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Revealing the impact of forest exotic plantations on water yield in large scale watersheds in South-Central Chile

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SUMMARY

Land-use and forest cover change play important roles in socio-economic processes and have been linked with water supply and other ecosystem services in various regions of the world. Water yield from watersheds is a major ecosystem service for human activities but has been dramatically altered by landscape management superimposed on climatic variability and change. Many studies from different regions of the world have documented that in small watersheds (<100 ha) fast-growing forest plantations reduce water yield. Nevertheless, these effects have not been adequately documented in large watersheds >10,000 ha. In this paper, we examine the temporal variation of the residuals between best-fit precipitation–runoff relationships and instrumental streamflow records for two large watersheds (Purapel en Nirivilo, PNN and Cauquenes en el Arrayán, CQA) located in the Mediterranean-climate coastal range of South-Central Chile. In these watersheds, high resolution satellite imagery shows a decline in native forest cover from 52.3% to 14.2% for PPN and 36.1% to 8.1% in CQA, between 1975 and 2000. Conversely, in the same period the percentage area covered by forest plantations, mainly *Pinus radiata,* increased from 12% to 55% in PPN, and 4.7% to 42% in CQA. We observed a decreasing trend in summer runoff residuals regressed against annual precipitation in the same period, with slopes significantly different from zero for PPN ($p = 0.035$) and CQA ($p = 0.008$). We interpreted this pattern as an evidence of change in the hydrological regime in these watersheds as a consequence of forest cover and land-use changes. From a reanalysis of the observed data we estimate a decrease in runoff from 13.1 to 7.5 mm/summer for PPN and from 7.3 to 5 mm/summer for CQA, refer to the period 1991–2000 compared to 1981–1990. Multiple regression analyses of annual and seasonal flows show that besides precipitation, percentage-cover of forest plantations is a statistically significant predictor of summer flow with a partial negative correlation of −0.45 and −0.44 for PPN and CQA, respectively, $p < 0.05$. This study clearly shows the important effect that land-use change can have on water yield and to our knowledge this is the first study documenting the decrease in summer runoff in a landscape where native forest cover has dramatically declined and forest exotic plantations have expanded. Similar methods could be used elsewhere to inform policy and decision-making regarding forest and land-use planning.

Introduction

Streamflow generation at different spatial and temporal scales can be generally seen as a complex interaction of terrestrial factors such as watershed geomorphology, soil type, vegetation and land-use, with atmospheric factors (precipitation, temperature, air humidity, wind, etc.), and the spatial and temporal variability of the aforementioned variables. Although it is usually accepted that precipitation is the major determinant of large-scale variability in monthly, seasonal and annual flows (Ward and Trimble, 2003). It is relevant to assess the effect of other influencing factors on water availability. In the context of increased awareness of the potential effects of climate change on water resources, other sources of variability, with arguably similar consequences and subject to more certain mitigation actions, should be addressed. One such factor is land cover, which directly affects interception, infiltration (and subsurface storage replenishing) and runoff processes across various temporal and spatial scales.

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Precipitation has decreased by approximately 40% in the last century in South-Central Chile (1901–2005, Trenberth et al., 2007). This climatic trend has caused, for example, a pronounced decrease in the Puelo River streamflow (41° and 42°20′S) documented from tree-rings and instrumental records since 1943, and this river may be representative of a larger geographic region throughout South-Central Chile and adjacent Argentina (35° to 46°S, Lara et al., 2008). In addition to precipitation changes, temperature has increased by 0.25 °C/decade in the Central Valley and western Andes of Central Chile (1979–2006, Falvey and Garreaud, 2009), a situation that may have affected the evapotranspirative demand of vegetation.

From a water yield standpoint, an important scientific approach has been that of comparing the effect of different land-uses, such as forestry, grassland, and urban on streamflow; the consensus being that forested areas "use" more water than grasslands (e.g. Ward and Trimble 2003; Andréassian, 2004). However, the effect of forested areas on low seasonal flows is less understood, as higher infiltration rates in forests may increase aquifer recharge, thus supporting base-flow during dry periods. The study of this effect is scale-dependent, because it is influenced by the spatial distribution of forests with respect to both areas of high aquifer recharge and base-flow generation (Calder, 1993).

Forest cover and land-use change are important variables in the hydrological cycle (Bosch and Hewlett 1982; Calder 1992; Calder et al., 1997; Iroumé and Huber 2002; Farley et al., 2005; Brown et al., 2005; Huber et al., 2008; Lara et al., in press). Several studies have documented the reduction of water yield in regions undergoing expansion of fast-growing forest plantations, revealing clear policy implications regarding land-use (Farley et al., 2005; Jackson et al., 2005). In Chile, Huber et al. (2008) reported the negative effect of *Pinus radiata* and *Eucalyptus* spp. exotic plantations on water balance compared to pasturelands and shrublands. A recent study documents a negative linear relationship between streamflow and forest plantations area (in percentage), and an inverse relationship for native forests (Lara et al., in press).

The assessment of the effects of forest cover and land-use changes on water yield has mainly been conducted in small watersheds <100 ha, most likely because at this scale pure vegetation types are easier to find, controlled vegetation treatments more likely to be successfully implemented and the effects of forest cover are easily detectable (Bosch and Hewlett 1982; Calder 1992; Scott and Lesch 1997; Andréassian, 2004; Brown et al., 2005; Jackson et al., 2005; Farley et al., 2005; Lara et al., in press). Nevertheless, for large watersheds (e.g. >10,000 ha) there is a lack of empirical evidence of runoff generation processes responding to land-use cover change, while precipitation variability is seen as the driving factor in streamflow (Wilk et al., 2001; Pizarro et al., 2006). Scaling up the effects of land-use and forest change on runoff from small to large-scale watersheds is not possible without understanding land-use change effects on large watersheds.

