Hydrologic Impacts of Afforestation in Uruguay, A 20-Year Paired Watershed Study

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Highlights

- A 20-year paired watershed study clearly showed that afforestation of a grassland watershed reduced water yield.
- Cumulative water yield over the 17-year tree growth period was reduced by 35%.
- Reductions of water yield were not evident until the fourth year after planting.
- Flow rates were reduced by 40% to 50% for high flows (less than 20% of the time) and by 50% to 70% for the remaining low flows.

Abstract. *Substantial portions of grassland are being converted to managed forest in Uruguay. Long-term paired watershed studies in other continents indicate that water yields from managed forest lands are reduced compared to grasslands. Few such studies have been conducted in South America. This paired watershed study was initiated in 2000 to determine the hydrologic impacts of changing land use from grassland to pine plantation in Uruguay. Outflow rates, water table depths, rainfall, and meteorological variables were continuously measured for a 20-year period on two adjacent watersheds, LC1 (69 ha) and LC2 (108 ha), at La Corona estancia located in the Tacuarembó River basin in northern Uruguay. During the three-year pretreatment period, both watersheds remained in pasture, the traditional land use in the region. In July 2003, one watershed (LC2) was planted with loblolly pine (Pinus taeda L.), while the other (LC1) remained in pasture. Afforestation decreased cumulative water yield over the 17-year period from planting, July 2003, to June 2020 by 35%. Annual water yields from LC2 were reduced, with the magnitude of reduction dependent on weather patterns and stage of tree growth. Differences in water yields between LC1 and LC2 were significant ($p < 0.01$) in the last twelve years of the study compared to the first 4 years. Changes in water yield were not detected until four years after planting. Analysis of daily flow duration data indicated that flow rates after planting on LC2 were reduced between 40% and 50% for high flows occurring less than 20% of the time and reduced by 50% to 70% for the remaining low flows. Our results are consistent with other research in areas with similar rainfall. Water table and cumulative outflow data suggest that the water yield reduction is associated with deeper-rooted pine having access to more water in the soil profile for transpiration, as compared to shallow-rooted grasses. Research continues on this site to quantify the hydrologic impacts of afforestation through the production cycle for pine in Uruguay.*

Keywords. *Cumulative outflow, ET_o, Flow duration curves, Forest hydrology, Loblolly pine, Water table, Water yield.*

Uruguay is located in a humid subtropical to temperate climate in eastern South America between 30° and 35° South latitudes. Most of the country is within the Campos grassland, a savannah of plains and rolling hills that covers an area of 500,000 km^2 in Uruguay, northeast Argentina, southern Brazil, and southern Paraguay (Pallares et al., 2005). Elevations within the Campos in Uruguay range from sea level to 500 m above mean sea level (AMSL). About 85% of Uruguay's land mass (175,020 km^2) was in agriculture in 2000, the highest percentage of any country in the world (World Bank, 2024). Historically, most of the native grasslands were used for livestock grazing while row crop farming was conducted on some of the more productive soils.

The Uruguayan government instituted financial incentives in the mid-1980s for conversion of some of the less productive pasturelands to forest plantations in an effort to diversify the rural economy. Approximately 4 million ha were designated for afforestation (Uruguay XXI, 2020). In response, national and multinational corporations purchased land and planted trees (primarily eucalyptus (*Eucalyptus globulus*), loblolly pine (*Pinus taeda* L.), and slash

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pine (*Pinus elliottii*)) over significant portions of the landscape. By 2023 about 1.22 million ha, or 30% of the land designated for afforestation, was planted (MapBiomas Uruguay, 2023). This scale of land use conversion raised concern among Uruguayan stakeholders and environmental groups. Issues of concern include negative impacts of widespread afforestation on social structure, biodiversity, soil quality, and water quality and quantity. The greatest concerns are about reductions in water yield and downstream water supply (Paruelo, 2012; Vihervaara et al., 2012; Wang and Fu, 2013).

Numerous watershed studies evaluating hydrologic impacts of afforestation and deforestation have been conducted in the United States of America (USA), Australia, New Zealand, South Africa, China, South America, and Europe. Literature reviews of studies conducted before 2000 clearly indicated that, other factors remaining equal, water yield from the landscapes with established trees is less than from landscapes with shorter vegetation (Hibbert, 1965 [39 studies]; Bosch and Hewlett, 1982 [94 studies]; Sahin and Hall, 1996 [145 studies]; Andréassian, 2004 [137 studies]; Brown et al., 2005 [166 studies]; Farley et al., 2005 [27 studies, only afforested]). The majority of the studies considered in these reviews were based on paired watershed experiments to determine the impact of tree harvesting. More recent reviews have also indicated that water yield from established forests is less than from watersheds with shorter vegetation (Filoso et al., 2017; Neary, 2016; Zhang et al., 2017). More variability in the responses was reported in reviews that included studies using methods other than paired watershed experiments. Most literature reviews stress that hydrological impacts (e.g., water yield) of changes in forest cover are variable, depending on spatial factors (climatology, geology, and topography) and on temporal factors (weather patterns and vegetation growth). Consequently, reviews stress the need for long-term experiments in many different locations. Most reviews place greatest value on paired watershed studies that are calibrated before treatments and that take measurements over long periods of time (Amatya et al., 2021; Andréassian, 2004; Bosch and Hewlett, 1982; Neary, 2016).

Only 18 of the 312 studies reported in a systematic review by Filoso et al. (2017) were located in South America, and just two of those were conducted in pampas/campos grassland regions. Published results from recent watershed studies conducted in the pampas/campos region (Almeida et al., 2016; Silveira et al., 2016; Smethurst et al., 2015) are consistent with findings of earlier studies. None of these relatively short-term studies (2 to 8 years) were calibrated before the afforestation treatment was applied but do report valuable information about different aspects of the hydrologic cycle. Alonso et al. (2024) used national gauging stations on watersheds with significant afforestation as well as flow data from experimental studies to conduct a data-based analysis of afforestation on water yield in Uruguay. They estimated that water yield reductions ranged from 21% to 32% in large basins and from 39% to 50% in small basins.

Modeling studies have been conducted using the data set presented herein. Von Stackelberg et al. (2007) calibrated and validated the Soil and Water Assessment Tool (SWAT) using the first four years of the data set presented herein to predict the yield reduction that would occur when pine trees were planted on the grassland watershed. They determined that the SWAT model predicted outflows from the two watersheds during the pretreatment period with reasonable accuracy (model efficiency greater than 0.71). Using the calibrated model with crop parameters for mature pine forest, von Stackelberg et al. (2007) predicted afforestation would reduce average annual outflows over a 33-year period by 23% to 30%. Nervi et al. (2025) evaluated multi-step calibration methods incorporating the long-term flow observations reported herein with satellite-based evapotranspiration (ET) and leaf area index (LAI) data to calibrate and verify the SWAT model. They found that multi-step calibration reduced uncertainty and improved flow predictions of SWAT during the verification period. For various afforestation scenarios, the best calibration methods predicted that afforestation would reduce mean annual outflows by 9% to 23%.

This paper presents the results of a 20-year paired watershed study to evaluate the timing, drivers, and long-term hydrologic impacts of land use conversion from grassland to pine plantation in the campos region of Uruguay. Results of this study quantify the effects of afforestation on (1) changes in outflows or water yield, (2) when those changes are first detected and how they vary with stage of growth and management practices, and (3) how those changes affect soil water distributions as indicated by changes in water table depths with time.

Materials and Methods

Site Description

Two small, adjacent watersheds (69 and 108 ha in size) were selected for study in the Tacuarembó River basin (fig. 1). The watersheds are located on La Corona (LC) estancia (31° 38’10” S, 55° 41’ 30” W) of the El Cerro tract owned and managed by Weyerhaeuser Uruguay (2000–2017) and Lumin (2017–2023). Both watersheds remained in the traditional land use of grazed pasture through the three-year pre-treatment period (July 2000 through June 2003). Pine seedlings were planted on the treatment watershed (LC2, 108 ha) in July 2003. The control watershed (LC1, 69 ha) remained in pasture with livestock grazing for the duration of the 20-year study.

Topography of the watersheds is a rolling landscape with protruding rocky hillocks of basalt and sandstone. Elevations vary from 130 to 204 m AMSL on LC1 and from 136 to 192 m AMSL on LC2 (fig. 1). The northern portion of watershed LC2 has a plateau and cliff area at higher elevations, with a similar, but smaller, feature in the western portion of watershed LC1. Land slopes mostly ranged from 2% to 15%, except in the cliff areas. The aspect of watershed LC1 is primarily to the east; watershed LC2 faces south and east.

An extensive natural network of incised channels conveys surface and subsurface flows from the landscape to the outlets of each watershed. Slopes of the stream channels range between 4% and 10% in the tributaries in the upper elevations of the watersheds and between 1% and 1.5% in the main channels at lower elevations. Soils in the lower and

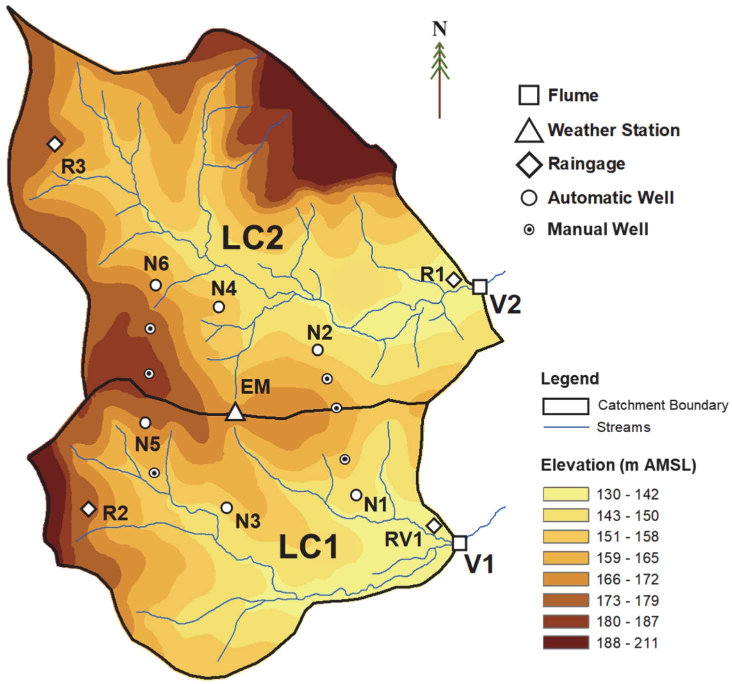

Figure 1. Topography, hydrography, and location of instrumentation with rain gauges, weather station, groundwater wells and flumes in the control (LC1; 69 ha) and treatment (LC2; 108 ha) watersheds (31° 38'10" S, 55° 41'30" W).

middle elevations are mostly sandy loam and sandy clay loam, ranging in depth from 0.8 to 1.7 m over sandstone. The higher elevations are outcroppings of basalt and sandstone overlain by a shallow topsoil layer ranging in depth from 0.10 to 0.35 m. The LC2 watershed has a greater proportion (27%) of shallow soils compared to LC1 (8%).

Pre-Treatment and Treatment Implementation

The two watersheds were managed as grassland with livestock grazing during the three-year pretreatment period (1 July 2000 through 30 June 2003). Grazing density (estimated by herd managers) was 0.9 cattle units per hectare, with one cattle unit defined as the foraging needs of one cow of 380 kg weight with calf.

