *et al.,* 2003) is:

$$P - AET - Q_s = \pm \Delta S \pm \Delta GW \quad (1)$$

where P is the incident rainfall, AET is the actual evapotranspiration, Q_s is the stream discharge at the watershed outlet, ΔS is the change in soil water storage and ΔGW is the change in groundwater storage. Actual evapotranspiration (AET) can be rewritten as:

$$AET = I + E + T \quad (2)$$

where I is the interception loss, E is soil evaporation and T is transpiration. Hereinafter, the sum of E and T is denoted as ET.

The interception loss from the tree canopy is computed as:

$$I = P - \left(P_d + P_f\right) \quad (3)$$

where P_d is the throughfall and P_f is the stemflow. Grass interception in the LC is assumed to be of the same interception order as that for firewall and riparian areas and for mulch under *Eucalyptus* areas.

To estimate recharge, the groundwater budget was also considered:

$$GW_{rech} - GW_{disch} = \pm \Delta GW \quad (4)$$

where GW_{rech} is groundwater recharge and GW_{disch} is groundwater discharge.

Changes in groundwater storage were estimated from water table fluctuations, and groundwater discharges were considered to be equal to the stream base flow.

Rainfall. Total incident rainfall was measured using Rain Wise recording tipping bucket rain gauges, with one located in an area free of trees in the DT watershed and with the second installed in the LC watershed. Both were positioned close to the centre of gravity of each watershed. The gauges were installed in accordance with World Meteorological Organization recommendations (WMO, 2008). Five-minute records were aggregated to compute the totals for each month and for each season and budget year (Tables II and III).

Interception losses. Rain falling over forest cover is redistributed into three components according to Equation (3). To quantify P_d and P_f, a redistribution plot was installed in an area located close to the DT watershed outlet.

Throughfall was recorded using the methodology applied by Iroumé and Huber (2000), which involves using a galvanized steel gutter with a triangular cross section of 0.15 m in width and 35 m in length (a catchment area of 5.25 m^2). It was installed in the direction of the catchment's natural slope and positioned diagonally between two rows of trees to take into account natural variations in the canopy. The collected throughfall was channelled into a 600-l tank in which water level variations were recorded by an OTT Thalimedes and by WT-HR TruTrack water level gauges. The tank was emptied through a small drain pump that was automatically operated by a level sensor and that was powered by a 12-V battery.

Ten trees positioned close to the galvanized steel gutter were selected and rubber collars were sealed around the tree trunks to measure the stemflow (Iroumé and Huber, 2000). These collars were connected to a 50 mm-diameter PVC pipe that channelled the stemflow to a main pipe below the gutter, which was connected to a second 600-l tank. The registration and emptying systems used were the same as those described for throughfall measurements.

Interception losses were computed as the difference between incident rainfall and the sum of throughfall and stemflow according to Equation (3).

Changes in soil water storage. Soil water content was quantified through the methodology based on thermalization of neutrons using a CPN 503DR hydroprobe. In the forested watershed, seven sites positioned close to the tree rows and seven sites between two tree rows were selected, whereas in the watershed covered with pastures, seven sites were selected. At each site, PVC access tubes were installed at depths of 15, 30, 50 and 70 cm below the soil surface. Measurements were carried on a monthly basis taking into account that soil water content is a state variable of interest only at the beginning and end of the evaluation period (Echeverría *et al.*, 2007; Huber *et al.*, 2010). Measurements were performed from October of 2006 to September of 2009.

Groundwater balance and recharge. Groundwater levels were measured at 10-min intervals using WT-HR TruTrack and OTT Thalimedes water level gauges positioned on three piezometers located in the upper, middle and lower slopes of the DT and LC watersheds. Data acquisition began in September of 2009.

Groundwater balance was estimated by combining two methods: hydrograph separation and water table fluctuation. The first method involves separating total streamflow into surface runoff and base flow and associating the latter with groundwater discharge (Risser *et al.*, 2005; Healy, 2010). The second method is based on the premise that variations in unconfined aquifers levels are because of groundwater recharge and discharge (Fan *et al.*, 2014).

Table II. Annual water budget (in mm) computed in the DT and LC watersheds in Uruguay, where P is the incident rainfall, I is the interception loss, ΔS is the change in soil water storage, ΔGW is the change in groundwater storage, GW_{rech} is groundwater recharge, GW_{disch} is groundwater discharge, ET is the sum of soil evaporation and transpiration and Q_s is the stream discharge at the watershed outlet. The actual evapotranspiration (AET) is the sum of ET and I.