In South-Central Chile under a Mediterranean-type climate, differences in water balance of forest plantations compared to grasslands have been explained as a consequence of canopy interception loss, changes in transpiration rates and their effects on subsurface storage (Huber et al., 2008). Because subsurface water storage is the main determinant of late summer base-flow in most watersheds in South-Central Chile, these relationships provide a clear indication of the effects that the conversion of native forests to plantations could have on water yield. Furthermore, the increase in water demand by fast-growing forest plantations intensifies with plantation age and density as well as climate trends referred to precipitation and temperatures regimes.

Given the challenge of documenting the effect of land-use and forest cover change on runoff for large watersheds, the objective of this paper is to quantify the change in runoff associated with the conversion of deciduous *Nothofagus* spp. native forests to exotic *P. radiata* plantations in the Coastal Range of South-Central Chile. To accomplish this objective we analyzed precipitation and streamflow instrumental records and land-use cover changes from satellite imagery corresponding to two catchments located in the above mentioned setting.

Data and methods

Watersheds, climate, forest cover and land-use change detection

We selected two watersheds defined by the Purapel en Nirivilo (PPN) (35°34′S, 72°12′W) and Cauquenes en El Arrayán (CQA) (35°57′S, 72°27′W) gauging stations, which are maintained by the Dirección General de Aguas (Dirección General de Aguas, 2006) or Chilean Water Bureau. The contributing areas for the PPN and CQA basins are 252.5 and 707.7 km^2, respectively. These watersheds are located in the Coastal Range of the Maule Region and experience a Mediterranean-type climate (Fig. 1). Annual precipitation amounts range between 835 mm and 717.5 mm in Quella (36°03′S, 72°05′W, QLA) and Nirivilo (35°32′S, 72°05′W, NRV) stations, respectively, with a maximum mean in June of 192.3 mm and 155.7 mm and a minimum mean in January of 3.95 mm and 5.01 mm for 1961–2004, respectively (Fig. 1; Dirección General de Aguas, 2006). Precipitation is highly concentrated, with 85% of the annual volume falling between April and September, and presents high inter-annual variability, with coefficients of variation in annual precipitation of 33% and 27% for PPN and CQA, respectively (Fig. 3). Mean annual temperature is 14.1 °C, with a mean maxima of 20.5 °C, in January and a mean minima of 8.2 °C in July (Parral station, 36°11′S, 71°49′W, Dirección General de Aguas, 2006). Soils originate from their hillsides erosion, the main texture being intermediate to fine sands. They belong to the Reguegua, Las Garzas, Purapel, San Lorenzo, Huapi, Vaquería, Cauquenes, Quella and Quipato soil series (Pizarro et al., 2006).

Forest cover and land-use change history for the PPN and CQA watersheds was obtained from Landsat imagery for the years 1975, 1990 and 2000. For CQA the available images covered 93% of the watershed area. These satellite images were geometrically, atmospherically and topographically corrected using ERDAS software (Environmental Systems Research Institute, Inc., USA) following a protocol similar to that described by Echeverría et al. (2006). A supervised land-use classification was performed; using 100 points visited on the field. We validated the classification in the field by applying a stratified random design by land-use type. For fieldwork classification we described the formation, composition, dominant species and degree of disturbance of 20–50% of all the vegetation stands (polygons) for each watershed, following the protocol described by Lara and Sandoval (2003). This protocol discriminates structure (old-growth, second-growth, or mixtures between them), height classes and canopy cover classes. We summarized and simplified these descriptions and arrived to the following land-use vegetation classification: (1) Native forests, mainly dominated by second-growth forests of deciduous *Nothofagus obliqua* and *Nothofagus glauca* and broadleaved evergreen sclerophyllous forests including species such as *Acacia caven*, *Quillaja saponaria* and *Maytenus boaria* (Donoso, 1993; Echeverría et al., 2006). Endangered species such as *Nothofagus alessandri*, *Pitavia punctata* and *Gomortega keule* can be found in the area as well (Donoso, 1993; Echeverría et al., 2006); (2) Forest exotic plantations, mainly *P. radiata* with different age classes managed with a 20-year rotation (Echeverría et al., 2006; Lara et al., 2006; Pizarro et al., 2006), established in a process that started during the early 1970s and has continued at an accelerated rate. This category included young exotic plantations (under 3–4 years) and recent clearcuts (<2 years after the image