The treatment watershed (LC2) was planted with loblolly pine seedlings in July 2003 (fig. 2), while the control watershed (LC1) remained in grassland with livestock grazing. In keeping with standard forestry practices, riparian corridors, equipment access lanes, and very steep areas were not planted, resulting in afforestation of 57% of the LC2 surface area. The trees were planted 2.5 m apart in furrows (approx. 10 cm deep and 70 cm wide). Planting density was 1,000 trees per ha, per the standard planting practices of Weyerhaeuser Uruguay. The area between furrows remained in grass vegetation, and the furrows were 4 m apart, aligned perpendicular to the hillslopes.

Trees were thinned and pruned in June and July 2008, five years after planting. Thinning reduced tree density from 1000 trees per ha to 650 trees per ha. Pruning removed branches to a height of 3 meters, reducing the live crown of each remaining tree by 50%. Additional pruning occurred on the best 350 trees in June 2011 and June 2013, removing branches to a height of 4.5 meters in 2011 and 6 meters in 2013. Biomass removed during pruning and thinning was left in place under the trees. The most significant change in biomass occurred in June 2017 (14 years after planting) when commercial thinning (harvesting) reduced tree density from 650 to 300 trees per ha.

Livestock management followed standard practice for the treatment watershed. Livestock were removed from LC2 prior to planting and were not returned during the period of establishment and early growth. After pruning and thinning in year 8 (age 5, 2008), livestock were returned to the watershed and allowed to graze on the understory.

Field Measurements

We instrumented the watersheds to measure precipitation, outflow rates, weather parameters, and water table elevations. Instrumentation (figs. 1 and 3) included a weather station, automatic rain gauge, four manual rain gauges, six water table elevation recorders, and stage recorders for flow measurement at the two outlet flumes. We continuously collected data from 1 July 2000 to date. The period of record analyzed and presented herein is the 20-year period from 1 July 2000 through 30 June 2020. Accordingly, year 1 of the study refers to the period from 1 July 2000 through 30 June 2001, year 2 to 1 July 2001 through 30 June 2002., etc. Results of the study reported here include the three-year pretreatment period (years 1-3) followed by the 17-year treatment period (years 4–20).

We installed a 3-meter-tall Campbell Scientific weather station equipped with automatic sensors and a CR10X datalogger over a grassy surface on the ridge between the two watersheds (fig. 1). Rainfall, air temperature, relative humidity, wind speed, wind direction, solar radiation, and net radiation were measured on a 30-second interval, summed or averaged and stored on a 15-minute basis for analysis. In addition to the weather station, rainfall was also continuously

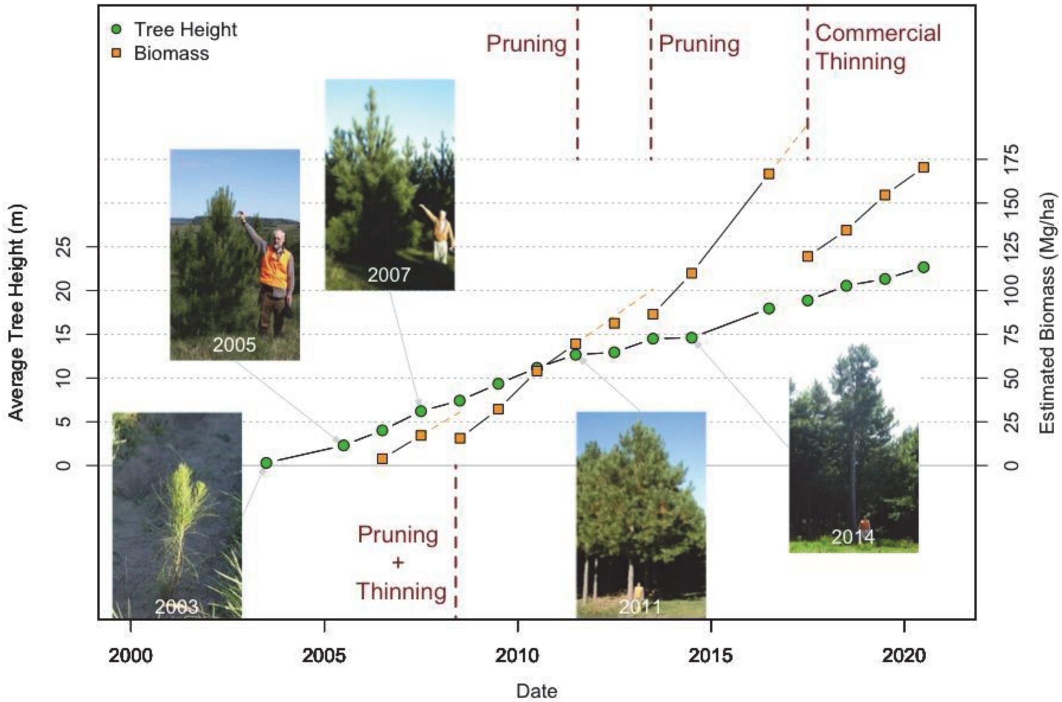

Figure 2. Average pine tree height and estimated biomass from planting in 2003 to 2020 on the treatment watershed LC2 (vertical dark brown dashed lines on plot at the top represent the timing of associated management; sloped dashed gold lines correspond to hypothesized biomass growth just before pruning and thinning events).

measured and recorded with a tipping bucket gauge (backed up by a manual gauge) located at R1 near the flume outlet of the watershed LC2 (fig. 1). We installed four additional manual gauges (RV1, R2, R3, and R4; fig. 1) to estimate the variability of rainfall. Rain gauge R4 (not shown) is located at the ranch house just south of watershed LC1.

Variability of precipitation across the two watersheds, as indicated by differences in measurements by the six rain gauges (2 recording and 4 manual), was found to be negligible. Rainfall data from gauge (R1) were used for our analyses because the instantaneous data better described rainfall intensity. Missing data from gauge R1 were supplemented using data from the weather station (EM). We used meteorological data collected at the weather station with the Penman-Monteith equations to estimate daily reference evapotranspiration (ET_o) for grass (Allen et al., 1998).

Flow rates at the outlet of the two watersheds were measured using HL flumes, which are 1.37 m in height (fig. 3). We designed concrete flumes with stainless steel measuring sections using the guidelines provided by Brakensiek et al. (1979). Stage at the flume entry was measured at 3-minute intervals with a Stevens Type F recorder with a float and weight system until September 2002, when an ISCO 720 flow probe was installed; thereafter, stage was recorded every 2 minutes. A rating equation (Gwinn and Parsons, 1976) best fit the calibration tests and was used to calculate flow rates through the flume outlet from measured flow stages. If stage elevations exceeded 1.37 m, flow rates were calculated assuming a broad-crested rectangular weir superimposed on the top of the HL flume. Stages greater than 1.37 m did not occur after April 2004, when emergency spillways with broad-crested weirs and separate stage recorders were installed adjacent to the HL-flumes to accommodate excess flows and more accurately measure high flow rates during rare large storm events.

The water table is perched in the soil profile above underlying bedrock (sandstone, basalt) during wet periods. Depth to water table documents elevation of the saturated (or nearly saturated) zone and provides an indication of soil water conditions in the profile. Shallow water table depths were measured and recorded hourly in wells (~2 m deep) installed throughout the watersheds (N1, N3, and N5 on LC1 and N2, N4, and N6 on LC2, fig. 1).

Calibration and Data Analysis

The classical approach in paired-watershed studies is to establish, during a pre-treatment calibration period, a relationship between flows of the treatment watershed with those of the control watershed (e.g., Andréassian, 2004; Loftis et al., 2001; Zégre et al., 2010). This relationship is then used, with measured flow at the control watershed as index data, to calculate the expected flows that would have been measured on the treatment watershed had the treatment not occurred. The net treatment effect is calculated as the difference between the measured and expected flows on the treatment watershed.

Regression methods are typically used for developing relationships between the treatment and control watersheds during the calibration period; however, regression methods can create statistical challenges due to concerns about normality, heteroscedasticity, and autocorrelation. Methods used to cope with these statistical constraints include transformation of the variables and addition of terms to account for seasonality and autocorrelation (Watson et al., 2001). Statistical constraints can be investigated by developing relationships for data intervals over different time steps (i.e., annual,

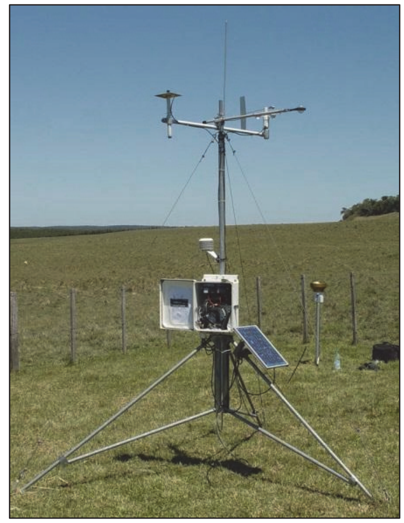

Figure 3. Left: flume at the outlet of LC1, the control watershed, which remained in pasture throughout the study. Right: the weather station, located as shown in figure 1.

semi-annual, quarterly, monthly, and daily). Data analyzed over longer time steps tend to conform better to statistical constraints, but at a cost of statistical strength due to fewer data points in the analysis.

Our calibration relationships were based on the regression equation proposed by Watson et al. (2001) and used in various other paired watershed studies. A general form of this equation can be written as:

$$F\big(y(t)\big) = a_1 + a_2 F\big(x(t)\big) + a_3 \sin\left(\frac{2\pi t}{T_{yr}}\right) + a_4 \cos\left(\frac{2\pi t}{T_{yr}}\right) + a_5\ AR \quad (1)$$

where

F() = transformation of x and y variables (i.e. log(x) or x^n)

y(t) = predicted outflow from the treatment watershed within time step, *t* (i.e. daily or monthly)

x(t) = measured outflow from the control watershed within time step, *t* (i.e. daily or monthly)

T_{yr} = number of time steps in a year

AR = auto regression function

a_1 = intercept

$a_{(2\text{-}5)}$ = coefficients for the variables (reported in eqs. 2 and 3 below).

We performed regression analyses over a range of transformations and time steps to determine those that met the statistical constraints. Statistical constraints were tested using the Shapiro-Wilk test for normality, the Breusch-Pagan and White test for heteroskedasticity, and the Durbin-Watson test for autocorrelation. We then selected the regression relationship that best fit the data, met the statistical constraints, and had the shortest time step. The selected regression was used to predict the net effects of the treatment for each time step of the treatment period and to determine if the predictions were outside of the 95% prediction limits of the regression. The selected regression was also used to calculate the yearly net effect of the treatment as a percentage of the expected outflow and to determine the net effect of the treatment over the entire treatment period.

The regression relationship for the daily time step that best fit the data was used to predict daily flow that would have occurred if the treatment watershed had remained in pasture. These daily predictions were used to develop various cumulative plots to visualize the timing of the changes in outflow during the treatment period, to predict Flow Duration Curves (FDC) that would be expected had the treatment watershed remained in pasture, and to calculate water balances over various time periods. While daily relationships rarely meet statistical constraints required to set predictive limits, daily flows estimated by regression relationships have been found to be very reasonable estimates of expected flows in other studies (Bren and Lane, 2014), particularly if they are consistent with relationships determined for longer time steps.