	Don Tomas (DT) (Forests)									La Cantera (LC) (Natural Grasslands)						
Budget year	ET	P	I	Qs	ΔS	ΔGW	GWrech	GWdisch	AET	P	Qs	ΔS	ΔGW	GWrech	GWdisch	AET
Oct 06–Set 07	859	1545	296	355	35				1155	1508	679	65				764
Oct 07–Set 08	661	927	188	79	0				849	930	285	2				643
Oct 08–Set 09	640	792	162	7	−18				802	817	179	−6				643
Oct 09–Set 10	1253	2523	425	772		73	381	308	1678	2418	1158		15	329	314	1245
Oct 10–Set 11	551	1231	200	484		−5	197	202	751	1245	545		−3	223	226	702
Oct 11–Set 12	651	1445	213	549		32	305	273	864	1484	735		9	269	260	740
Oct 12–Set 13	702	1817	336	818		−38	363	401	1038	1767	916		53	353	300	797
Oct 13–Set 14	889	1601	320	444		−52	329	382	1209	1566	862		−4	264	268	707

Table III. The seasonal water budget (in mm) computed in the DT and LC watersheds in Uruguay, where P is the incident rainfall, I is the interception loss, ΔS is the change in soil water storage, ΔGW is the change in groundwater storage, GW_{rech} is groundwater recharge, GW_{disch} is groundwater discharge, ET is the sum of soil evaporation and transpiration and Q_s is the stream discharge at the watershed outlet. The actual evapotranspiration (AET) is the sum of ET and I.

	Don Tomas (DT) (Forests)									La Cantera (LC) (Natural Grasslands)						
Budget season	ET	P	I	Qs	ΔS	ΔGW	GWrech	GWdisch	AET	P	Qs	ΔS	ΔGW	GWrech	GWdisch	AET
Oct 06–Mar 07	695	1122	217	141	69				912	1100	300	95				705
Abr 07–Set 07	164	423	79	214	−34				243	408	379	−31				59
Oct 07–Mar 08	404	618	132	75	8				536	616	179	−38				474
Abr 08–Set 08	257	309	56	3.6	−8				313	315	106	40				168
Oct 08–Mar 09	406	516	106	7.3	−3				512	542	110	−4				436
Abr 09–Set 09	234	276	56	0	−14				290	275	70	−2				207
Oct 09–Mar10	960	1967	323	606		78	230	152	1283	1889	911		17	211	195	961
Abr 10–Set10	293	557	102	167		−6	150	156	395	529	247		−2	117	120	285
Oct 10–Mar11	226	437	86	193		−68	20	88	312	460	138		−39	41	80	362
Abr 11–Set11	325	793	114	291		63	177	114	439	784	407		37	182	146	341
Oct 11–Mar12	361	745	99	289		−4	160	164	460	784	309		−9	135	144	484
Abr12–Set12	290	700	114	260		36	145	109	404	700	426		18	134	116	257
Oct12–Mar13	506	1358	248	622		−18	189	207	754	1321	622		0	190	190	700
Abr13–Set13	195	460	88	196		−20	175	194	283	445	294		53	163	110	98
Oct13–Mar14	646	881	172	157		−94	98	192	818	849	380		−10	116	126	479
Abr14–Set14	243	721	148	287		42	231	189	392	716	482		6	148	142	228

For the hydrograph separation, a single parameter digital filter was considered (Eckhardt, 2005):

$$Q^i_{s,srf} = \alpha Q^{i-1}_{s,srf} + (1+\alpha)\left(Q^i_s - Q^{i-1}_s\right)/2 \qquad (5)$$

where $Q^i_{s,srf}$ is the surface runoff at time i, Q^i_s is the total streamflow at time i and α is the filter parameter.

Base flow $Q^i_{s,base}$ is determined by subtracting the surface runoff from the total streamflow:

$$Q^i_{s,base} = Q^i_s - Q^i_{s,srf}. \qquad (6)$$

The water-table fluctuation method assumes the following expression for groundwater recharge and discharge:

$$GW_{rech} = \begin{cases} \Delta H S_y & if \ \ \Delta H > 0 \\ 0 & if \ \ \Delta H \leq 0 \end{cases}$$
$$GW_{disch} = \begin{cases} 0 & if \ \ \Delta H > 0 \\ \Delta H S_y & if \ \ \Delta H \leq 0 \end{cases} \qquad (7)$$

where GW_{rech} and GW_{disch} are groundwater recharge and discharge, ΔH denotes changes in groundwater levels and S_y is the aquifer specific yield.

Base flow and groundwater discharge were calculated on a daily basis. The optimum value of the filter parameter α was determined through visual inspections of the hydrograph and of the resulting surface runoff and base flow. Specific yield S_y values were chosen so that seasonal average groundwater discharges were equal to base flows.