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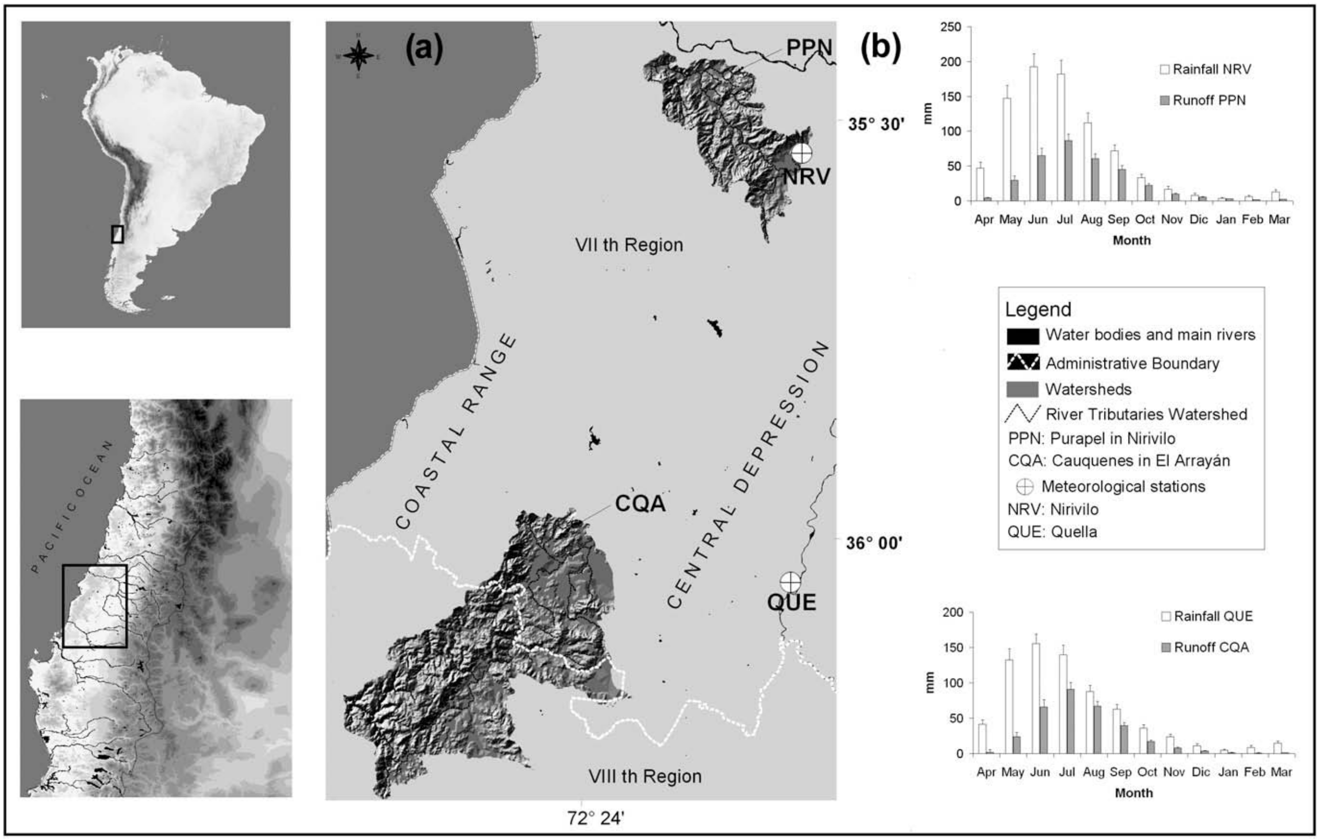

Fig. 1. (a) Regional map showing the location of the watersheds including the precipitation station Quella (QUE) and Nirivilo (NRV) and (b) monthly precipitation in Nirivilo (NRV) and Quella (QUE); and monthly runoff in Purapel en Nirivilo (PPN) and Cauquenes en el Arrayán (CQA) watersheds.

date); (3) Shrublands of various height classes (1–4 m) and canopy cover (50–75% and 75–100%), dominated by *Baccharis* spp., *Schinus polygamus* and *Acacia caven* and (4) Agriculture and pasturelands. Final maps were developed and incorporated into a Geographic Information System using Arcview 3.3 (Environmental Systems Research Institute, Inc., USA) (Fig. 2).

Streamflow, precipitation and statistical data analysis

We examined daily streamflow data provided by Dirección General de Aguas (2006) from the PPN and CQA gauging stations, and meteorological (precipitation) information from the Nirivilo (NRV) and Quella (QUE) stations (Fig. 1). Data from these stations is of good quality – in terms of data gaps – compared with the average for the region, and covers an adequate period of record that extends from 1961 to 2004. We verified the daily streamflow and precipitation data exhaustively in order to detect anomalous values that could cause errors when computing monthly means. Likewise, in order to minimize the risk of bias, monthly averages were computed only for those months having more than 20 days of available daily data for the month. Otherwise, we estimated the missing monthly value by regression with nearby stations having similar hydrologic regime, stream network complexity, watershed size, and land-use. For PPN we filled in 17.4% of missing data (monthly streamflow values), and 14% in CQA. In the case of precipitation, 2% and 1.8% of the data were filled in for NRV and QUE, respectively. Based on this data we computed water-year (April to March) and seasonal totals, with seasons defined as 4-month periods: winter (AMJJ), spring (ASON) and summer (DJFM) (Niemeyer and Cereceda, 1984) (Fig. 2).

In order to isolate the strong effect of precipitation variability and to highlight the potentially more subtle implications of land-use change on flow regime, we analyzed the annual and seasonal residuals, computed as the difference between observed flow and that predicted from a best-fit statistical relationship including only annual precipitation as the predictor variable. We performed linear regression analysis between the annual and seasonal time series of log-precipitation and log-streamflow. The underlying hypothesis was that if runoff is solely a function of precipitation, a relation of the type:

$$q_i = \pi(P_i) + \varepsilon_i$$

where q_i = annual or seasonal runoff; P_i = annual precipitation; π = is a function; and $\varepsilon_i \propto N(0, \sigma_\varepsilon^2)$, should hold true. Any effects different from precipitation controlling runoff are assumed to be constant, embedded in the coefficients of the function π, and independently distributed. Random effects are represented by ε_i.

We analyzed the trend of the residuals for annual values and different seasons. If land-use changes were affecting the precipitation–runoff relationship, we would expect that the residual term ε_i would not be constant in time. Therefore we used a null hypothesis of no trend for the residuals as a function of time. When plotted against time, the behavior of these residuals can attest to systematic variations in the processes that produce streamflow within the watershed. Positive (negative) residuals can be interpreted as the watershed generating less (more) flow than predicted by precipitation, thus any trends in time affecting the residuals allow us to infer that an additional predictor for streamflow variability must exist (statistical error of estimations).

