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Buffer effects of streamside native forests on water provision in watersheds dominated by exotic forest plantations

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ABSTRACT

The Valdivian rainforest ecoregion in Chile (35°–48°S) has a high conservation priority worldwide. These forests are also keys for social welfare as a result of their supply of timber as well as ecosystem services. Forests in the ecoregion have been extensively converted to fast growing *Pinus radiata* and *Eucalyptus* spp. plantations for timber production promoted by public policies and timber companies. This study describes the results of detailed measurements of hydrology and stream water chemistry in eight small watersheds in south central Chile, subjected to replacement of native temperate rainforest by exotic *Eucalyptus* plantations. In this system, watersheds have streamside buffers of native forest (SNFW) with varying widths. Results indicate that retention of SNFW counteracts hydrologic effects of *Eucalyptus* plantations, which are widely known to reduce water yields. A 1.4% rate of increase of the run-off coefficient for each metre of increase of SNFW was observed. In addition, a decrease in the concentrations of total nitrogen, dissolved inorganic nitrogen (DIN), nitrate-N, and different sized fractions of particulate organic matter were found in streams draining these plantations as a function of increasing SNFW. Streamside buffer widths of 17–22 m for total nitrogen and DIN concentrations and ≥36 m for sediments were required to provide comparable values to reference watersheds (100% native forest). The findings from this study suggest that SNFW may significantly reduce adverse effects from exotic species forestry plantations on water provision in an area of south central Chile where exotic forest plantations are rapidly expanding. Copyright © 2014 John Wiley & Sons, Ltd.

KEY WORDS ecosystem services; nitrogen; riparian zones; water yield; Chile; unpolluted

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INTRODUCTION

Forest vegetation along stream channels provides direct and indirect functions for maintenance of water provision (quality and quantity) (Naiman *et al*., 1998; Lindenmayer and Franklin, 2002; Dosskey *et al*., 2010). These riparian forests have been shown to (1) filter pesticides, sediments, and nutrients that are transported by rainfall, run-off, and groundwater flow processes (Boothroyd *et al*., 2004; Baker *et al*., 2006; Yamada *et al*., 2007); (2) maintain aquatic biota and stream ecosystems through temperature and flow regulation (Arismendi *et al*., 2013); (3) contribute to bank stabilization, large woody debris supply to the channel, and shading effect on streams (Brosofske *et al*., 1997; Lindenmayer and Franklin, 2002; Medina-Vogel *et al*., 2003); and (4) maintain biodiversity, including mammals (e.g. river otter, Medina-Vogel *et al*., 2003), flora (Gregory *et al*., 1991; Naiman *et al*., 1998), and macroinvertebrates (Mancilla *et al*., 2009; Miserendino *et al*., 2011). Forest vegetation is also one of the sources of dissolved organic carbon in run-off (Bishop *et al*., 1994) and a source of organic matter to stream channels (Dosskey *et al*., 2010). In forested headwater streams, riparian vegetation also plays a role in streamflow regulation through rain canopy interception and transpiration (Bond *et al*., 2002; Gribovski *et al*., 2008; Szilágyi *et al*., 2008; Stromberg *et al*., 2010). Nitrogen (N) and phosphorus (P) uptake/production by riparian vegetation regulates export of these nutrients to higher order streams, lakes, and estuaries (Lowrance *et al*., 1984; Peterjohn and Correl 1984; Osborne and Kovacic 1993;

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Compton *et al.*, 2003), thereby reducing ecological stress caused by eutrophication and potential public health risks (Dosskey *et al.*, 2010).

Starting from this theoretical framework, some key questions arise, such as the following: (a) What is the minimum width that a riparian buffer must have to meet the previously mentioned ecosystem functions for different sites, dominant land uses, and vegetation covers in the watershed? (b) Is it possible to estimate the rate at which run-off, nutrient, and sediment discharge change in relation to variations in streamside buffers of native forest (SNFW)? However, ecosystem complexity and spatial and temporal variability impose a challenge for assessing effects of streamside native forests on run-off and nutrient export, which determine water quantity and quality as key ecosystem services in watersheds with different land uses/covers.