Stream discharge at the basin outlet. Stream discharge (Qs) was measured using a sharp-edged V notch weir installed at the outlet of each watershed, i.e. DT and LC. These weirs were designed for a maximum flow rate of $1\,m^3s^{-1}$ corresponding to events with return periods of less than 1.5 years. Water level fluctuations in time were recorded every 5 min using OTT Thalimedes water level gauges. Storm flows greater than $1\,m^3s^{-1}$ were estimated by measuring the level of water discharged through the lateral exceedance weir, and the stage–discharge curve was computed from the hydrodynamic model of the main stream. The public domain model HEC-RAS 4.0, developed by the Hydrologic Engineering Center (HEC) Corps of Engineers U.S. Army (USACE), was implemented for this purpose. Input data consisted of cross sections of main stream and water levels measured at the outlet of each watershed and a cross-section located upstream. Few events exceeded the weir crest in LC, showing agreement with the selected return period. The estimated volume for the greatest of these events was 0.03% of the corresponding monthly runoff volume. The weir in DT was exceeded twice over the study period.

External groundwater income. During particularly dry periods (e.g. January of 2008 to September of 2009), the stream discharge was intermittent in DT over periods of up to six months of zero runoff, while LC always showed permanent runoff. However, such headwaters in small watersheds with shallow soils are characterized by ephemeral streamflows. Therefore, permanent runoff in

LC may be because of differences in catchment hydrological responses or to external groundwater income.

Dry periods, for which piezometric data were available, were analysed and an external groundwater income was considered for LC. A scenario without external groundwater income was also analysed to evaluate the sensibility of the results to this hypothesis.

Data analysis. The database was analysed to detect data inconsistencies and to fill in missing data because of measurement equipment malfunctions. Daily and monthly data were plotted to detect inconsistencies or measurement errors (e.g. rainfall–rainfall and rainfall–streamflow), and differences between daily datasets were computed. Missing monthly rainfall data were filled by correlating LC and DT, as rainfall are not expected to change drastically over a distance of 5 km. For missing monthly stream discharge data, two procedures were used: correlating monthly rainfall and stream discharge and correlating LC and DT stream discharge. Missing interception data because of throughfall or stemflow measurement failures were filled using a previously calibrated interception model (Alonso, 2011) according to Gash *et al.'s* (1995) sparse canopy rainfall interception model.

RESULTS

Input data

Only nine non-simultaneous months with partially missing rainfall data were identified in DT and LC. For DT, 88 days with missing rainfall data were identified in the examined months, but 54 days were rainless according to surrounding rain gauges. The corresponding values for LC were 239 days with missing rainfall data and 147 rainless days. Missing monthly rainfall data were filled according to the satisfactory correlation found between LC and DT ($R^2 = 0.976$). Regarding monthly stream discharge 19 and 27 months with partially missing stream level records were identified in DT and LC, respectively. These gaps are mainly because of battery failures, as field visits were conducted approximately every 30 days because of limited financial budgets. Stream discharge data in LC were filled using the acceptable relationship found between monthly rainfall and stream discharge ($R^2 = 0.696$) whereas in DT the relationship with LC stream discharge ($R^2 = 0.824$) was used, as the correlation between monthly rainfall and stream discharge was found to be poor ($R^2 = 0.434$) because of stream discharge dispersion. This dispersion depends on the interrelationship between rainfall intensity, interception and streamflow patterns. Canopy interception is high when rain intensity is low, reaching up to 100% and decreasing to 10% with increasing rain intensity. Before filling in missing stream discharge data, values resulting from the regression were compared with partially measured data, and when the latter were greater, the measured data were used.

Soil water content data only covered the first three budget years because of hydroprobe failures. Measured changes in soil water storages for the two experimental watersheds ranged from 2.2 to 95.4 mm seasonally and from 0.2 to 64.6 mm annually. These maximum measured values represent 8.7% of the corresponding seasonal rainfall and 4.3% of the corresponding annual rainfall. Changes in groundwater storage were measured from October of 2009, when recording level gauges were installed at each piezometer.

Water budget

The hydrological water budget was computed on a seasonal and annual basis according to Equations (1) to (4). The results are given in Tables II and III.

Some hydrological indicators and relationships were defined to interpret water budgets. Specific discharge changes because of the substitution of natural grasslands with forest plantations were computed as

$$SDR = \left(\frac{Q_p}{P_p} - \frac{Q_f}{P_f}\right) \times 100(\%) \qquad (8)$$

where SDR is the percentage decline in specific discharge; Q_p and P_p denote seasonal or annual stream discharge and rainfall measured in LC (natural pastures), respectively; and Q_f and P_f are measured in DT (forests).