Metrics for Detection and Quantification of the Impacts of Afforestation

To detect and quantify the impacts of afforestation, we developed metrics for flow rates, flow volumes, and a new cumulative water table depth indicator. We then determined how the impacts varied between pre- and post-treatment and between control versus treatment watersheds. The first method compiled the measured and predicted monthly outflows into yearly sums. The effects of afforestation were estimated by the differences expressed in mm and percentage difference between the expected and measured *annual* flow volumes. The second method divided the study period into five 4-year periods and used one-way analysis of variance (ANOVA) to determine if the flow differences between LC1 and LC2 changed between each 4-year period. The ANOVA was performed on quarterly flow data using the Welch procedure (Welch, 1951) with pairwise comparisons made by the Games and Howell test (Games and Howell, 1976). The third method identified when the measured monthly flows at LC2 fell outside of the 95% prediction limits of the pre-treatment relationship described above. We think this method is well-suited for detection purposes although it might not capture when changes became enduring. The fourth method established a time series of daily cumulative measured and expected flow volumes and created double mass curves to visualize when changes in flow occurred and the durability of those changes. A fifth method sorted the daily measured and expected flow volumes from highest to lowest to create flow duration curves (Searcy, 1959) for time periods before and after afforestation. The flow duration curves were used to assess the impact of afforestation over the range of flow rates observed on the watersheds. The sixth method summed daily water table depths to obtain a time series of cumulative water table depths. Averages of the daily water table values at wells N1 and N3 in watershed LC1 and at wells N2 and N4 in LC2 (fig. 1) were calculated and cumulated. These locations represented relatively large areas of similar soils and elevations on the two watersheds. A time series of the difference between these cumulative average depths was developed and plotted as a function of cumulative flow volume at the control watershed as a corollary indicator to detect the impact of afforestation. The seventh (last) method used a water balance to estimate actual ET to help identify the drivers for the impact. This method used water table records to identify periods when soil water storage was approximately the same at the beginning and end. We assumed that the change in soil water storage was zero; thus, ET = precipitation - outflow for these selected periods. Time periods were selected when the average water table depths (N1 and N3 for LC1; N2 and N4 for LC2) were both receding, at the same depths, and relatively shallow (between 0.4 and 1.1 m) to assure that the distributions of soil moisture above the water table were similar at the beginning and end of the period.

RESULTS

PRECIPITATION AND REFERENCE EVAPOTRANSPIRATION (ET_O)

Average annual precipitation measured on the site during the 20-year study period was 1599 mm (table 1), which is 14% greater than the 1400 mm average recorded during the previous 21-year period (1979-1999) at the INIA Tacuarembó weather station (15 km southwest of the study site). Average annual precipitation recorded at the INIA station during the study period (1575 mm) was within 2% of that measured at the study site, indicating that precipitation in the region during the study period was above the historical average. All precipitation occurred as rainfall with year-to-year variability much greater in years 1–10 (Coefficient of Variation (CV) = 0.39) compared to years 11–20 (CV = 0.17) (table 1). Very wet conditions occurred during the three pretreatment years of the study when both sites were in pasture. Average annual precipitation (2225 mm) during pretreatment was 39% greater than the study period mean. Relatively dry conditions occurred during the six years (years 4–9)

Table 1. Annual (July-June) rainfall and ET_o observed for each year of the study. Measured outflows for the Pasture site (LC1) and Pine site (LC2) are also shown along with expected values for LC2 (LC2Ex) had it remained in pasture (calculated by relationships developed during the calibration period (eq. 2) and summing the expected monthly values for each year). The pretreatment period (bold) ended in June 2003. Trees were planted on LC2 in early July 2003. The calibration period (blue) ended in June 2004. Values in the gray shaded rows represent mean values for the five 4-years periods of the study.

Year of Study	Date	Annual Rain (mm)	Annual ET_o (mm)	Annual Flow (mm) Measured LC1	Measured LC2	Expected LC2EX	Annual Reduction (mm)		Percent Reduction (%)
		1	2	3	4	5	3 − 4	5 − 4	(5 - 4)/5
1	**Jul 00-Jun 01**	**1843**	**1384**	**1021**	**1123**	**1090**	**-102**	**-33**	**-3**
2	**Jul 01-Jun 02**	**2225**	**1255**	**1300**	**1315**	**1361**	**-15**	**46**	**3**
3	**Jul 02-Jun 03**	**2608**	**1269**	**1584**	**1671**	**1626**	**-87**	**-45**	**-3**
4	Jul 03-Jun 04	1049	1392	239	272	299	-33	26	9
5	Jul 04-Jun 05	1392	1466	503	545	570	-41	25	4
6	Jul 05-Jun 06	978	1411	246	281	295	-35	14	5
7	Jul 06-Jun 07	1414	1293	397	357	469	40	112	24
8	Jul 07-Jun 08	980	1391	229	217	283	13	66	23
9	Jul 08-Jun 09	993	1480	135	130	187	5	58	31
10	Jul 09-Jun 10	2172	1194	1015	821	1050	195	230	22
11	Jul 10-Jun 11	1143	1400	237	189	302	48	113	37
12	Jul 11-Jun 12	1303	1373	308	203	376	106	173	46
13	Jul 12-Jun 13	1700	1246	632	329	706	303	377	53
14	Jul 13-Jun 14	1768	1225	690	387	761	303	374	49
15	Jul 14-Jun 15	1503	1240	475	287	541	188	253	47
16	Jul 15-Jun 16	2035	1075	931	568	996	363	427	43
17	Jul 16-Jun 17	1799	1166	716	484	794	232	310	39
18	Jul 17-Jun 18	1470	1345	666	478	719	188	241	33
19	Jul 18-Jun 19	1851	1085	752	479	828	273	350	42
20	Jul 19-Jun 20	1762	1304	691	477	753	214	275	37
1 - 4	Jul 00-Jun 04	1931	1324	1036	1095	1094	-59 a[a]	-1	0
5 - 8	Jul 04-Jun 08	1191	1390	344	350	404	-6 a	54	13
9 - 12	Jul 08-Jun 12	1403	1362	424	335	479	88 b	143	30
13 - 16	Jul 12-Jun 16	1751	1197	682	393	751	289 c	358	48
17 - 20	Jul 16-Jun 20	1721	1225	706	479	773	227 c	294	38
Treatment period mean		1489	1299	521	383	584	139	202	35
Treatment period CV		0.26	0.10	0.51	0.46	0.46	0.97	0.69	0.45
Entire period mean		1599	1300	638	531	700	108	170	24
Entire period CV		0.29	0.09	0.61	0.76	0.55	1.35	0.88	0.77

[a] Values with different letters are significantly different at less than the 0.05 level as determined by ANOVA of the quarterly differences between LC1 and LC2.

following planting in July 2003, with average annual precipitation (1134 mm) 29% below the mean for the study period. Annual precipitation in the tenth year of the study (2172 mm) was 36% above the study mean. There was less year-to-year variation in the last ten years of the study when the average annual precipitation (1647 mm) was 2% above the study period mean.

Annual ET_o (grass reference) was much less variable (CV = 0.09) than was rainfall (CV = 0.29) during the period of study (table 1). Average annual ET_o for the 20-year period of observation at the study site was 1300 mm, which was only 3% greater than the past 20-year average annual corrected pan ET (1262 mm – pan coefficient 0.75) collected from the INIA weather station in Tacuarembó (von Stackelberg, 2005; von Stackelberg et al., 2007). The 3-year average annual ET_o for the pretreatment period (1303 mm) was close to the 17-year average annual ET_o for the treatment period (1299 mm).

On a long-term basis, the difference between rainfall and ET represents the amount of water available for outflow from the watershed, assuming negligible storage and seepage losses. Average monthly rainfall and ET_o are shown in figure 4 for the study period (2000–2020). It is well understood that actual ET depends on the vegetation and soil water conditions and may be either greater or less than ET_o, which is generally defined as the ET rate that would occur for well-watered short grass. ET from forested lands would generally be greater than ET_o for wet periods (Amatya and Harrison, 2016; Sun et al., 2010) but less than ET_o for dry conditions. Annual rainfall exceeded ET_o in all three years of the pretreatment period and in 10 of the 17 years of the treatment period (table 1). On average, annual rainfall (1489 mm) exceeded annual ET_o (1299 mm) for the 17-year treatment period. During the 20-year study period (2000–2020), average monthly rainfall exceeded ET_o from mid-autumn to early spring (April-October), while average monthly ET_o exceeded rainfall from mid-spring through early summer (November-March) (fig. 4).

Outflow Relationships During the Calibration Period

Since rainfall during the pretreatment period was well above average for the study period and rainfall during year four was well below, we included year four in the calibration period. This made rainfall during the calibration period more representative of that which occurred in the treatment period. Other afforestation studies have shown that the effect of trees on water yield in the first year after planting is negligible and this first year has been included as part of the calibration period in some studies (Bren and Lane, 2014).

Water yield from LC2 was 5.7% greater than that from LC1 during the calibration period. Close examination of the calibration results indicates that outflow characteristics of the two watersheds are fundamentally different at low flows. Differences in outflow between the watersheds were greater during low flow periods (fig. 5b), while outflows during high flow periods were very similar (fig. 5a). There is also a positive non-zero intercept (eqs. 2 and 3, fig. 5b) indicating that flows occur from the treatment watershed when flows from the control (LC1) are zero. This clearly shows the ever-presence of base flow in the treatment watershed, LC2, while the control watershed LC1, had periods of zero outflows in some years. Some differences in flows would be expected due to differences in watershed characteristics such as topography, geology, and soil depth and properties. Differences in watershed characteristics would lead to differences in the water balance components, namely flow, seepage, or ET. We observed that the flow differences between LC1 and LC2 were greater during wet periods. We therefore assumed that most of the differences in flow were due to lateral and/or deep seepage, which may occur in both watersheds but is apparently greater from the control, LC1, than from the treatment watershed, LC2.

A calibration equation on monthly data that transformed x and y by ½ power captured the observed flow patterns well. The transformation relationship also met the statistical constraints for normality and homoscedasticity. Adding a term

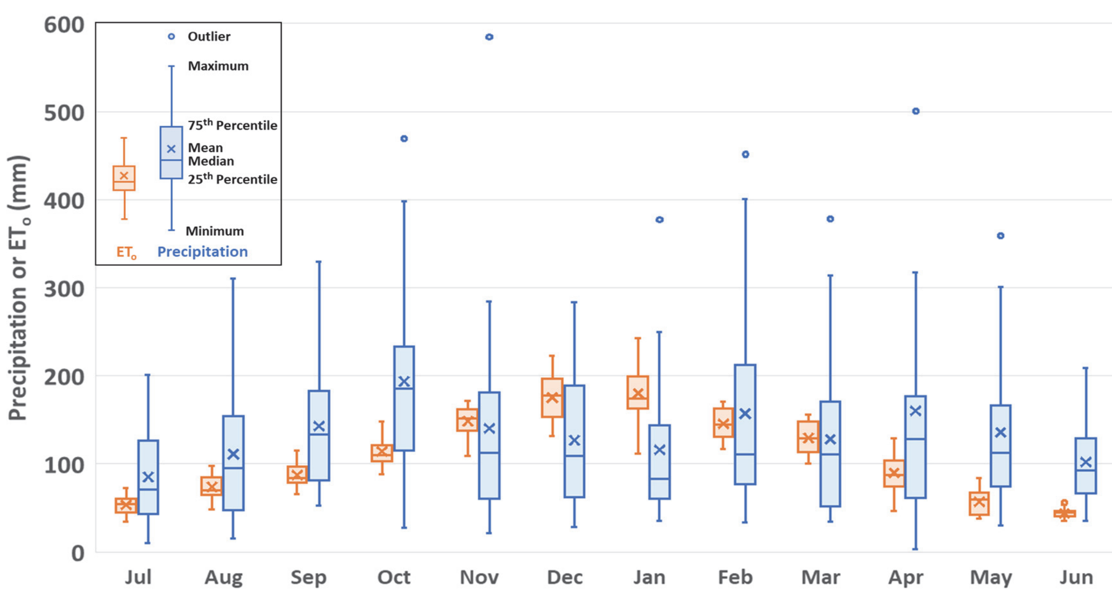

Figure 4. Distribution, as a box plot, of monthly rainfall and ET_o (grass reference) as determined from measured data at the weather station site during the study period (2000–2020).