Some essential questions regarding the efficiency of SNFW and structure have been scientifically debated for more than 20 years (Osborne and Kovacic, 1993) and represent an important consideration in the frontiers of riparian ecosystem science (Lowrance, 1998). These issues continue being valid for less-studied ecosystems that are scarcely represented in the international literature, such as temperate forests in the Southern Hemisphere (Quinn *et al.*, 2004). They are comparatively unpolluted in relation to North American and European ecosystems, which have been impacted by chronic air pollution (e.g. acid rain, N deposition) (Hedin *et al.*, 1995).

In this context, we investigated the buffer effects of SNFW in watersheds dominated by industrial exotic eucalyptus (*Eucalyptus globulus*) plantations. This is a relevant issue, given that globally exotic plantations keep expanding, whereas primary forests remain in decline and in great need of protection (Paquette and Messier 2010). Many countries rely on large areas of exotic plantations for timber production, such as New Zealand, South Africa, and Chile, with a total of approximately 5.8 million ha (FAO, 2010). Expansion of these industrial plantations, in many cases at the cost of native forests, has increased the challenge of sustaining biodiversity. This results in impacts on ecosystem processes at a landscape scale (Lindenmayer *et al.*, 2011). For example, changes in run-off rate (Little *et al.*, 2009) may occur as a result of conversion of native forests to exotic plantations, which may reduce accessibility to ecosystem services such as water provision (Lara *et al.*, 2009).

Using a set of watersheds dominated by *E. globulus* fast-growing plantations as a study system (where native forests are retained along streamside channels), our predictions were the following: (a) There is a significant positive relationship between the SNFW and run-off, as well as a negative effect on nutrient and sediment discharge in watersheds dominated by exotic plantations; and (b) there is a threshold in SNFW in which these ecosystem functions will show no significant differences compared with watersheds covered completely by native forests. Testing these predictions through the approach presented here is the basis for advancing our understanding of the native forested streamside width required for water provision, an important ecosystem service.

METHODS

General location and site description

The selected study area is located in the Chilean Valdivian rainforest ecoregion listed among those with the highest conservation priority in the world because of high biodiversity, endemism, and threats (e.g. conversion to exotic fast-growing plantations, human-set fires) (Olson and Dinerstein, 1998). Furthermore, this region has not been impacted by chronic air pollution and has been recognized as one of the more pristine areas worldwide with extremely low atmospheric N deposition and nutrient concentrations in stream water (Hedin *et al.*, 1995; Weathers and Likens, 1997; Perakis and Hedin, 2002).

We studied eight experimental forested headwater catchments located within the *Reserva Costera Valdiviana* (RCV) (Figure 1a). This area is a private reserve owned by The Nature Conservancy since 2003 to protect 50 000 ha of coastal rainforest ecosystems and to restore Valdivian rainforests. Between 1993 and 1999, 3000 ha of native forests in this area were clear-cut, burned, and converted to exotic *Eucalyptus* plantations (Farías and Tecklin, 2003; Little and Lara, 2010; Little *et al.*, 2013). At present, the RCV is being developed as a long-term ecosystem research site in the Coastal Range of Southern Chile (Anderson *et al.*, 2012).

The eight watersheds are located at 39°58′S, 73°35′W (Figure 1b). Elevations range from 6 to 445 m a.s.l. At long term (1960–2010), mean annual rainfall near the site (Valdivia city) is 2300 mm under a humid temperate climate with Mediterranean influence. Precipitation occurs mainly as rain between June and August. Summer (January–March) rainfall is limited, representing $<10\%$ of the annual value (Lara *et al.*, 2009; Dirección General de Aguas, unpublished); snowfall is rare. Study watersheds are located in the Coastal Range, which is composed of Paleozoic metamorphic rocks, partially overlaid by Tertiary marine sediments with intermediate slope angles. In the western zone of RCV, outcrops of granitic intrusions from the Cretaceous period occur (Servicio Nacional de Geología y Minería, 1982; Le Roux and Elgueta, 2000; Duhart *et al.*, 2001). Soils have a volcanic origin and are typic Haplohumults (Ultisols; CIREN 2001), with low pH (4.2–4.8), low nutrient contents ($<2\,mg\,kg^{-1}$ for $Ca(OH)_2$-extractable NO_3^- and Olsen-P), with high Al concentration and saturation ($>170\,mg\,kg^{-1}$ and $>44\%$, respectively).