In order to obtain a quantitative estimate of the summer runoff trend in time, we reconstructed runoff for the observed historical period using the least-squares linear equation using the mean annual rainfall value for 1960–2004 (see caption, Fig. 3) as an independent variable and added a 5-year moving-average smoothed

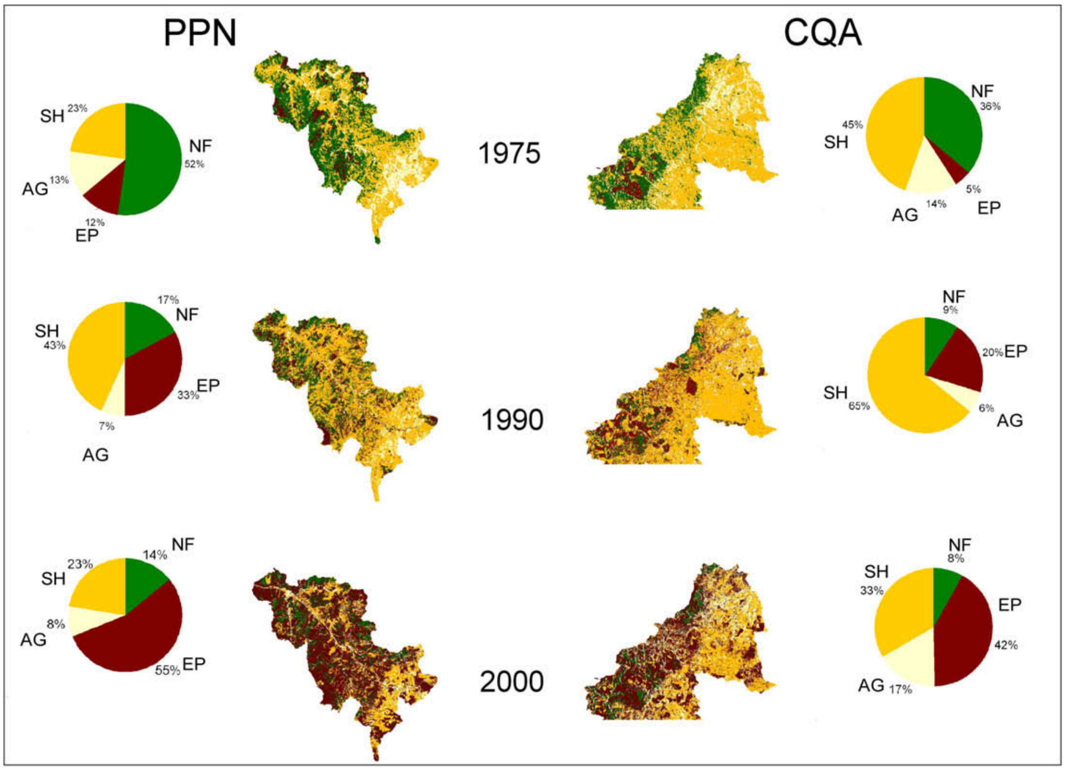

Fig. 2. Total area in squared kilometers and percentage of the different land-use/forest cover classes in the Purapel in Nirivilo and Cauquenes en El Arrayán river watersheds for 1975, 1990, 2000 time period. (NF: native forest; EP: forest exotic plantation including young plantation and recent clearcuts; AG: agriculture and pasture land; SH: shrubland.)

estimation of the summer runoff residual. This method permitted the analysis of the long-term trend in summer runoff, removing the strongest features of inter-annual variability of both precipitation and residuals.

Finally, we applied the multiple regression model following the forward stepwise technique, in order to estimate a functional relationship between seasonal runoff as the dependent variable, and precipitation and land-use cover as independent variables (Cooley and Lohnes 1971). We developed a family of equations for the same dependent variable (e.g., summer runoff), each including independent variables not mutually correlated. Finally, we selected the equation that achieved the maximum adjusted coefficient of multiple determination (R^2_{adj}), with a significant model ($p < 0.05$) (Sokal and Rohlf 1995). All equations were checked for normal distribution of raw residuals. The analyses were performed using Statistica 6.0 software (Statsoft, Inc., Tulsa, Oklahoma, USA). We correlated the log runoff data with the log-precipitation and percentage of land-use over a 5-year window centered on the time stamp of the satellite image. For example, precipitation and streamflow data for 1973–1977 was correlated with land-use cover of the 1975 image. We tried five different yearly windows, and this was the one that provided the highest correlations.

Results

Land use-land cover change (1975–2000) and runoff trends

The satellite images show important changes in land-use and forest cover during the 1975–2000 period in PPN and CQA (Fig. 2). Native forests declined from 52.3% in 1975 to 14.2% in 2000 at PPN, and from 36.1% to 8.1% at CQA. In the same period *P. radiata* exotic plantations increased from 12% to 55% at PPN and from 4.7% to 42% at CQA (Fig. 2).

Mean runoff monthly series calculated for the 1961–2004 period for both watersheds show a clear response to precipitation under a pluvial regime with a maximum in July, a month after maximum monthly rainfall (Fig. 1). Over 95% of the annual streamflow value is concentrated between April and September (Figs. 1 and 3). The runoff magnitude is very low in the summer season (December to March) representing only 3.7% and 2.2% of the annual value for PPN and CQA, respectively. Runoff time series show high inter-annual variability in both watersheds, responding to annual rainfall variability (Fig. 3). Annual and seasonal runoff time series show significant Pearson's correlation with annual precipitation (Fig. 3).