SDR values were represented as a function of annual and seasonal rainfall, showing a decreasing trend as rainfall increases (Figures 2 and 3). These values were compared with interception losses because of canopies in the same figure. However, the coefficients of determination (R^2) for the correlation between rainfall and SDR and rainfall and interception losses are relatively low. This is mainly attributed to the fact that the aggregated rainfall does not fully explain interseasonal or interannual variations in these hydrological indicators. Water yield decline also depends on the temporal distribution of rainfall, meteorological conditions (i.e. cloudiness, solar radiation and temperature) and soil moisture content. Tables II and III show that particularly wet and dry years have dominated over the years that historically have been considered as mean years (1208-mm annual rainfall) with unusually rainy springs and summers and dry autumns and winters. The hydrological years between October of 2006 and September of 2014 can be classified as four wet years, two dry years and two mean years according to a neighbouring non-recording rain gauge with daily rainfall

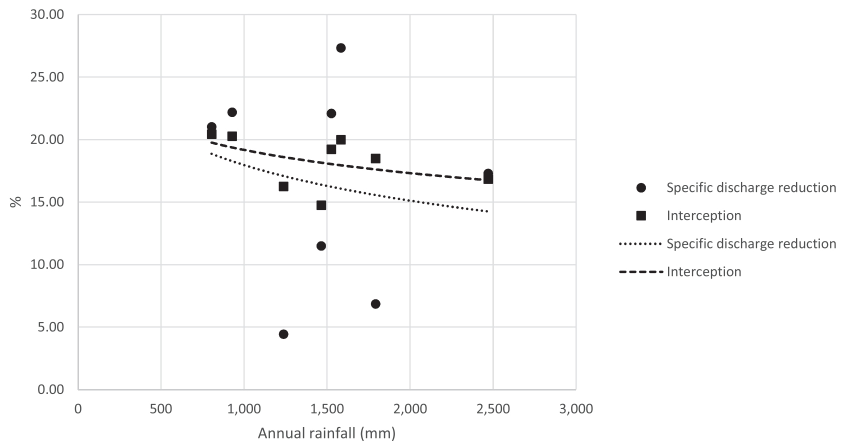

Figure 2. The relationship between annual rainfall and specific discharge reduction (*SDR*) and interception (*I*) computed for two experimental watersheds in western Uruguay

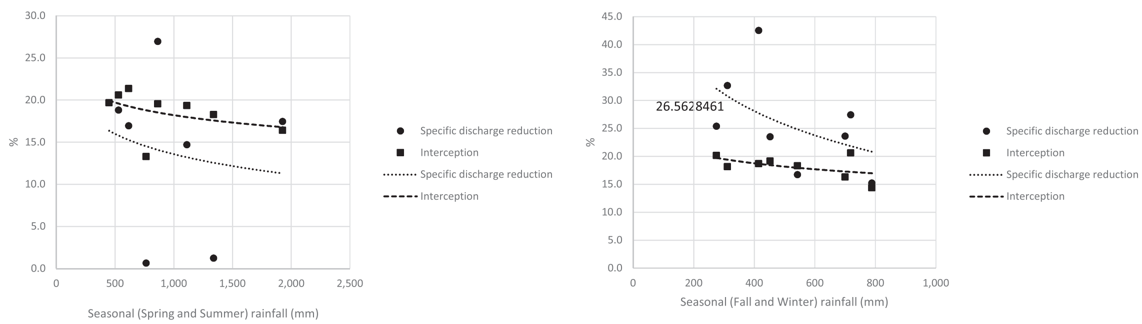

Figure 3. The relationship between seasonal rainfall and specific discharge reduction (*SDR*) and interception (*I*) computed for two experimental watersheds in western Uruguay. Spring and summer data are shown on the left and fall and winter data are shown on the right

data series from 1950 to 2014. The degree of annual specific discharge decline ranges from 17 to 27% of the corresponding annual rainfall, although not for three years during which values of between 4 and 11.5% are likely because of unidentified measurement equipment failures. These percentages have not been considered in drawing conclusions. The specific discharge decline corresponding to mean annual rainfall is 17.2%.

Figure 3 shows that specific discharge decline reached roughly 16% of P for dry springs and summers with a rainfall of approximately 500 mm. However, for dry autumns and winters with rainfall ranging from 275 to 400 mm, the degree of specific discharge decline was greater (32 to 28% of P), representing a high percentage of the limited rainwater available.