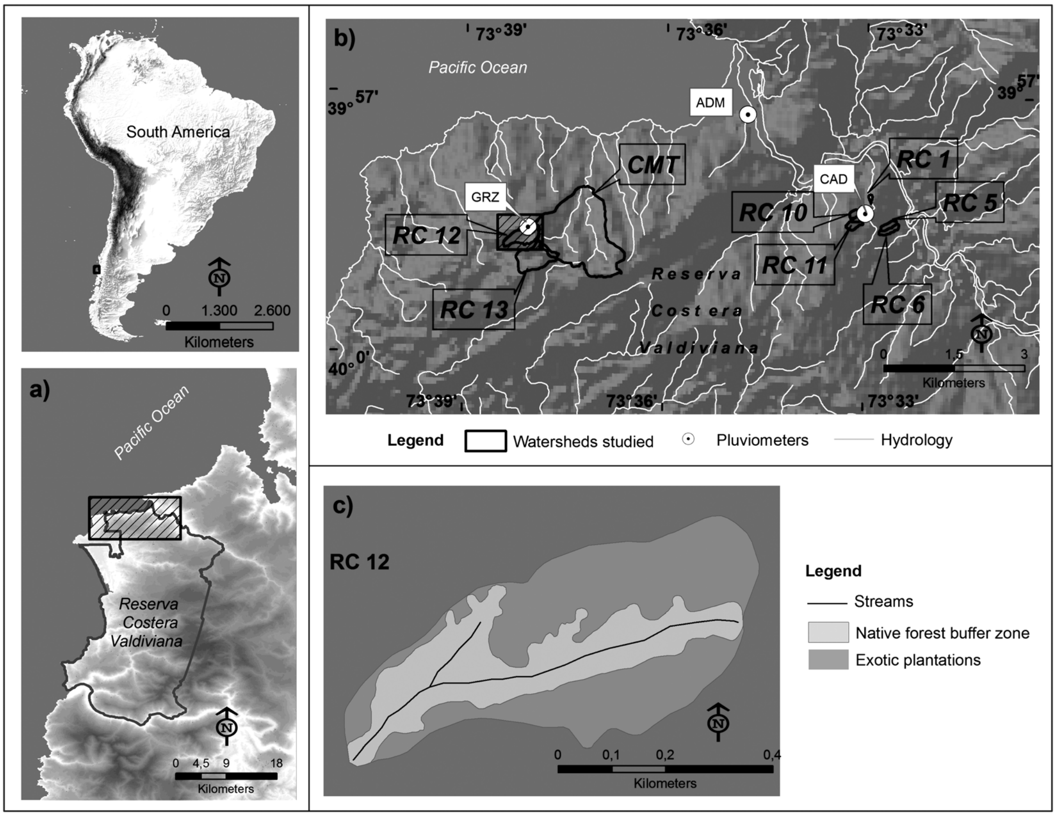

Figure 1. Location map showing the general area on the coast of Southern Chile where the research was conducted (a), sites included for run-off and rainfall monitoring (b), and an example of watershed boundary including streamside native forest (c).

Watershed characteristics, land use, and streamside native forest

Watershed boundaries were mapped and forest land cover was evaluated using natural-color, near-vertical 1:8000–1:10 000 scale aerial photos taken of each watershed. Digital orthophotos were developed using ERDAS1 software (Environmental Systems Research Institute, Inc., USA). Standard photointerpretation and ground truthing techniques were used (Lillesand and Kiefer, 1994). Final maps were developed and incorporated into a geographic information system using Arcview® 3.1 (Environmental Systems Research Institute, Inc., USA).