Annual trends of the residuals indicate different patterns for PPN and CQA (Fig. 4). We only found a positive and significant trend in the residuals for CQA for annual and winter runoff regressed against annual precipitation ($r = 0.42$, $p = 0.0047$ and $r = 0.44$, $p = 0.0026$, respectively). However, both watersheds show decreasing trends in the residuals of summer runoff with slopes significantly different from zero ($p = 0.035$ and $p = 0.008$ for PPN and CQA, respectively (Fig. 4). An important feature observed after 1990 for both catchments is that the adjusted values of the residuals were always negative, yielding estimated runoff values consistently below the 44-year mean for both watersheds (Figs. 3 and 4). Runoff for PPN shows a relatively stable pattern before 1990, oscillating between 10 and 18 mm/summer, whereas CQA presents a similar pattern of variation, with values between 6 and 8 mm/summer in the same period (Fig. 5). We estimated an average decrease of 13.1–7.5 mm/summer and 7.3–5 mm/summer for PPN and CQA summer runoff, respectively. These estimates refer to the period 1991–2000 compared to 1981–1990 (Fig. 5).

The multi-regression analysis showed that annual and seasonal runoffs were mainly and significantly explained by annual precipitation. Nevertheless, for summer runoff, a negative significant correlation with the percentage of forest exotic plantation appears in

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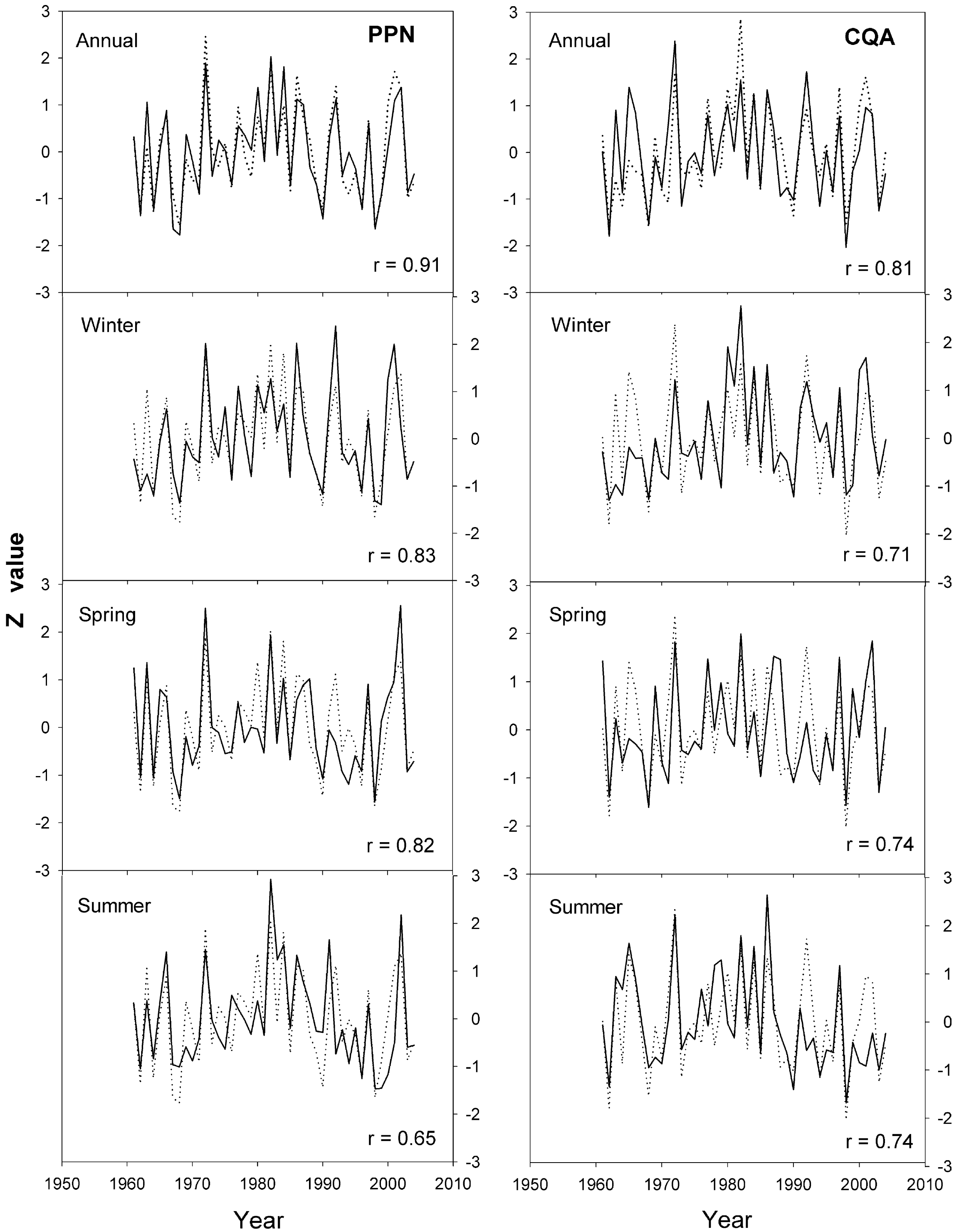

Fig. 3. Annual precipitation (dashed lines) related to annual and seasonal streamflow[1] (solid lines) variability for PPN and CQA watersheds during 1961–2004. Pearson's coefficient of correlation (r) between annual precipitation (logarithmic unit) and annual and seasonal streamflow (logarithmic unit) for PPN and CQA ($n = 44$). All r coefficients are significant $p < 0.001$. Z values represent a standard score considering a normal distribution of the value, with mean cero and variance 1 (Zar, 1999).

addition to precipitation (Beta coefficients of −0.45 and −0.44 for PPN and CQA, respectively; $p < 0.05$, Table 1).