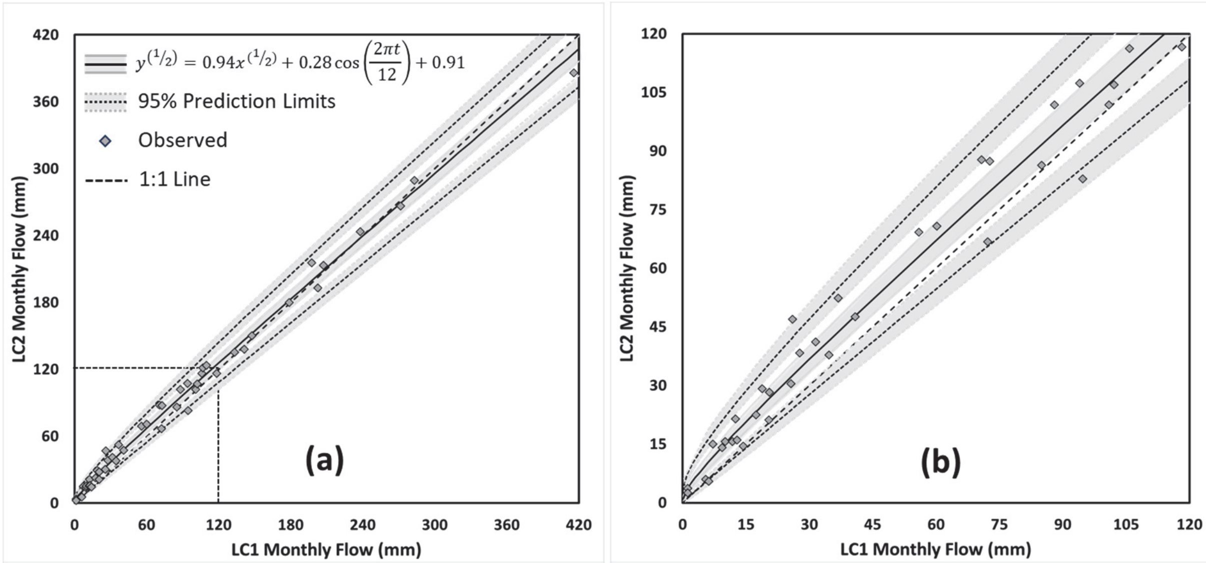

Figure 5. Relationship between monthly flows from watersheds LC2 and LC1 during the calibration period (2000-04). The regression model with 95% prediction limits is shown in x-y dimensions with gray areas showing the ranges defined by the seasonal (cosine) variable. The 1:1 line is also shown. Figure 5b shows details for the low flow range where monthly flows at LC2 are greater than at LC1.

for seasonality allowed the equation to meet the autocorrelation constraint. Only the cosine term was needed to account for seasonality since the data set began during winter. The resulting calibration for the monthly time step is:

$$Q_{LC2}^{\frac{1}{2}} = 0.94Q_{LC1}^{\frac{1}{2}} + 0.28\cos\left(\frac{2\pi t}{12}\right) + 0.91 \tag{2}$$

where
Q_{LC2} = Monthly flow from LC2 (mm)
Q_{LC1} = Monthly flow from LC1 (mm)
t = Month of year.

We used equation 2 with measured monthly flow (Q_{LC1}) from LC1 for every month during the treatment period to predict monthly outflow expected (Q_{LC2EX}) from LC2 had it remained in pasture. With the predictions of expected flow for each month of the treatment period, we could determine those months when the measured outflow fell outside of the 95% prediction limits of the expected flow. The monthly outflows were also summed over the year to calculate annual expected outflow and percent change for each year and over the treatment period.

The daily data set showed that flows from the treatment watershed were greater than those from the control watershed during the calibration period. The transformation used in the monthly calibration equation was also used in the daily calibration equation and likewise captured the higher flow rates from the treatment watershed during low flow periods. The relationship did not meet the constraints for normality, homoscedasticity, or autocorrelation. While addition of terms for seasonality and autocorrelation improved the relationships, statistical constraints for normality, homoscedasticity, or autocorrelation could not be met.

The calibration equation for the daily data was used to predict the expected daily outflow from the treatment watershed had it remained in pasture. These daily predictions were used to quantify changes during the treatment period using cumulative indicators and to create flow duration curves that would be expected for the treatment watershed had it remained in pasture. The calibration equation for the daily time step is:

$$Q_{LC2}^{\frac{1}{2}} = 0.94Q_{LC1}^{\frac{1}{2}} + 0.043\cos\left(\frac{2\pi t}{365}\right) + 0.0042AR + 0.91 \tag{3}$$

where
Q_{LC2} = Daily flow from LC2 (mm)
Q_{LC1} = Daily flow from LC1 (mm)
t = day of year
AR = auto regression function.

Effect of Afforestation on Water Yield

Water Yield Reduction Observed from Annual Indicators

Annual water yield from the afforested watershed (LC2) was lower than what would have been expected if it had remained in pasture (LC2EX) during the treatment period. Annual water yields from LC2 during the first three years after planting (years 4–6) were only 14 to 26 mm less than from LC2EX, with percent reductions ranging from 4% to 9%. Annual indicators show a notable decrease in water yield from the treatment watershed starting in year 7, four years after the planting of pine seedlings (table 1). Reductions in annual outflow from year 7 onward were greater than 57 mm and 20%. Percent reductions in annual outflow from LC2 peaked at 53% (377 mm) in Year 13, ten years after planting. Percent reductions in annual outflow from LC2 then declined to 39% (310 mm) in year 17. In Year 18, the first year after commercial thinning in June 2017, percent reduction (33%, 241 mm) was the lowest observed since Year 10 (22%, 230 mm). Percent reductions in annual outflow from LC2 in the final two years were greater than in year 18, 42%

(350 mm) and 37% (275 mm) for years 19 and 20, respectively. For the entire 17-year treatment period, mean annual measured outflows from the treatment watershed were 202 mm or 35% below expected outflows (table 1).

Some of the variability in the annual flow reduction values in table 1 is due to differences in soil water storages at the beginning and end of each year. The mean values over 4-year periods reduce the variability and give a closer approximation of average flow reductions, but still do not remove variability associated with differences in soil water storages at the beginning and end of the 4-year periods.

With the study period grouped into five 4-year periods, a one-way ANOVA (Welch, 1951) showed that there were significant differences ($p < 0.001$) between the means of the water yields from LC1 and LC2 (table 1). Differences in mean water yields for the first period after calibration were not statistically different ($p = 0.88$) from those for the calibration period but were different from those of the last three periods ($p = 0.003$, $p < 0.001$, and $p < 0.0001$ for the second, third, and fourth periods, respectively). Differences in mean water yields for the last two periods were not different from each other ($p = 0.52$) but were different from those of the calibration period ($p < 0.001$) and the second period ($p < 0.01$).

Water Yield Reduction Computed at the Monthly Scale

Water yield reductions at the monthly scale are presented as differences between measured flows at the treatment watershed (LC2) and flows predicted by equation 2 for LC2EX (fig. 6). Flow differences are compared to the 95% prediction limits to determine when reductions in outflow from LC2 for each month were significantly different (fig. 6). It is tempting to use the prediction interval method alone as a powerful tool to communicate to stakeholders on the timing of the beginning of afforestation impacts. Using the 95% prediction interval alone, one could conclude that the effect might have started in year 5, where measured flow was outside of the prediction interval. While the monthly flow difference was outside the prediction intervals once in year 5, monthly outflows were less than expected in only 5 of the 12 months. The monthly breakdown reveals that in year 7 the measured monthly outflows from the treatment watershed (LC2) were less than expected for all months of the year, and that there were five months when the differences were outside the 95% regression prediction intervals (fig. 6). After year 7, occurrences of monthly flow differences outside the prediction intervals increased to 11 per year in year 13 and remained at that level until year 20, with the exception of year 18, which followed a commercial thinning in the last month of year 17.

Yield Reduction Computed from Daily Values

The classic double mass curve (DMC) of daily flows (fig. 7A) shows that cumulative outflow from the treatment watershed, Q_{LC2} (rainbow curve), begins to deviate in year 7 from the expected cumulative flow had the watershed remained in pasture (Q_{LC2EX}), with deviation increasing with time. An increasing difference is shown between the cumulative expected flows from the treatment watershed (Q_{LC2EX}) and cumulative flows from the control (Q_{LC1}), as indicated by the 1:1 line in figure 7A. Had the two watersheds been identical, the two outflow relationships, Q_{LC2EX} and Q_{LC1}, would have been the same, at least for the calibration period. They clearly are not. As discussed previously, we assume this difference to be due to deep and/or lateral seepage from the control watershed, LC1. An analysis based on long-term water balances (not shown) indicated that the relative seepage from LC1 varied from 0.06 to 0.25 mm/d with an

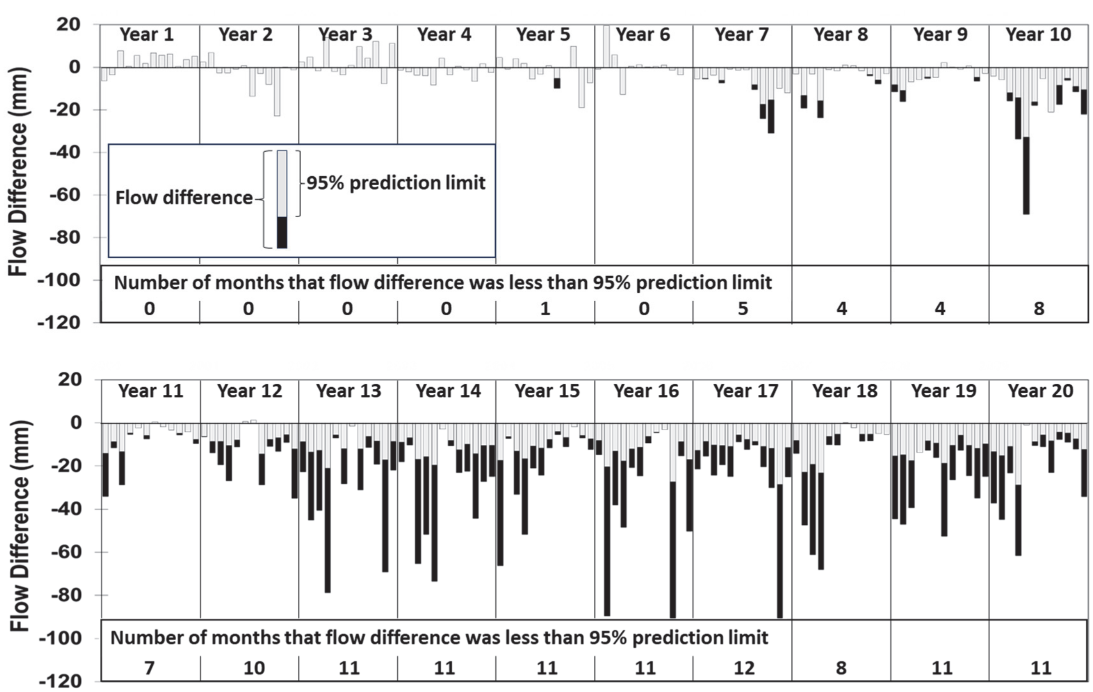

Figure 6. Monthly deviations of measured outflows from expected outflows for the treatment watershed (LC2) during the pretreatment and treatment periods. Expected outflows and 95% prediction limits were calculated using the regression relationship for monthly values shown in equation 2. The beginning month of each year is July.