The difference in *AET* was also computed as a percentage of annual and seasonal rainfall according to equation (9). This hydrological indicator explains the relationship between rainfall, land use and *AET* (Zhang *et al.*, 1999, 2001).

$$\Delta AET = \left(\frac{AET_f}{P_f} - \frac{AET_p}{P_p}\right) \times 100\ (\%) \tag{9}$$

where ΔAET is the percentage difference in AET; AET_f and P_f denote seasonal and annual AET and rainfall measured in DT (forests), respectively; and AET_p and P_p are measured for LC (natural pastures).

Figure 4 presents the computed annual and seasonal ΔAET as a function of corresponding rainfall. The computed annual ΔAET ranges from 4 to 32% of corresponding annual rainfall with an average value of 18%. However, we found differences of up to 46% in seasonal ΔAET between forests and pastures for seasons with low accumulated rainfall, mainly because of low rainfall intensities (e.g. less than 1 mm/h) and high canopy interception (e.g. up to 100%), which is in agreement with the results shown in Figure 3. As seasonal and annual rainfall increases, the ΔAET difference between forest and grass as a percentage of the corresponding accumulated rainfall decreases mainly as a result of high rainfall intensity and throughfall.

Figure 5 reproduces scatter plots developed by Zhang *et al.* (1999, 2001) which represent AET for forested and grassed catchments as a function of annual rainfall based on data collected from 250 catchment yield experiments for 29 countries. The plot shown in Figure 5 also provides

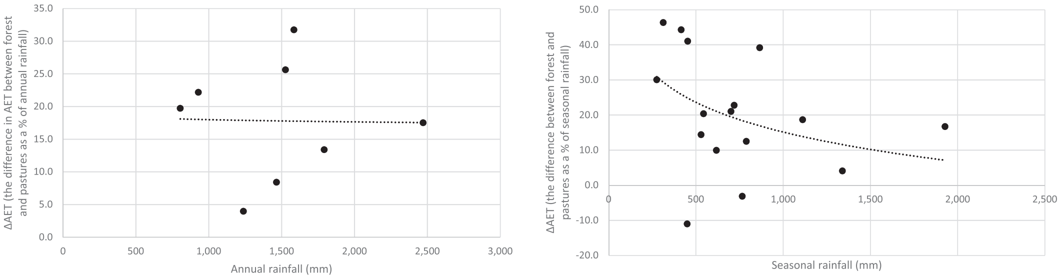

Figure 4. The relationship between rainfall and ΔAET (the difference in actual evapotranspiration between forest and natural pastures) computed for two experimental watersheds in western Uruguay. Annual data are shown on the left and seasonal data are shown on the right

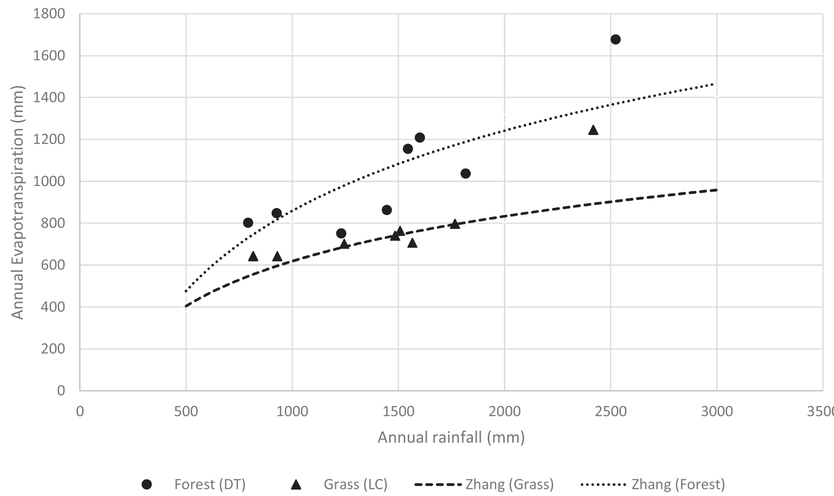

Figure 5. Annual evapotranspiration computed from water balances in DT (forest plantation) and LC (natural grasses) compared to the relationships presented by Zhang *et al*. (1999, 2001)

the AET computed from water balances for LC and presents DT acceptable agreement though not for the budget year with the highest rainfall.

Groundwater balance and recharge

The driest period for which piezometric data were available, presenting several zero stream discharge days for DT and permanent stream discharge for LC, occurred between 22 January and 21 July 2010 (181 days). During this period, 137 days presented a 0-mm stream discharge value for DT, 113 of which generated a minimum stream discharge value of 0.39 mm in LC. Therefore, an external groundwater income of 0.39 mm/day (142 mm/year) was assumed for LC.

Changes in groundwater storage, recharge and discharge were calculated daily according to Equations (4) and (7) for both watersheds.