[1] Units are expressed in millimeters (mm) and the standard error is presented in parentheses (SE). Period 1961–2004 ($n = 44$). PPN: Mean annual rainfall 874.9 (43.6), mean annual runoff 336.6 (28.8), winter runoff 185.8 (18.9), spring runoff 138.3 (12.4), summer runoff 12.5 (1.3); CQA: Mean annual rainfall 753.7 (31.7), mean annual runoff 320.7 (26.6), winter runoff 181.9 (19.6), spring runoff 131.7 (10.6), summer runoff 6.9 (0.5).

Discussion

Land-use change

The large-scale treatments imposed by the expansion of commercial forest plantations in South-Central Chile offers a valuable experimental setting. In the study area, land-use analyses indicate a fast conversion process from native forests to plantations in

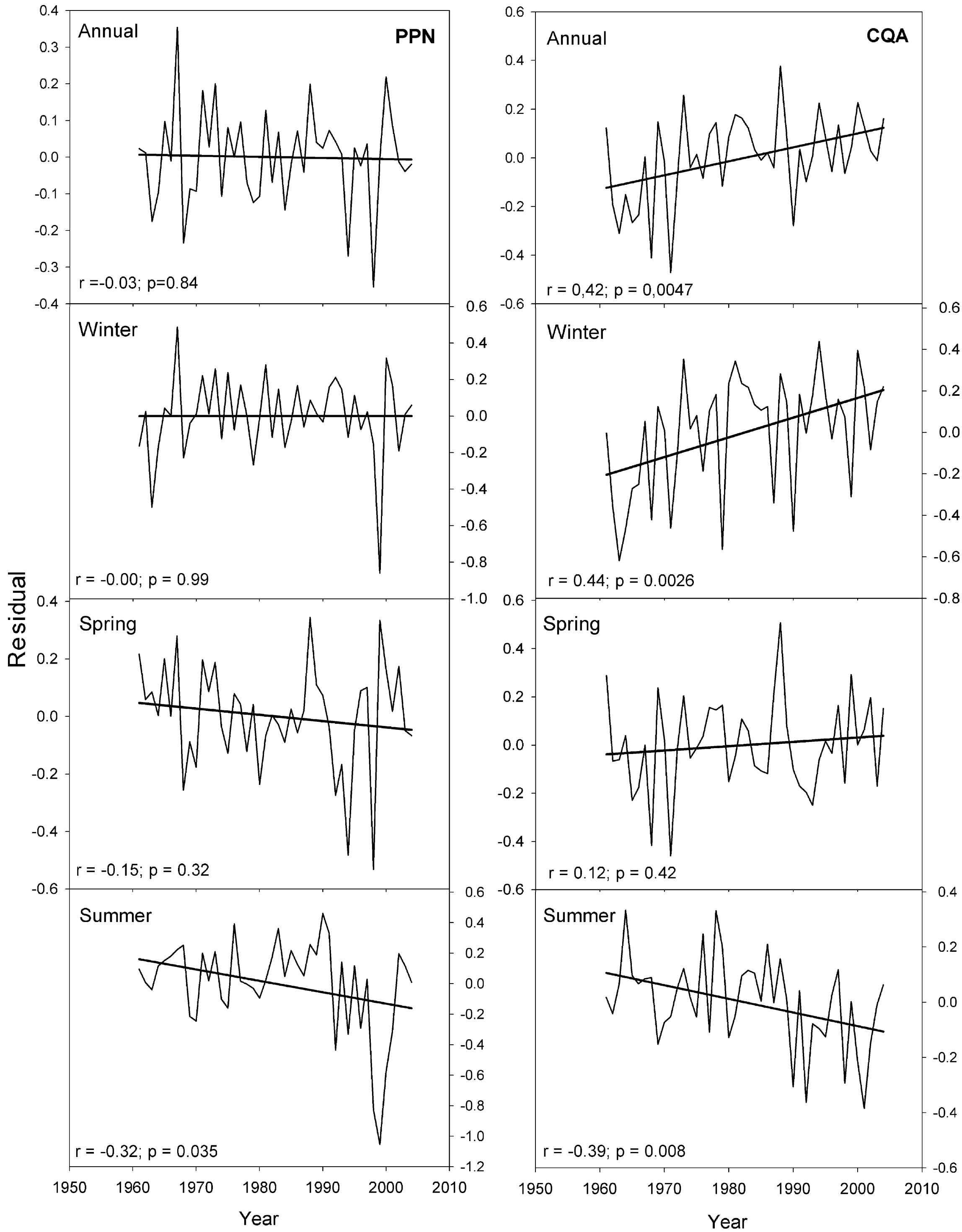

Fig. 4. Residuals of the annual and seasonal streamflow from its regression against annual precipitation during 1961–2004. A trend line has been included in each graph, indicating the Person's coefficient of correlation (*r*) and *p* the probability that the slope is different from 0.

both watersheds. These results are consistent with the conversion of native forests to exotic plantations at an estimated mean annual rate of 4.5% from 1975 to 2000 in 0.58 million hectares in the Maule region, including the PPN and most of CQA watershed (Echeverría et al., 2006). Similarly, results obtained by Pizarro et al. (2006) using aerial photographs in the PPN watershed, show a reduction in native forest area from 52% in 1978 to 20% in 1997 and an increase in *P. radiata* plantations from 19% to 52% in the same period.