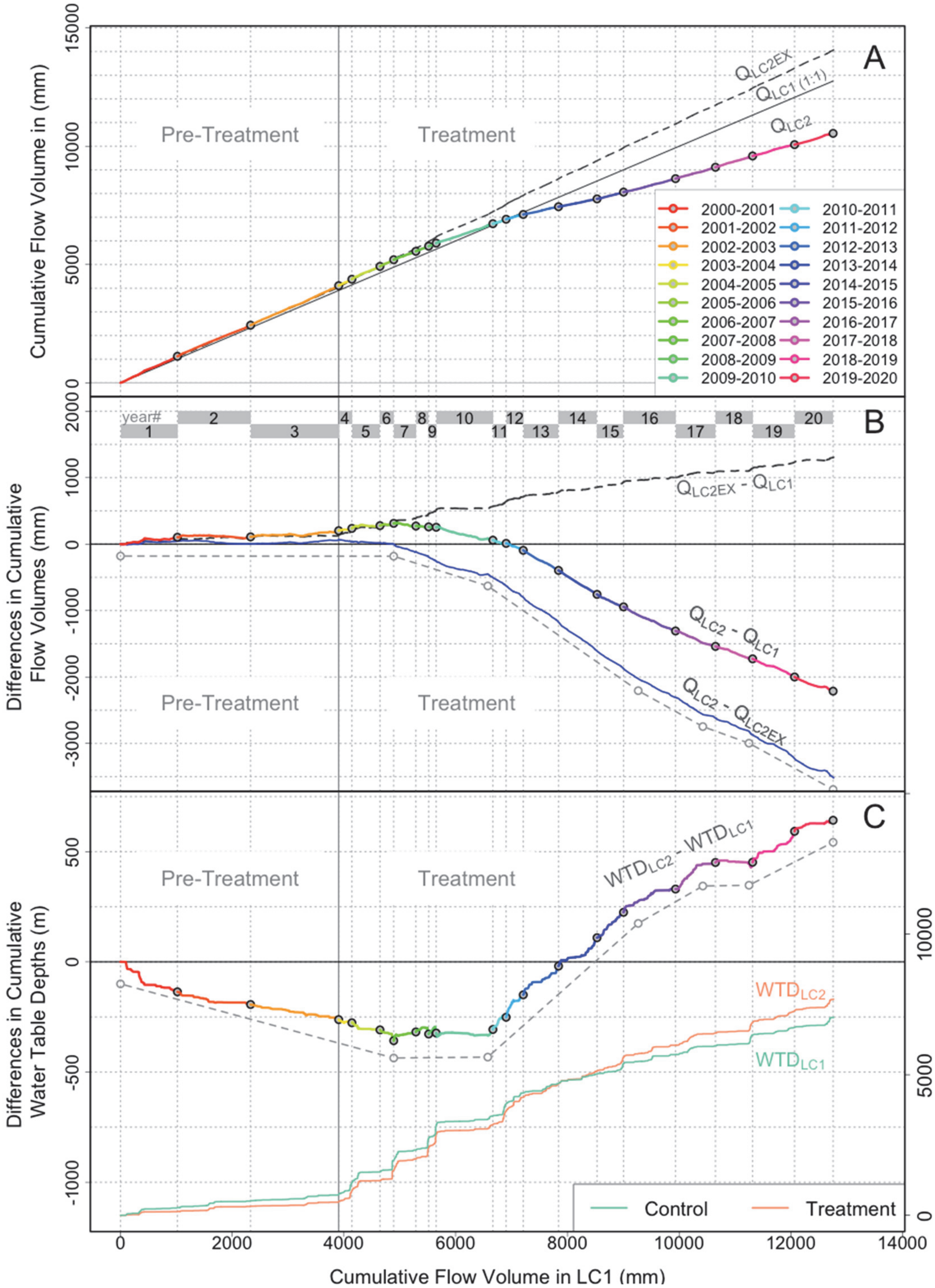

Figure 7. (A) Double mass curve of cumulative flow from LC2 versus cumulative flow from LC1. (B) Time series as a function of the cumulative flow volume at the control watershed, of the deviation between expected flow volumes at LC2 minus those at LC1 (broken black line), of the deviation between measured flow volumes at LC2 minus those at LC1 (rainbow-colored line with black dots marking the years), and of the calculated net treatment effect (blue line) accompanied by a parallel trend line with observed breaks (dashed gray line with points). (C) Time series as a function of the cumulative flow volume at the control watershed, of the deviation between the measured cumulative water table depths at LC2 minus those at LC1 (rainbow-colored line) accompanied by a parallel trend line with observed breaks (dashed gray line with points).

average of 0.18 mm/d over the course of the 20-year study. The plot of Q_{LC2EX}-Q_{LC1} in figure 7B shows the cumulative impact of seepage. The rainbow curve (Q_{LC2}-Q_{LC1}) shows the impact of afforestation that would have been assessed if the watersheds had not been paired and subjected to the calibration period prior to treatment; that is, if seepage had been ignored. That approach could have concluded that no impact was visible until year 10 or 11 and that afforestation resulted in a calculated flow reduction of 2300 mm or 22.9% of the expected outflow from the treatment watershed had it remained in pasture over the 17-year treatment period. A more accurate estimate of the actual effect is indicated by the blue curve in figure 7B (Q_{LC2}-Q_{LC2EX}), which considers the impact of seepage and shows that the net effect of afforestation reduced water yield by 3,547 mm or 35% of the expected 10,056 mm water yield (since planting), had LC2 remained in pasture over the 17-year treatment period.

The blue line (Q_{LC2}-Q_{LC2EX}) and accompanying trend line in figure 7B suggest that the overall 35% reduction of water yield occurred in several phases after planting. A first phase, years 4 through 6, shows very little or no apparent effect (flat blue line). A second phase, years 7 through 10, shows an apparent net effect where the average reduction was 23% of the expected cumulative flow over that period. A third phase from year 11 to the first quarter of flow volume in year 16 shows that the average net reduction increased to 47% of the expected cumulative flow over that period. Results for the fourth phase, between the end of phase 3 in year 16 and the end of data collection in year 20, show an overall net reduction of 38% within which one can observe a period from the end of year 17 through most of year 18 where the net reduction decreased to 33%. This temporary decrease is in synchrony with the commercial thinning that occurred during the last month (June) of year 17 when over half the trees were removed (fig. 2). The fourth phase starts one year before the thinning occurred and corresponds to a 5-year series that comprised nearly 40% of the after-planting outflow. The lower net reduction (38% vs. 47%) between phases four and three may thus be related to these relatively wetter and drier conditions, respectively, during these two phases. A mass balance to specifically address the impact of thinning is presented below. The definitions of the phases and trends are somewhat subjective but do correspond to apparent breaks in the blue curve in figure 7B.

Effect of Afforestation as Indicated by Outflows, Water Table Depth, and Stage of Growth

Effects of rainfall and ET_o on cumulative outflow and soil water conditions (represented by depth to the perched water table) are shown for the two land uses in figure 8 for years 5–8 when the pines were young (0%–12% of mature tree biomass; fig. 2) and for years 16–19 when pines were mature. The beginning of the hydrologic year is arbitrarily defined here as 1 July, which is near the start of winter (21 June in Uruguay), and coincides with the beginning of data collection, 1 July 2000, and of tree planting, 1 July 2003.

Rainfall was usually in excess of ET_o (see blue bars, fig. 8) in autumn through early spring (April-Oct.), which increased soil moisture and raised the perched water table to within 0.5 m of the soil surface during the first 6 months of most years. As the amount of water removed from the profile by ET increased to exceed cumulative rainfall during late spring and summer, the soil dried out and water tables fell to depths below the bottom of the observation wells, as indicated by the flat portion of the water table plots (about 1.4 m deep) in figure 8. Then the cycle repeated. In some years (e.g., years 6 and 8) rainfall was not sufficient to raise the water table to near surface in autumn, and the wet period in the following winter/spring was delayed or never fully developed (e.g., year 7). Note that the water table depths and cumulative outflows for the control (LC1, pasture) and treatment (LC2,