The watersheds were categorized into two land use/cover types: (a) native forests constituted strips of differing widths along the main stream channel (SNFW) and (b) 13-year-old fast-growing exotic *E. globulus* plantations located in the remainder of the watersheds. Exceptions were watersheds RC13 and CMT, which were almost completely covered by native forests and were considered as references for comparisons of run-off coefficients and water quality parameters that were studied (refer to Table I, Figure 1b). Forest inventories used transects that were perpendicular to the stream in order to characterize SNFW and circular plots of 500 m^2 for land use characterization within the exotic forest plantations (Table I). After this characterization, it was concluded that the streamside native forest had the floristic composition typical of temperate evergreen forests of Southern Chile.

Minimum SNFW was 2.5 m (RC1), and the maximum was 36 m in RC12 (Table I). Areas outside of SNFW were constituted by *E. globulus* plantations, with a density of 1300 to 1700 trees ha^{-1} and a basal area ranging from 27 to 40 $m^2 ha^{-1}$. Eucalypt plantation was the dominant land use in all the watersheds ranging from 61.8% in RC12 to 98.7% in RC1 expressed as the percentage cover over the total watershed (Table I).

Native forest in this area is composed of evergreen broadleaved tree species, mainly *Nothofagus nitida*, *Laureliopsis philippiana*, *Amomyrtus luma*, and *Drimys winteri*; *Chusquea quila* bamboo; and the ferns *Lophosoria quadripinnata* and *Blechnum chilense*. Vegetation cover (tree, shrubs, herbs) is ~100% and includes dense understory dominated by *C. quila* bamboo thickets (4–6 m of height), a typical species associated with unstable sites, such as riparian environments (González *et al.*, 2002).

Rainfall–run-off monitoring and computing

Daily run-off was monitored during five hydrological years (April 2006 to March 2011) (Figure 2), using 90° V-notch weirs and standard procedures (Ward and Trimble, 2004).

Table I. Watershed and streamside native forest characteristics studied in the Reserva Costera Valdiviana.

Watershed characteristics						Streamside forest characteristics			
Watershed code	Area (ha)	Elevation (range) (m a.s.l.)	Mean width[a] (m)	Plantation cover[b] (%)	Basal area of forest plantation $m^2\ ha^{-1}$	Mean width each streamside[c] (m)	Mean slope[c] (%)	Area (ha)	Dominant native species[d]
Plantation with native forest in streamside buffer area									
RC1	0.85	46–88	60	98.7	40.5	2.5	34.4	0.011	NG, BC, LQ
RC5	3.74	6–107	109	76.1	33.7	14.8	46.6	0.895	AL, CQ, LQ
RC6	5.26	6–124	98	69.4	34.5	14.6	45.2	1.610	AL, CQ, DW
RC10	3.43	116–164	164	63.1	36.6	22.5	41.0	1.265	LQ, LR, DW
RC11	4.28	115–195	136	74.8	29.4	17.2	42.7	1.080	AL, CQ, DW
RC12	20.41	283–398	240	61.8	27.1	36.1	—	7.799	NN, LP
Reference watersheds without plantation									
RC13	34.77	305–385	364	0.3	n.d.	151.5[e]	—	34.66	NN, LP
CMT	224.7	50–445	1051	1.5	n.d.	109.7[e]	—	221.3	GA, DW, AL

[a] Measured with seven to eight transects perpendicular through the main stream channel.
[b] Difference from 100% corresponds to native forest.
[c] Measured in the field for 16 transects per watershed.
[d] Main trees: *Amomyrtus luma* (AL), *Drimys winteri* (DW), *Gevuina avellana* (GA), *Laureliopsis philippiana* (LP), *Nothofagus nitida* (NN); shrub: *Chusquea quila* (CQ); ferns: *Blechnum chilense* (BC), *Lophosoria quadripinnata* (LQ); Vine: *Lapageria rosea* (LR); herb: *Nertera granadensis* (NG).
[e] In these cases (watersheds almost completely covered by native forest), the mean width of the streamside was divided by two if only a main stream was present in the watershed, by four if two streams were present, etc. n.d., non-determined.