Echeverría et al. (2006) showed that the main change in forest cover was through more than half (53%) of native forest cover (existing in 1975) gradually being converted into exotic-species plantations by 2000. Another substantial area (40%) was the transformation in shrublands. Shrublands increased from 1975 to 1990 due to the clearing of *Nothofagus* spp. second-growth forests. At a later stage, by 2000, most of these shrublands had been covered to exotic plantations, explaining the decrease in shrubland area. In addition, the conversion of

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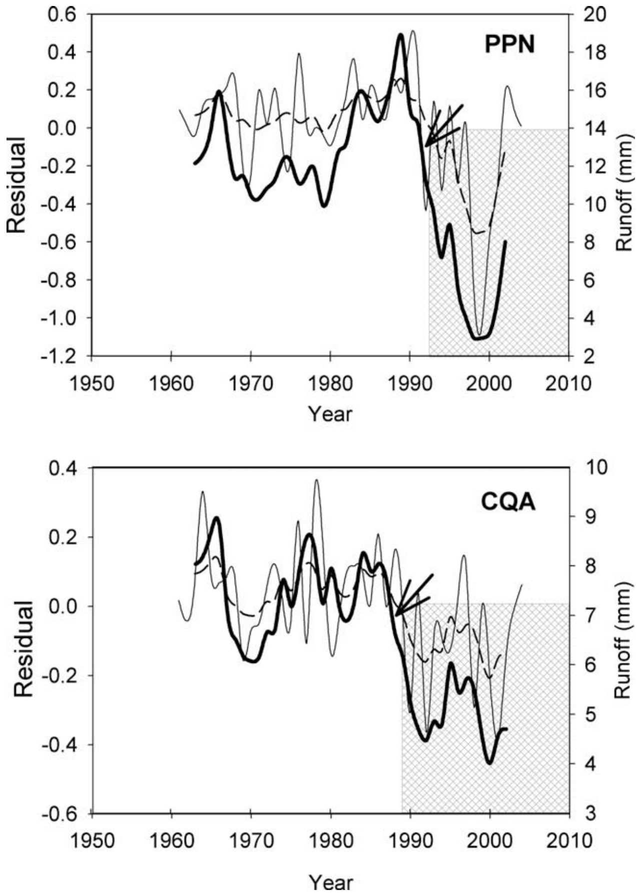

Fig. 5. Residuals of the summer runoff (December–March) from its regression against annual precipitation during 1961–2004 (thin solid line) for PPN and CQA. Trend line of the residuals from a moving-average smoothed are shown in dashed line. Bold solid lines represent the temporal change in total summer runoff predicted using the linear equation that relate summer runoff and annual precipitation form Fig. 2, where $q_i = -3.92 + 1.68\ P_i + \varepsilon_i$ for PPN and $q_i = -3.62 + 1.42\ P_i + \varepsilon_i$ for CQA. P_i is the mean annual rainfall observed in the period (See caption Fig. 3) and ε_i is the residual value from a moving-average smoothed. The arrows indicate the mean runoff value between 1961 and 2004 and the shaded areas represent the period with negative residuals estimated by a moving-average smoothed.

native forests to exotic plantations contributed to forest fragmentation associated with a decrease in forest patch size and a rapid increase in the density of small patches (Echeverría et al., 2006).

Precipitation and runoff trends

Precipitation is highly seasonal, with 85% of the mean annual value for 1961–2004 concentrated during the Southern Hemisphere fall and winter (April–September). Annual rainfall has a high inter-annual variability, with coefficients of variation of 33% and 28% for NRV and QUE, respectively (Figs. 1 and 3). The runoff trends show that annual rainfall is an important determinant not only of annual runoff but also of seasonal runoff (Fig. 3). Changes in low flow are even more important than changes in annual flow, as the dry season is when reduced water supply has the most severe effect for users, particularly in arid and semi-arid regions (Scott and Smith, 1997; Farley et al., 2005).

South-Central Chile under a Mediterranean-type climate is a semi-arid region and therefore it is expected to be especially susceptible in runoff changes associated with the establishment of fast-growing forest plantations. PPN and CQA watersheds show similar patterns in land-use change with an increase in exotic forest plantations area and the decrease in native forests cover from 1975 to 2000, and both have a decreasing trend in the residuals between summer runoff regressed against annual precipitation for this period (Fig. 4). Nevertheless, for annual and winter runoff residuals only CQA increase in this period while PPN does not show a trend (Fig. 4). This difference in the hydrologic regime between both watersheds indicates that they can not be only explained by land-use change and vegetation cover expressed as percentage area. Other possible explanations that need to be further investigated include differences in geomorphology, plantation spatial arrangement, age and density, as well as clearcutting patch size.

Within the decreasing trend in the summer runoff residuals, negative values dominate since 1990 onward, with values in mm/summer always below the 44-year mean (Figs. 4 and 5). This trend shows a decrease of 42.7% and 31.9% for the period 1991–2000 compared to 1981–1990 for PPN and CQA, respectively (Fig. 5). In addition, the Multiple Regression Model of summer runoff regressed against precipitation and land-use area indicates a significant Beta partial correlation, positive for precipitation and negative for exotic forest plantation (Table 1).

These findings can be explained as a consequence of *P. radiata* forest plantations expansion and their effect in the reduction of summer runoff in large watersheds (252.5 and 707.7 km^2). These results are in agreement with other studies in Chile that have indicated that conversion of native forests to *P. radiata* and *Eucalyptus* spp. plantations decreases streamflow in small watersheds especially in summertime (Otero et al., 1994; Lara et al., in press). A decrease in 20.4% in summer runoff for each 10% increase in exotic forest plantation area in six watersheds using linear regression was showed by Lara et al. (in press). These results are also consistent with high canopy interception losses of 40% of the annual

Table 1
Beta coefficients, adjusted coefficient of multiple regression (R^2_{adj}), Fisher (F) and probability (P) values of the multiple regression models among the different runoff as a function of precipitation and the land cover categories, in PPN and CQA watersheds.