The digital filtering approach shown in Equation (5) was first applied to separate hydrographs recorded at the outlets of both watersheds daily. The optimum value of filtering parameter α was determined for each watershed through a visual inspection of the hydrograph and resulting base flow (see Figure 6 and Table IV).

Next, the aquifer discharge was calculated from water table records using Equation (7). The piezometer located in the lower slope of the DT watershed registered an almost continuous data series, with only one season presenting partial data (24% missing data). No data from the other two piezometers in DT were considered, as the water table fell below the piezometer depth for much of the study period.

The piezometer located in the lower slope of the LC watershed registered an almost continuous data series, with only one season presenting incomplete data (24% missing data). The piezometer located in the middle slope recorded six seasons without missing data. A single specific yield value was considered for both LC piezometers. Recharge and discharge were individually calculated for each piezometer using equation (7). Recharge and discharge levels of both piezometers were correlated for the common data period (six

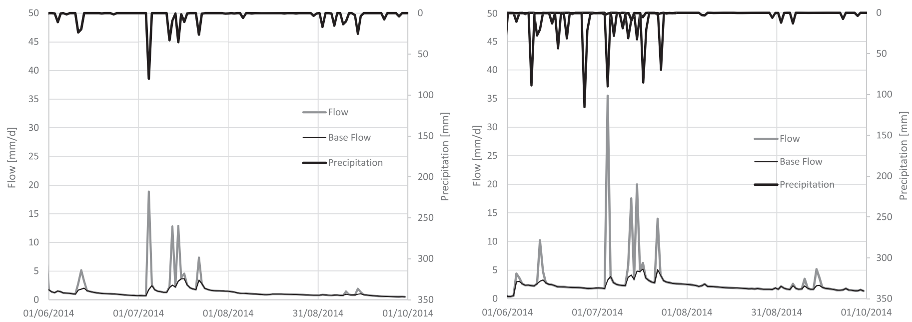

Figure 6. Portion of the hydrograph separated via single parameter digital filtering. DT (forest plantation) and LC (natural pasture) values are shown on the left and right, respectively

Table IV. Digital filter parameter value for watershed hydrograph separation, specific yields and seasonal average base flow and discharge.

	Don Tomas (DT) (Forests)	La Cantera (LC) (Natural grasslands)
Parameter for digital filtering	0.93	0.95
Average base flow [mm/season]	157	137
Specific yield	0.065	0.039
Average aquifer discharge [mm/season]	157	137

seasons), and this correlation was used to estimate values for the missing seasons. The average results calculated from the two piezometers data were associated to LC watershed.

The specific yield value was calibrated for each watershed so the average aquifer discharge matched the average base flow (see Figure 7 and Table IV). The external groundwater income was subtracted from the average base flow in LC.

As shown in Tables II and III, groundwater exchange flows from DT (recharge and discharge) are slightly higher than those from LC. On average, DT recharge reaches 315 mm/year (18.3% of incident rainfall), while that for LC is 288 mm/year (17.0% of annual rainfall).

DISCUSSION

Water budget

Length of the data series and seasonality. The results obtained from computing the seasonal water budget are based on a short series dominated by anomalous wet springs and summers and by relatively dry autumns and winters. Therefore, they are opposite to those found through a previous research based on 34 year-long series (Silveira and Alonso, 2009). This work, which is based on rainfall-runoff data collected before and after afforestation, shows that annual streamflow has decreased because of forest plantations in a large basin with a surface area of 2097 km^2 located in northeastern Uruguay.

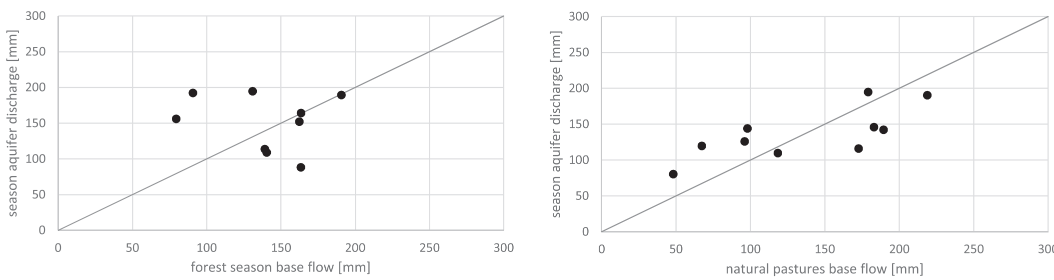

Figure 7. Seasonal base flow and aquifer discharge. Specific yield values were calibrated to obtain the same average value for base flow and aquifer discharge. DT (forest plantation) and LC (natural pasture) values are shown on the left and right, respectively