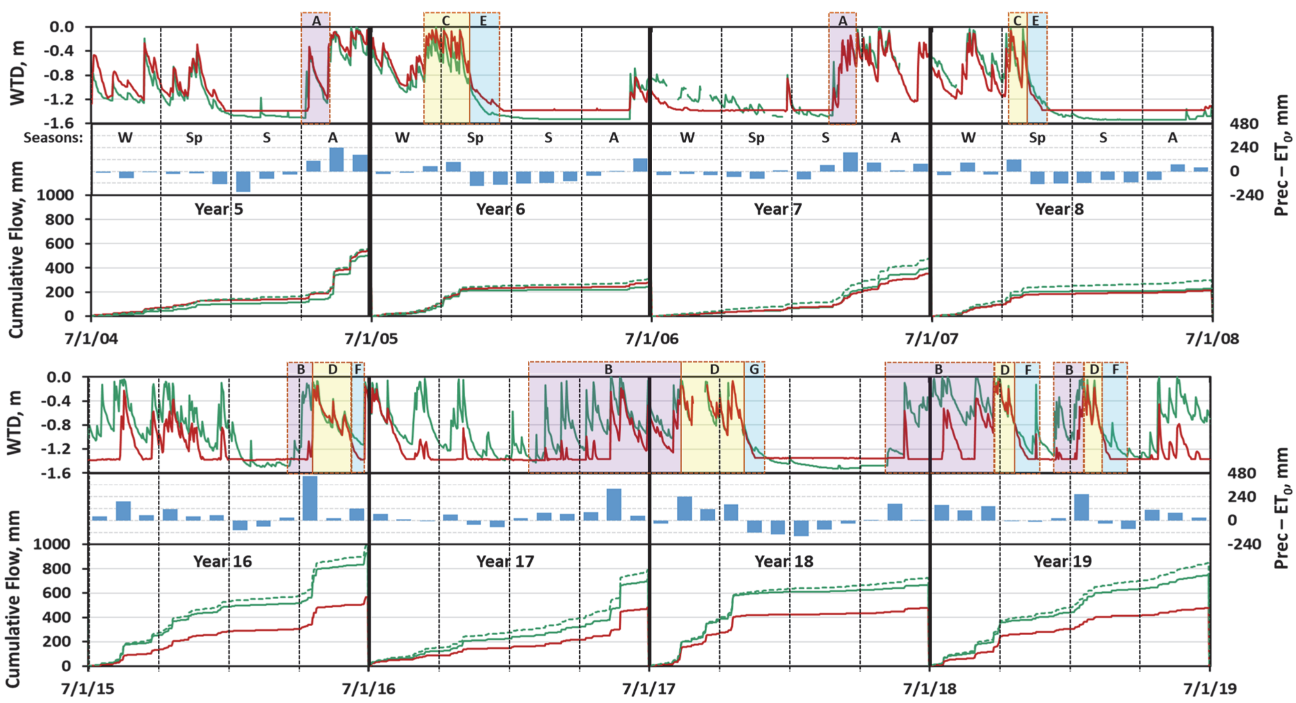

Figure 8. Time series of daily water table depths (WTD), monthly precipitation minus reference ET (Prec - ET_o), and daily cumulative flow during early tree establishment (years 5–8; top plots) and when trees are mature (years 16–19; bottom plots). Continuous lines are derived from observations for LC1, the control watershed in pasture, and LC2, the treatment watershed planted in pine, in green and brown, respectively. The dashed lines correspond to calculated cumulative flow at LC2, had it remained in pasture (LC2EX). W, Sp, S, and A denote the Winter, Spring, Summer, and Autumn seasons. The quantity (Prec-ET_o) is plotted (bar chart) for each month. Selected time periods of interest are shown in colored rectangles. Purple for water table recovery after dry periods. Yellow for water table drawdown when water table depths are less than 0.9 m. Blue for water table drawdown when water table depths are greater than 0.9 m.

pine) watersheds were similar for years 5–8 when the pine was young (2 to 5 years after planting). While similar, the water tables at LC2 were generally higher than at LC1 in years 5 and 6, but lower than at LC1 in years 7 and 8. Differences of cumulative flows between LC2 and LC2EX also begin to appear in years 7 and 8, consistent with previous observations of changes in outflow beginning in year 7.

Similarities between LC1 and LC2 were no longer the case as the pine matured. Starting in the latter part of year 10 (not shown) and following, there was a notable difference in the water table depths and outflows in the forested (LC2) and pastured (LC1) watersheds, as shown for years 16–19 in figure 8. Although these years were wetter than years 5–8, as the trees matured, the water table in LC2 did not usually rise as high nor remain at shallow depths as long as it did in the LC1 pastured watershed. This difference is attributed to the deeper rooting depths of the pine trees as they matured, and hence the ability to remove more water from the profile during dry periods compared to pasture. Most outflow from the watersheds occurred during seasons when the difference, Prec-ET_o at the monthly scale (blue bars, fig. 8), was positive and the water table was elevated. Outflows during periods when the water table was deeper than 1.2 m were relatively low, as indicated by the flat slopes of the cumulative outflow curves. In contrast to years 5–8, outflows from the treatment (forested) watershed (LC2) during years 16–19 were substantially less than from the control (pasture) watershed (LC1) and from LC2 had it remained in pasture (LC2EX in fig. 8). Only during very wet periods did the water table in the forested watershed (LC2) rise to the soil surface.

The effect of afforestation on water table depth is also indicated in figure 7C. The indicator in figure 7C is based on cumulative measured water table depths in the control and treatment watersheds relative to cumulative flow at the control watershed. While the concept of cumulative water table depth is somewhat abstract, it may be seen simply as the summation needed to determine an average water table depth. The rainbow curve in figure 7C represents the difference between the cumulative water table depths in the two watersheds. The slope between any two points on this curve is the average difference in the water table depths between those points. This curve shows how the slope changes throughout the experiment and is a good visual indicator of the impact of the land use change on the water table. Figure 7C shows six clear trends (highlighted by the parallel dashed gray line with round symbols) for the water table depth indicator that are consistent with the phases discussed above for impacts on water yields (fig. 7B). During the pretreatment period and the first three years after planting (phase 1), the water table depth indicator has a negative slope, indicating that the water table was consistently deeper at the control watershed than it was at LC2 for the first 6 years of record. This is consistent with the observation of greater seepage from LC1 than from LC2. Over the next four years (phase 2, years 7 to 10), the slope of the indicator is near zero, indicating very little difference in water table depths. This suggests that increased ET from the deeper-rooted pine on LC2 compensated for the greater seepage from LC1, resulting in similar water table depths for the period. For years 11–15 (phase 3), the differences in cumulative water table depths changed to an increasing trend (slope is positive), suggesting that increasingly greater ET from the more mature pine trees resulted in water table depths consistently deeper on LC2 than on LC1. There is a clear inflection starting in year 16 for the beginning of phase 4, the trajectory of which is more variable with several flat sections corresponding to when the water tables rose to the surface together or when both recorders bottomed out (fig. 8). This trajectory becomes particularly flat following the commercial thinning at the end of year 17 and continues through year 18. Overall, the trends indicated in figure 7C are consistent with the water table observations shown in figure 8.

Water Balance Analysis

The continuous daily water table depth data collected in the study makes improved water balance analyses possible since time periods for the analyses can be selected that begin and end with the same water table depth and thus approximately the same soil water storages. Time periods were chosen to begin and end during receding water table conditions to minimize errors due to hysteresis. This minimizes the change in soil water storage over the selected periods and simplifies the water balance to ET = Precipitation - Outflow, allowing for a reasonable estimate of ET over the time period as well as more accurate outflow estimates for comparing LC2 to LC2EX. How ET and outflow vary with stage of growth was analyzed using a simple mass balance for selected periods during the years plotted in figure 8; results are given in table 2.

Afforestation had little effect on ET during years 5–8 (2 to 5 years after planting) compared to its impact in later years. Average daily ET ranged from 2.1 to 2.6 mm/d for the treatment watershed (LC2), while the average daily ET that would have been expected in the absence of afforestation (LC2EX) on the treatment watershed ranged from 2.1 to 2.4 mm/d (table 2). During the later years (16–19), average daily ET for the

Table 2. Results of a mass balance for selected periods from years 5 to 8 (2 to 5 years after planting) and from years 16 to 19 (13 to 16 years after planting) showing the effects of afforestation on outflow and evapotranspiration.

	Selected Period					Outflow			ET (daily average)					
	Start	End	Days	Rain (mm)	ET_o (mm)	LC2 (mm)	LC2EX (mm)	%Dif	LC2 (mm/d)	LC2EX (mm/d)	Dif	ET_o (mm)	ET/ET_o LC2	ET/ET_o LC2EX
YR5	9/16/04	7/28/05	316	1281	1309	504	524	-3.7	2.46	2.40	0.06	4.14	0.59	0.58
YR6	8/10/05	6/13/06	308	897	1281	247	260	-5.4	2.11	2.07	0.05	4.16	0.51	0.50
YR7	6/13/06	10/25/07	500	1801	1587	524	697	-24.9	2.55	2.21	0.35	3.17	0.81	0.70
YR8	10/25/07	7/25/08	275	704	1173	71	104	-31.2	2.30	2.19	0.12	4.27	0.54	0.51
YR16	8/23/15	7/28/16	341	1796	1028	524	873	-40	3.73	2.71	1.03	3.01	1.24	0.90
YR17	7/16/16	6/2/17	322	1629	1107	419	692	-39.5	3.76	2.91	0.85	3.44	1.09	0.85
YR18	10/4/17	10/17/18	379	1594	1363	462	720	-35.8	2.98	2.30	0.68	3.60	0.83	0.64
YR19	10/7/18	11/8/19	398	2092	1245	615	1013	-39.3	3.71	2.71	1.00	3.10	1.19	0.87

treatment watershed (LC2) was greater, ranging from 3.0 to 3.8 mm/d. Differences in daily average ET between LC2 and LC2EX ranged from 0.05 to 0.35 mm/d for years 5–8 compared to 0.68 to 1.02 mm/d for years 16–19.

Further study of the water balance analyses for years 5–8 supports previous observations that the effects of afforestation were first evident in year 7. Afforestation reduced outflow from LC2 for years 5 and 6 by only 3.7% and 5.5%, respectively, compared to that from LC2EX. For years 7 and 8, afforestation reduced outflow from LC2 by 24.9% and 31.2%, respectively. Differences in daily average ET between LC2 and LC2EX were 0.05 and 0.06 mm/d for years 5 and 6, compared to 0.35 and 0.12 mm/d for years 7 and 8, respectively.

Water Balance Analysis of Commercial Thinning

Impacts of the commercial thinning that removed 52% of the tree biomass in June, YR 2017 (fig. 2) were analyzed by conducting water balance analyses on data collected both before and after the thinning. Prior to thinning, results show that, over two nearly year-long term periods, i.e., 341 days (8/23/15 to 7/28/16) during YR16 and 322 days during YR 17 (7/16/16 to 6/2/17), afforestation reduced outflow from LC2 by 40% and 39.5%, respectively, compared to that from LC2EX (table 2). Analysis of the 379-day period (10/4/17 to 10/17/18) following thinning showed outflow from LC2 was reduced by 35.8% compared to that expected from LC2EX. Further, ET from LC2 was reduced from an average of 3.75 mm/d for YR16 and YR17 (prior to thinning) to 2.98 mm/d for the 379-day period (YR 18+) after thinning. After that the hydrology returned to conditions close to those prior to thinning. Results of water balance analyses for the 398-day period (YR19+, table 2) showed that the reduction in outflow from LC2 compared to LC2EX rose to 39% and ET from LC2 to 3.71 mm/d. In summary, our water balance analyses indicate that a commercial thinning that removed 52% of the biomass from the afforested watershed LC2 reduced ET by an average of less than 1 mm/day and resulted in an approximate 4% reduction of the impact of afforestation on outflows for a relatively brief period, one to two years.

Effect of Afforestation on Flow Duration Curves

Outflows from both watersheds may be characterized as very flashy responses to rainfall with most of the flow occurring during a small proportion of the time. Over the 20 years of continuous flow measurements, an average of 77% and 69% of the annual flow volumes from the control (LC1) and treatment (LC2) watersheds, respectively, occurred in 5% of the time corresponding to the highest flows (data not shown). The highest flows (occurring only 2% of the time) accounted for an average of 67% and 58% of the total flow volumes from the control and treatment watersheds, respectively. Much of the hydrology was thus dominated by quick flow and surface runoff.

Flow duration curves (FDC) show an inversion of the ranking from pre- to post-treatment between control and treatment watersheds and suggest that the effects of afforestation were visible for the vast majority of flows (figs. 9A and 9B). FDC derived for the first four pre-treatment years show that flows at LC2 were greater than those of the control watershed nearly 100% of the time (fig. 9A). For the post-treatment 2011–2020 years, corresponding to phases 3 and 4 when the highest impact of afforestation was observed (fig. 7), flow rates at the treatment watershed were lower than those of the control watershed more than 92% of the time (fig. 9B).

The percentage effect of afforestation on FDC was estimated by the percentage difference between the expected (LC2EX) and the measured 2011–2020 post-treatment (LC2) flow values (fig. 9C). Afforestation decreased the 20% highest flows by between 40% and 50% or by more than 0.5 mm/d, although the effect on the top 1% flow was estimated to be much lower (~20%).