Log runoff	Log annual pp	Native forest (%)	Exotic tree plantation (%)	Agriculture and pasture land (%)	Shrubland (%)	R^2 (adj.)	F	p
PPN								
Annual	0.94*	–	–	–	–	0.87	$F_{1,13} = 94.8$	0.0000
Winter	0.83*	–	−0.19	–	–	0.69	$F_{2,12} = 16.9$	0.0003
Spring	0.79*	–	–	–	–	0.59	$F_{1,13} = 21.3$	0.0004
Summer	0.52*	–	−0.45*	–	–	0.42	$F_{2,12} = 5.92$	0.0153
CQA								
Annual	0.85*	–	–	–	–	0.69	$F_{1,13} = 32.9$	0.0000
Winter	0.78*	–	–	–	–	0.58	$F_{1,13} = 20.1$	0.0006
Spring	0.72*	–	–	–	–	0.49	$F_{1,13} = 14.5$	0.0022
Summer	0.59*	–	−0.44*	–	–	0.49	$F_{2,12} = 8.03$	0.0061

* Parameters that turned out to be statistically significant in the model with $p < 0.05$.

rainfall, net evapotranspiration of 55% for a *P. radiata* plantation in Palhuén, a site located near our study area (Huber et al., 2008). A decrease in percolation of 51% for a grassland to only 5% in the *P. radiata* plantation was also reported for this site (Huber et al., 2008). Similar patterns indicating high transpiration demands have been described for *Eucalypt* spp. and *P. radiata* stands (Calder et al., 1997; Scott and Lesch, 1997; Farley et al., 2005; Scott and Prinsloo, 2008).

A previous attempt to document the effect of vegetation cover change and the expansion of forest plantations on streamflow for the Purapel river basin, which corresponds to PPN in this study, could not demonstrate this effect (Pizarro et al., 2006). They concluded that streamflow changes in the 1960–2000 period could be statistically explained only by the decadal variability of precipitation.

To our knowledge there are not specific eco-physiological studies that have been published comparing transpiration rates of Chilean native trees compared to exotic-species used in commercial plantations. However, fast-growing tree species sustain high transpiration rates Arkley (1963), Lambers et al., 1998. Adult trees of *Eucalyptus globulus* in Ethiopia reach transpiration rates of 55 l per day (Fetene and Beck 2004), a volume four to five times higher than the amount transpired by slow growing trees such as *Podocarpus falcatus* and *Cupressus lusitanica*, with the same size and in the same environmental conditions. In semi-arid conditions in Argentinean North Patagonia, Licata et al. (2008) documented that *Pinus ponderosa* plantations had an average water use which is 64% and 33% greater than for a native *Austrocedrus chilensis* forest stand for wet and dry years, respectively. Water depletion occurred simultaneously at all soil depths for all the plots even in the wet year. However, the *P. ponderosa* plantation used a greater amount of water from deeper soil layers compared to all the other plots (Licata et al., 2008).

Despite the coincidence of our results with the general pattern of runoff decrease observed in forest plantations compared to other land-uses and vegetation cover, findings reported in this paper need to be taken with caution. This since it was developed from the land-use data for three dates (1975–1990 and 2000), each of them used as representative for a 5-year window whereas the precipitation and streamflow data was annual, which somewhat reduced the statistical power of the Multiple Regression Analysis.

Conclusions

This study makes a significant contribution to documentation and understanding of streamflow variability and to the identification of natural and human-induced forcing in runoff for large watersheds (i.e. >100 km^2). The residual analysis used here permitted the assessment of the decrease in summer runoff as a response to the increase in *P. radiata* plantations area, at the expense of native forests in the 1975–2000 in South-Central Chile. This response was masked by the inter-annual precipitation variability in a previous study done in the same area.

Other studies have demonstrated negative impacts of forest plantations on runoff for small watersheds. Nevertheless, to our knowledge this is the first study documenting the effects of land-use change and forest cover on runoff in large watersheds. Similar methods could be used elsewhere providing valuable information for policy and decision-making in forest and land-use planning.

In addition to precipitation-driven restrictions of water supply in the dry season, water availability problems have been intensified by an increase in water demand for several uses in the last decades (Lara et al., 2003) and may be more serious in the future due to a projected decreasing trend for precipitation and increase in temperatures due to climatic change (Fuenzalida et al., 2006). This situation could change the evapotranspirative demands of species and could decrease the water reserves (e.g. reduction in snow precipitation and accumulation) changing the hydrological patterns especially in mixed regime watersheds.

In addition to land-use change as an explanation to runoff variations, alternative hypothesis should be considered. Future studies might assess the potential influence of geomorphology factors, temperature variations and its evaporative effect, changes in storm intensities. Other hypothesis should test the effects of land-use spatial patterns, stand age and density, as well forest management schemes (e.g. silvicultural treatments, rotation period, clearcut size). Addressing the relationship between hydrologic processes at various spatial and temporal scales is also an important goal.

The current and projected water demand for various uses including agriculture, hydropower generation, tourism and drinking water underline the need to improve land-use and forest management to assure water supply as an important ecosystem service (Lara et al., in press). Preservation of this ecosystem service need to be considered in decision-making regarding land-use planning. In Chile, water consumption by fast-growing species (especially in areas where water is scarce) is not being considered by either private investors when deciding the location of plantations or by the government when allocating forest incentives (Huber et al., 2008). Results presented in this paper indicate the need of a better balance between the area occupied by exotic forest plantations, native forests, agriculture and pasturelands as well as other uses in each watershed. This is necessary in order to increase water yield under a changing climate and at the same time assure timber and agriculture production. Restoration of native forests using the incentives provided by the native forest law approved in 2008 might be an important tool to recover runoff and to increase summer water supply, which is especially critical under a Mediterranean-type climate.

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