The area converted into *Eucalyptus* and *Pinus* plantations covers roughly 30% of the basin surface. The streamflow has decreased by between 36.5 and 8.2%, corresponding to annual rainfall between 900 and 1700 mm, respectively. The present study also shows that runoff decline is more pronounced during the spring and summer (October-March) (diminishing by between 38.4 and 25.2%) and is lower during the autumn and winter (decreasing by between 20.3 and 15%) depending on seasonal rainfall totals. Nevertheless, the SDR percentages are of the same order of magnitude as those found for the experimental DT and LC watersheds. This suggests that SDR is correlated with available rainwater rather than with climatological seasons.

Stream discharge. Rodriguez Suarez *et al.* (2014) assessed the effects of *Eucalyptus globulus* afforestation in a small catchment of Galicia (NW Spain). The mean annual rainfall here is similar to that of western Uruguay, although the monthly distribution is characterized by dry and temperate summers. Streamflow decline accounted for 22% of total stream discharge. This was related to an increase in rainfall interception during wet periods and to an increase in evapotranspiration during dry periods. Bosch and Hewlett (1982), in their review of paired catchment studies focused on temperate zones, concluded that Eucalyptus plantations cause a roughly 40-mm change in annual water yields with each 10% change in vegetation cover. These reported magnitude values are in agreement with our study results, where 56% of the catchment surface was afforested and where the reduced specific discharge in mean hydrological years was of the same magnitude as the interception (17.2%) and was higher (28 to 32%) during drier years because of increased evapotranspiration.

AET. Peel *et al.* (2002) note that actual values of *AET* for each land use or vegetation type are heavily dependent on local climatic and physiographic conditions. Their research paper shows ΔAET values for forest (evergreen and deciduous areas combined) and grass areas plotted against mean annual rainfall based on relationships developed by Zhang *et al.* (1999, 2001). The value of ΔAET corresponding to our regional mean annual rainfall of 1208 mm is roughly 22%, whereas for our experimental watersheds, the equivalent value is 18%. Nevertheless, the plot developed by Peel *et al.* (2002) shows ΔAET values of between 5 and 21% for mean annual rainfall below 1000 mm (dry years), whereas our findings show an ΔAET value of close to 18% for mean annual rainfall between 800 and 1000 mm. Low rainfall intensity combined with climatic patterns and the presence of an open aquifer close to the ground surface may explain the higher percentages found in our experimental watersheds.

Lima (2011) also found that the relationship between AET and mean annual rainfall observed by Zhang *et al.* (1999, 2001) is consistent with data collected from experimental catchment areas of the Catchment Area Monitoring Programme (PROMAB) of the Forestry Science and Research Institute (IPEF) of Brazil.

Interception. The average canopy interception loss for 8- to 16-year-old *E. globulus* plantations in western Uruguay was recorded as 18.3% of rainfall. For southern Australia, Benyon and Doody (2015) measured an interception loss of 19% (±4.9) of annual rainfall for *E. globulus* plantations, with closed canopies receiving 505–771 mm of rainfall. Huber *et al.* (2010) reported 10 and 11% interception for two 9-year-old Eucalyptus globulus catchments located in the Coastal Range of southern central Chile. The measured period spanned 14 months, where the total accumulated rainfall reached 2149 mm characterized by a rainfall season running from May to September. For another 9-year-old Eucalyptus spp. plantation, Lane *et al.* (2004) found intercepts of 15 to 27% of annual rainfall (roughly 1800 mm). Therefore, the interception value found in this study is similar to the reported value for the same forest type, and differences that resulted are likely because of other forest characteristics (age and density), climatic patterns, or rainfall temporal distribution and intensity.

Groundwater recharge. Afforestation effects on groundwater found through one case study are difficult to generalize to other locations. Features such as rainfall regimes, soil types, geological conditions, water table depths and alternative land uses affect groundwater balance components and can vary significantly between locations. While some researchers report a reduction in groundwater recharge because of afforestation (Fan *et al.*, 2014), others report the opposite response (Krishnaswamy *et al.*, 2013) and others project a mixed response that varies over time (Wyatt *et al.*, 2015). In this study, groundwater exchanges flows (recharge and discharge) calculated from the afforested DT basin were similar to those found for the pastured LC basin. On average, DT recharge reached 315 mm/year (18.3% of incident rainfall) and those for LC reached 288 mm/year (17.0% of annual rainfall). The recharge values computed for DT and LC fall within the recharge range presented in the literature: 11 to 52% of rainfall for grasslands and 4 to 39% for afforested *Eucalyptus* areas (Fan *et al.*, 2014, Le Maitre *et al.*, 1999, Allison and Hughes, 1972). Therefore, local factors mitigate the effects of grassland conversion to forests on groundwater recharge.