In absolute values, the 1st, 2nd, and 3rd percentiles of highest flows decreased by 16, 8.3, and 4.4 mm/d, respectively (not shown in fig. 9C). The percentage impact of afforestation increased from 45% to 70% from the 20th to the 60th percentile of sorted flows, and then stayed stable around 70% for the lowest 35% of the flows. In absolute values, flow decreases were lower than 0.5 mm/d for 80% of the flow corresponding to the lowest flows.

The numbers reported above correspond to the years where the impacts of afforestation were highest. Computing FDC over the whole period after planting (data not shown) reveals patterns similar to those displayed in figure 8C, only with lower values. The flow reduction for the 20% highest flows varied between 30% and 40%, instead of 40% and 50%, increased from 35% to 65% from the 20th to the 60th percentile of sorted flows, instead of 45% to 70%, and then stayed stable around 65% for the lowest 35% of the flows, instead of 70%.

Discussion

Reported herein are results of a long-term classic paired watershed study, which includes a calibration period. The paired watershed approach has been the standard study design for determining hydrologic impacts of changes in vegetation, land use, and management practices since the early 1900s (Andréassian, 2004; Bates, 1921; Neary, 2016). Few dispute the validity of the design to quantify changes in watersheds, particularly for post-harvest runoff (Alila et al., 2009). Laurén et al. (2009) demonstrated that uncertainty in pretreatment data and thus the calibration relationship of paired watershed studies may influence estimates of the magnitude and duration of the treatment effects. While the validity of the concept of the paired watershed approach is well recognized, detecting hydrologic effects of land conversions using this approach can be time-consuming, expensive, cost-prohibitive, and often limited by treatment options. Some dispute the value of long-term paired watershed studies because of the expense in time, effort, and funds required to collect one data point in space and time. This is evident in the limited number of long-term studies conducted to determine the hydrologic impact of afforestation. To our knowledge, this is the first long-term paired watershed study reported in the Pampas region of South America.

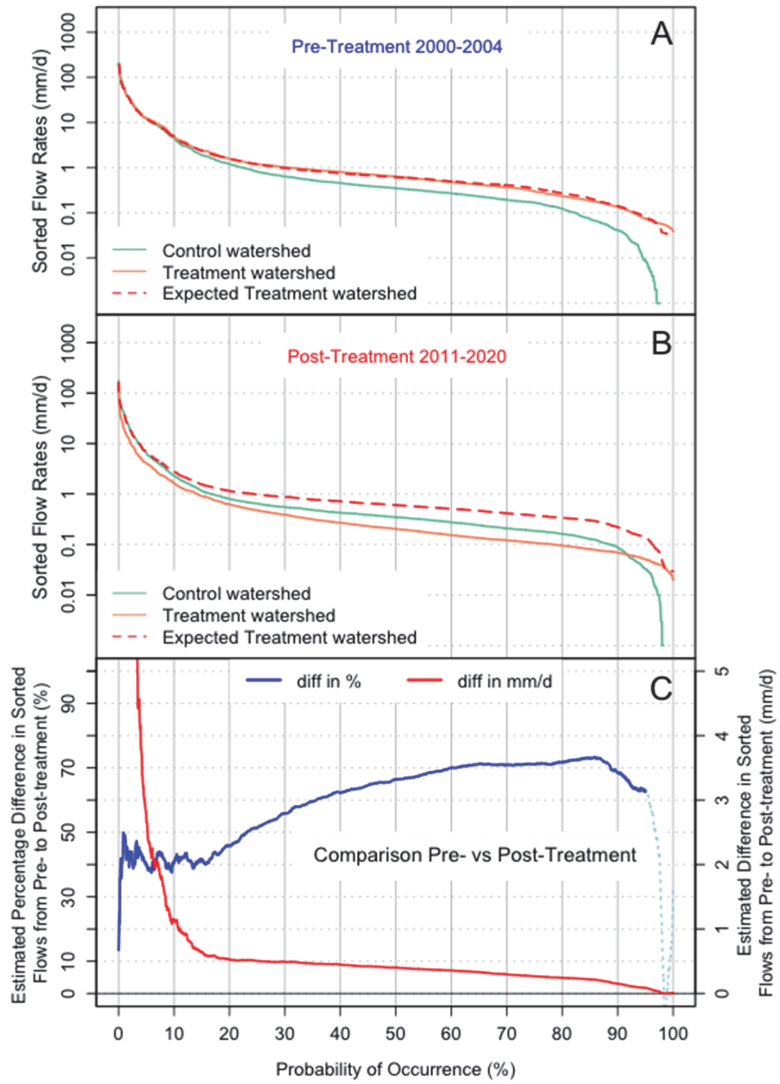

Figure 9. Flow duration curves derived over (A) the 2000–2004 period corresponding to the pre-treatment and first year after planting (top), (B) the similarly wet post-treatment years over the 2015–2019 period (middle), and (C) the differences between the curves for pre-treatment (LC2 expected) and post-treatment (LC2) periods (bottom).

Comparison to Other Paired Watershed Studies

In order to compare results of our study to those from reviews or other studies, we need to consider differences in site characteristics and weather patterns between studies. Literature reviews of paired watershed studies (Andréassian, 2004; Bosch and Hewlett, 1982; Brown et al., 2005; Farley et al., 2005; Neary, 2016; Sahin and Hall, 1996) do account for the effects of vegetation type and mean annual precipitation. Most reviews include studies of harvesting impacts as well as afforestation studies. They categorize studies based on forest type (scrub, conifer, eucalyptus, or hardwood) and, in cases of afforestation, the vegetation prior to treatment (grassland or scrub). Most reviews report that absolute changes (mm) in water yield increase with increasing average annual precipitation. Farley et al. (2005) report that in the case of afforestation, reductions in absolute water yield (mm) increase with average annual precipitation while relative reductions (%) decrease. For comparing our results to those of other studies, we limited our comparisons to studies of afforestation of grasslands with conifers, and to studies where average annual precipitation ranged from 1250 mm to 1750 mm.

Results of our paired watershed study are consistent with the body of knowledge presented in multiple literature reviews and support the findings that water yields from forested lands are less than from grasslands or recently harvested lands. Annual reductions observed in our study (368 mm or 48%) during the four-year period of maximum reductions (July 2012 to June 2016) were within the range (220 mm to 720 mm) reported by Brown et al. (2005) for conifer sites with precipitation near 1500 mm. Farley et al. (2005) limited their review to afforestation studies and reported mean annual yield reductions over the growth period of the trees. They reported annual yield reductions of 167 mm, or 40%, averaged for 9 studies where grassland was afforested by pine. Since the mean of average annual precipitation for the 9 studies reported was 1260 mm compared to 1489 mm in our study, our mean annual water yield reductions of 206 mm, or 35%, observed over the growth period of the trees in our study, are very comparable.

The way that results are analyzed and presented must also be considered when comparing afforestation studies. Most reviews reported that water yield impacts increased with increased affected area, but the nature of that relationship has not been determined due to variability. Assuming that this relationship is linear, Bosch and Hewlett (1982) and Sahin and Hall (1996) presented water yield changes in terms of mm per 10% change in cover. Scott et al. (2000) used similar methods, as well as, % reduction per 10% change. Farley et al. (2005), however, reported water yield changes uncorrected for the proportion of catchment area affected. When corrected for affected area, they reported increases in water yields on the order of 5%. Most of the reviews reported annual water yield changes for the periods of maximum impact (shortly after treatment in harvest studies or after canopy closure in afforestation studies), while Sahin and Hall (1996) and Farley et al. (2005) reported annual yield changes as the average over the period of study (from treatment to the end of the study). For comparing the results of our La Corona study to those of other afforestation studies of individual sites, we present water yield reductions that are adjusted for watershed area planted (assuming a linear relationship). We also present yield reductions for the period of maximum reductions as well as average reductions over the period of the study.

The individual paired watershed studies on afforested sites conducted at Cathedral Peaks, South Africa (Scott et al., 2000), and at Glendhu, New Zealand (Fahey and Payne, 2017), are most relevant to our study in that grasslands were afforested with pine trees (Pinus radiata) in locations with relatively high mean annual precipitation (1500 mm for Cathedral Peaks and 1330 mm for Glendhu). Results are compared in table 3. Absolute and relative water yield reductions during periods of maximum reduction were lower at La Corona than at the Cathedral Peaks sites but higher than at the Glendhu site. After adjusting for planted area, absolute reductions for La Corona (54 mm/10% planted area) were similar to those at Cathedral Peaks III (57 mm/10% planted area) but were higher compared to those at Cathedral Peaks II (48 mm/10% planted area) and at Glendhu (41 mm/10% planted area). Relative water yield reductions were greater at La Corona (7.5%/10% planted area) than at the Cathedral Peaks sites (6.7%/10% planted area and 6.0%/10% planted) and the Glendhu site (5.0%/10% planted). Absolute and relative water yield reductions averaged over the growth period of the trees were lower at La Corona than at Cathedral Peaks II but higher than at Cathedral Peaks III and the Glendhu site. The same pattern occurred for absolute reductions when adjusted for planted area, but relative reductions were greater for La Corona than for the other sites.

Reduction of water yield occurred sooner after planting at La Corona than at the Cathedral Peaks sites and the Glendhu site. Brown et al. (2013) estimated the time taken for change in streamflow to occur by fitting the model of Scott and Smith (1997) to the yearly streamflow data for afforested studies, including Cathedral Peaks and Glendhu. They estimated that 10% of the maximum reduction (10% of 273 mm) at Glendhu occurred 6 years after planting. At Cathedral Peaks II and III, they estimated that 10% of the observed reduction occurred 9 years and 5 years after planting, respectively. Ninety percent of the observed reductions occurred at Glendhu by 9 years after planting and at Cathedral Peaks II and III by 14 and 12 years after planting, respectively. Using the same methods as Brown et al. (2013), we estimated that 10% of the observed reduction at La Corona occurred by 3 years after planting and 90% of the observed reduction had occurred by 9 years after planting.

Water yield reductions at La Corona were generally greater than those at Glendhu and similar to those at the Cathedral Peaks. A likely reason for lower water yield reductions at Glendhu is that ET_o at Glendhu (900 mm/yr) was lower than at La Corona (1300 mm/yr). Water uptake by the trees and ET were likely higher at La Corona. ET_o at Cathedral Peaks (1200 mm/yr) was similar to that at La Corona. Water yield reductions over the entire treatment periods were generally greater at La Corona, particularly when adjusted for planted area. In the case of Glendhu, the differences are likely due to slower-growing trees. Trees at the Cathedral Peaks III site were damaged by fire two years after planting, which slowed the development of the plantation. The trees at both of the Cathedral Peaks sites were also damaged by insect larvae. Slower tree growth and tree damage may explain the lower water yield reductions over the treatment periods for the Cathedral Peaks experiments.