Uncertainty in water budget components

As stated above, data from soil water content were measured during the first three budget years. However, measurements collected for these years suggest that errors were introduced by not considering changes in soil water content by a maximum of approximately 4% of the corresponding annual rainfall, increasing the AET by approximately 10%. These values are in agreement with those described by Zhang *et al.* (1999), which suggest that changes in soil water storage often only account for 5 to 10% of annual rainfall.

As noted above, external groundwater income was assumed for LC. If such income had not been considered, a specific yield value of 0.076 would have to be considered to equalize base flow values with the estimated recharge value (Table V). This would have produced an average recharge value of 587 mm/year (34.6% of incident average rainfall), which seems overestimated, and especially when compared to recharge values (between 10% and 24% of incident average rainfall) obtained for the Guarani Aquifer in an outcropping area located 150 km away from the study area (Collazo, 2006; Gómez, 2007).

The above points illustrate the influence of external groundwater income on water budgets, and on resulting recharge in particular: any change in the external groundwater income value will directly affect the recharge value. Therefore, to reduce uncertainty, more efforts must focus on characterizing external groundwater income by integrating more piezometric measures and geochemical data. Another parameter that, according to the presented methodology, affects recharge is the specific yield. The inclusion of specific yield estimators based on laboratory or field measures will also reduce water balance uncertainties.

Table V. Annual water budget (in mm) computed in the LC watershed without considering external groundwater inflows, where P is the incident rainfall, ΔGW is the change in groundwater storage, GW_{rech} is groundwater recharge, GW_{disch} is groundwater discharge, AET is the actual evapotranspiration and Q_s is the specific discharge at the watershed outlet.

	La Cantera (LC) (Natural grasslands)					
Budget year	P	Qs	ΔGW	GW_{rech}	GW_{disch}	AET
Oct 09–Set 10	2418	1301	30	671	641	1088
Oct 10–Set 11	1245	687	−5	456	461	563
Oct 11–Set 12	1484	877	18	549	531	589
Oct 12–Set 13	1767	1058	109	721	612	600
Oct 13–Set 14	1566	1005	−7	540	547	568

CONCLUSIONS

This study highlights seasonal and annual water budgets measured in two experimental watersheds located in western Uruguay over an eight-year period (October of 2006 – September of 2014).

The main conclusions of the present study are as follows:

- In mean hydrological years, the reduced specific discharge (17.2%) occurring in forested watersheds is of the same magnitude as the interception of mean annual rainfall.
- The reduced specific discharge roughly doubles (28 to 32%) in seasons and years characterized by low rainwater availability (275 to 400 mm). Depending on water uses downstream, this is of relevance for decision makers responsible for water resources management.
- The difference in AET between forest and natural pastures reaches up to 50% during drought seasons where rainfall are lower than 500 mm, and this value tends to decrease to less than 10% during wet seasons with rainfall exceeding the mean annual rainfall (1208 mm).
- Groundwater flow exchanges (recharge and discharge) are similar in both watersheds (roughly 17 to 18% of incident annual rainfall). Thus, local factors such as climate, soil and geological conditions prevent effects on groundwater recharge resulting from the conversion of natural grasslands to forests.

The annual water budget is consistent and shows acceptable agreement with relationships presented by Zhang *et al.* (1999) on the basis of 250 catchment-scale measurements collected from 29 countries worldwide. Nevertheless, the seasonal water budget based on 8 years characterized by anomalous wet springs and summers and dry autumns and winters shows that the specific discharge reduction was higher during the autumn and winter and lower during the spring and summer. These results are opposite to those of a previous study based on a 34-year-long data series showing that the specific decline in discharge is higher during the spring and summer and lower during the autumn and winter. This highlights the important effects of climate variability on stream flow discharge and provides evidence of the need to implement long-term monitoring programmes that are representative of intraseasonal and interannual climatic variability.

The results of this study suggest that differences in groundwater recharge between similar basins with different ecosystems may be the result of a combination of factors such as climatic patterns, rain intensity, basin slopes, soil types, soil tillage patterns used prior to tree planting, root depths and plantation densities. These hydrologic responses provide some support for the hypothesis provided by Krishnaswamy *et al.* (2013), according to which differences in infiltration between land cover rather than evapotranspiration determine groundwater recharge variations.

Therefore, more research efforts should involve obtaining reliable measures of groundwater flows, soil properties and percolation patterns to better understand the hydrological effects of vegetation cover changes on shallow aquifer systems.

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