The impacts of afforestation on the distribution of flow rates within the flow regime have been reported for some paired watershed studies. Lane et al. (2005) and Brown et al. (2013) developed methods to assess the effects of afforestation on flow duration curves and applied those methods to the Cathedral Peaks and Glendhu watershed studies. Lane et al. (2005) reported that flow reductions across all percentiles were relatively uniform, with percent reductions ranging from 40% to 70% for the Cathedral Peaks watersheds and between 20% and 40% for the Glendhu watershed. Brown et al. (2013) also reported that flow reductions were relatively uniform across all percentiles, with percent reductions ranging from 35% to 50% for the Cathedral Peaks watersheds and between 20% and 25% for the Glendhu watershed. Using the methods of Brown et al. (2013), we determined that while flow reductions at La Corona were somewhat more

Table 3. Mean annual water yield reductions for La Corona and three other paired watershed studies where grassland was afforested with pine trees. Relative (%) and absolute (mm) water yield reductions are given for the period after canopy closure and for the entire treatment period. Yield reductions are given for the total watershed area and for the area only planted with trees, expressed as yield per 10% of planted area.

	Mean Annual Water Yield Reduction							
	Reduction after Canopy Closure				Reduction for Entire Treatment Period			
	Watershed Area		Planted Area		Watershed Area		Planted Area	
	mm	%	mm/10%	%/10%	mm	%	mm/10%	%/10%
Cathedral Peaks II	425	50	57	6.7	296	37	40	4.9
Cathedral Peaks III	410	52	48	6.0	194	26	23	3.0
Glendhu	273	33	41	5.0	198	24	29	3.5
La Corona	309	43	54	7.5	206	35	36	6.1

variable than at the Cathedral Peaks and Glendhu watersheds, they were still relatively uniform across percentiles, with reductions ranging from 40% to 75%.

Mechanisms

The long-term water table data set is unique to this study. While water table data have been collected and presented in other studies comparing grass and forest vegetation, long-term water table data from multiple wells have not been presented in other paired watershed studies of afforestation. These data provide insight into the mechanisms causing reductions of water yield due to afforestation and support current thinking about differences in evapotranspiration from grass and trees.

Most researchers generally agree that ET will be greater from a catchment covered with trees than from the same catchment covered with grasses or other herbaceous vegetation, except for extremely wet or dry conditions (Zhang et al., 2001). By catchment water balance, higher ET will result in lower water yield from the catchment with trees. Two general hypotheses exist about the mechanisms that lead to higher ET from trees: one related to the deeper rooting depth of the trees and the other relating to the taller and larger canopy of the trees.

Deeper roots allow mature trees to access a larger volume of soil water than is available to shallow rooted pasture vegetation (Hodnett et al., 1995). Trees continue transpiring soil water from deeper soil layers during dry periods when shallow rooted vegetation cannot access the deeper soil water. If deep-rooted trees use more soil water than shallow-rooted grasses when water tables are deep, then more water would be required after a dry period to saturate the soil and bring the water table to the surface under trees than under pasture. The observed water table data clearly show that more rainfall was required to bring the water table to the soil surface under mature trees than under grass (see period B, fig. 8) or young trees (see period A, fig. 8). Observed water tables were similar during very wet periods for both shallow- and deep-rooted vegetation (see periods C and D, fig. 8). When the observed water tables fell below 0.9 m, the drawdown rate became greater under mature trees than under grass (see period F, fig. 8) or under young trees (see period E, fig. 8). This would be expected as the distance between the water table and the root zone of the shallow-rooted vegetation increased, thus reducing the rate of upward flux of soil water from the water table to the root zone. Note that after the thinning operation (June 2017), the water table under the forested watershed (LC2) did not fall below that under pasture (LC1) until both reached a depth of 1.2 m (see period G, fig. 8). Prior to thinning (e.g., period F, fig. 8) the departure between water tables LC1 and LC2 was at a depth of 0.9 m. Analyses of the observed water table data support the hypothesis that a cause of higher ET from the mature forest is due to deeper roots accessing more soil water than available to shallow-rooted vegetation.

Researchers also hypothesize that the reference ET of trees is greater than the reference ET of grass (ET_o) due to the greater height and leaf area index (LAI) of the canopy, resulting in higher rainfall interception and lower aerodynamic resistance, and to decreased albedo, resulting in higher net radiation (Amatya and Harrison, 2016; Sun et al., 2016). Sun et al. (2010) used eddy covariance methods to measure ET from a 13-year old loblolly pine forest in eastern NC. Average annual ET was 13% greater than ET_o. Actual ET calculated from the water balance analysis of the mature forest at La Corona was likewise greater than ET_o in three of the four analyses (table 2). The other analysis where ET from the mature forest was less than ET_o occurred in the time period after commercial thinning. Water balance analyses that utilized the observed water table data support another current theory that the reference ET of trees is greater than the reference ET of grass.

Reduced Impact Near End of Study

Percent reduction of annual outflow due to afforestation at La Corona decreased from an average of 48% for years 13–16 to 38% for years 17–20 (table 1). Based on long-term studies, Scott and Prinsloo (2008) reported that as plantations matured, they had diminished effects on streamflows relative to earlier in the plantation cycles. They cite other long-term studies that show similar patterns, though with different time scales for different species and site-specific situations (Kuczera, 1987; Langford, 1976; Olbrich et al., 1993; Scott and Prinsloo, 2008). For example, Olbrich et al. (1993) found that young *Eucalyptus grandis* stands had peak transpiration rates at around three years of age and that stand-level LAI and transpiration declined after this peak. Langford (1976) and Kuczera (1987) found that increases in streamflow due to wildfires in mountain ash forests in Australia were followed by large decreases in flow during the regrowth and maturation of young ash, but streamflows increased toward pre-fire levels after 30 years. Amatya et al. (2016) found decreasing ET (potentially increasing streamflow) of a pine stand planted in 1974 in a coastal North Carolina site as it matured from 14 (in 1988) to 34 years (2008). Haydon et al. (1997) related increasing streamflow in these mature forests to declining sapwood area and interception loss. While such mechanisms may explain reduced impacts of afforestation on annual outflows of old plantations, we do not believe that they are the cause of the increase in outflows for years 17–20 compared to years 13–16 of the current study. The differences we observed were not statistically significant, and the commercial thinning, which reduced tree density by 54% (from 650 to 300 trees/ha), occurred during the later period. Our water balance analysis indicated this thinning reduced the impact of afforestation on annual outflow from 40% for the two years prior to thinning (years 16 and 17) to 36% for year 18. Impacts of the 2017 commercial thinning may have extended through years 19 and 20, having a role, although probably minor, in the relatively lower reductions in outflows due to afforestation in the later years (table 1). A global review of the impacts of thinning suggests that streamflow increases after thinning and the magnitude of streamflow increases depending on thinning intensity, site conditions, and precipitation and ET conditions (del Campo et al., 2022). The duration of streamflow increases in those studies ranged from 2 to 10 years depending on thinning intensity and ecological factors related to tree growth. The short duration of the streamflow increase at La Corona is likely due to the rapid recovery of the pine trees from the thinning operation (fig. 2).

MANAGEMENT CONSIDERATIONS

Afforestation will lead to reductions in water yields in Uruguay and needs to be considered in water management planning. The area of managed pine and eucalyptus forests increased from 158,000 ha in 1994 to 1,220,000 ha in 2023, which is 7% of Uruguay's land area (MapBiomass Uruguay, 2023). Currently, the greatest rate of land use change in Uruguay is the conversion of grasslands to row crop (grain) agriculture. Row crop agriculture expanded from 1,200,000 ha in 1994 to 2,850,000 ha in 2023. (Nosetto et al., 2012) predicted that conversion of grasslands to row crops in the pampas of Argentina will increase water yield, particularly if single cropping is used. Numerous other studies show that water yield is greater from row crop agriculture than from grasslands (Scanlon et al., 2007; Schilling et al., 2008). As the need to increase production of food and fiber increases, the hydrologic impacts of afforestation may serve to counteract the hydrologic impacts of converting grasslands to row cropland. While reductions in water yield will occur with afforestation in Uruguay, these reductions may not be as great or as potentially damaging as reductions in water yields in areas with less rainfall or where rainfall is substantially less than ET_o (Konapala et al., 2020). Significant areas of forest cover exist in other geographical locations that have annual rainfall and ET_o values similar to those in Uruguay (Konapala et al., 2020). Sun et al. (2006) used the Zhang equation (Zhang et al., 2001) to determine the impact of afforestation in China and estimated that water yield reductions of about 50% would occur in the semi-arid Loess Plateau region of northern China, while water yield reductions of 20% to 40% would occur in the more humid areas farther south. The authors also noted that yield reductions would not be as great on a regional scale since land uses across regions will be mixed due to the need for food production in the rural areas. Areas in the USA with rainfall and ET_o values similar to those in Uruguay include southern Georgia, Alabama, Mississippi, and Louisiana, where land uses are a patchwork of natural forest, managed forest, pasture, and cropland. Over 60% of these areas are forested and over 27% of these forests (17% of the total area) are pine plantations, yet water yields from these areas have supported significant populations, economies, and ecosystems. Water management planners will need to collect data from watershed studies along with watershed and regional-scale models that consider multiple land uses to plan for water yield changes occurring on a regional scale in Uruguay over a range of wet and dry weather conditions.

SUMMARY AND CONCLUSIONS

A paired watershed study was initiated in 2000 to determine the hydrologic impacts of changing land use from grassland (pasture) to pine plantation in the Tacuarembó region (Pampas grasslands) of Uruguay. Two adjacent watersheds (69 and 108 ha) were instrumented to continuously measure outflow rates, water table elevations, rainfall, and meteorological variables over a 20-year period (1 July 2000 to 30 June 2020). During the initial 3-yr pretreatment period, both watersheds were maintained in pasture. In July 2003, one watershed (LC2; 108 ha) was planted with loblolly pine, with trees covering 57% of the watershed area. The other watershed (LC1; 69 ha) remained in pasture. The effects of afforestation on outflow and water table depth were determined using several indicators, including statistical analyses of monthly and quarterly outflows, visual inspection of cumulative daily values for outflow and water table depth, and water balance analyses. Statistical analyses of study results clearly indicated that afforestation reduced outflow from the forested watershed (table 1, fig. 6). We calculated that afforestation reduced cumulative outflow over the 17 years of tree growth by 35%, or 3500 mm (207 mm/yr). Annual water yield reductions varied from year to year depending on stage of growth and weather patterns. Mean annual reduction in flow during the last 9 years of the study, when trees were mature, was 43% (309 mm/yr). Statistical analyses and visual inspection of daily cumulative outflows also showed reductions due to afforestation and that the reductions were first evident in the fourth year after tree planting (figs. 6, 7A, and 7B). Analyses of cumulative indicators for daily flow and water table depth showed four main phases of flow reductions. The first phase from year 1 to 7 with no reduction followed by a 23% reduction in years 8 to 10. The greatest reduction (47%) occurred in the third phase (years 11 to 16) before dropping to 38% in the last 4 years (fig. 7). During the final phase, a 33% reduction in water yield was observed for one year following commercial thinning. A water balance analysis determined that the yield reduction for a 379-day period after commercial thinning was 36% compared to 40% for the preceding and following years (table 2). Water balance analyses indicated that ET from the mature trees was greater than from grass and younger trees (table 2). These analyses, along with water table observations, support the current theory that deeper-rooted trees increase ET by accessing more soil water than shallow-rooted vegetation and by increasing ET_o due to decreased albedo and reduced aerodynamic resistance. Analysis of daily flow duration data indicated that flow rates eight years after planting were reduced between 40% and 50% for high flows occurring less than 20% of the time and reduced by 50% to 70% for the remaining low flows (fig. 9). Our water yield reduction results are consistent with other afforestation research studies in regions with similar rainfall (table 3).

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