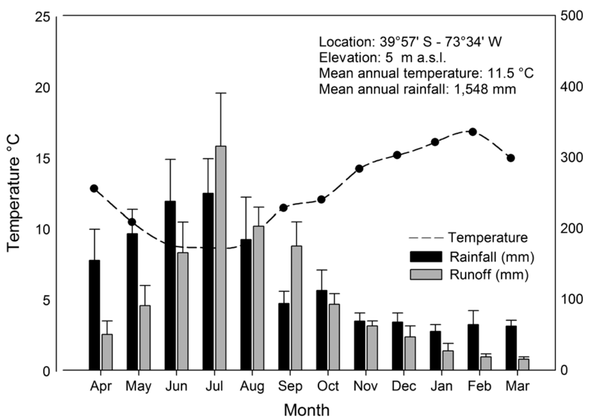

Figure 2. Monthly run-off and rainfall in the studied area. Values correspond to the mean of five hydrological years, and error bars are one standard error. Rainfall data were recorded from three pluviometers, and run-off was recorded in seven watersheds using weirs and data loggers.

From November 2008 to March 2011, streamflow was recorded using HOBO® pressure sensors (U20-001-01, Onset Computer Corporation, USA), using a 15-min resolution. For each watershed, we computed annual run-off coefficients [run-off/precipitation (R/P)] as a function of precipitation, streamflow, and watershed area. Precipitation was recorded using three HOBO® Event H-07-002-04 tipping bucket rain gauges located in three open areas near the watersheds (refer to locations in Figure 1b). Precipitation resolution per event was 0.2 mm. Because our study area is exposed to strong winds coming from the Pacific Ocean, horizontal rain that could not be captured by pluviometers may cause an underestimation in rainfall and, consequently, an overestimation in the run-off coefficients (R/P). However, as the climate regime and the equipment used are the same for all watersheds, any bias in our measurements is a constant factor that should not affect the estimation of the rate of change of R/P as a function of predictor variables.

In order to estimate the annual R/P influenced by the streamside native forest and to separate it from the effect of forest plantations, the R/P residuals were analysed, computed as the difference between observed R/P and that predicted from a linear regression analysis between R/P and the percentage of forest plantation cover:

$$R/P = \mathrm{f}(\%PL) + \varepsilon$$

where R/P is the annual run-off coefficient; *PL* is the plantation cover as a percentage; f is a function; and $\varepsilon\ \alpha\ (0,\ \sigma^2_e)$. Effects apart from forest plantation cover controlling annual run-off coefficient are assumed to be constant, embedded in the coefficients of the function f, and independently distributed. Random effects are represented by ε (Zar, 1999).

Nutrient and sediment concentrations

Dissolved nutrients were sampled every 15 days from July 2008 to October 2009, accumulating a total of 30 samples from each headwater stream. Samples were taken 5 m upstream from the V-notch weir location, from a central point of the stream surface. Total nitrogen (N), NO_3–N, NH_4–N, total phosphorus (P), and PO_4–P were analysed, given their importance for ecosystem processes affecting aquatic productivity and eutrophication (Vitousek *et al.*, 1997). Dissolved N and P are also important indicators of land use effects on water quality, especially in ecosystems with low atmospheric pollution (Oyarzún *et al.*, 2007; Little *et al.*, 2008).

Water samples were collected in plastic bottles that had been previously washed with 5% hydrochloric acid. Bottles were rinsed with 100 ml of water taken from the stream. Bottles were capped and shaken, and the water was discarded. Thereafter, 500 ml samples were collected, the capped bottles were kept at cool temperatures (<5 °C), and the samples were frozen within 10 h of being collected. Samples were chemically analysed at the Limnology Laboratory at Universidad Austral de Chile, following APHA (2005) and Koroleff (1983) procedures. Detection limits were 2 $\mu g\,l^{-1}$ (PO_4^{3-}–P), 5 $\mu g\,l^{-1}$ (total P), 3 $\mu g\,l^{-1}$ (NO_3^-– N and NH_4^+–N), and 20 $\mu g\,l^{-1}$ (total N).

Suspended solids (SS), an indicator of hillslope erosion and sediment delivery to streams, were sampled every 15 days from July 2008 to November 2010 accumulating a total of *ca*. 60 samples in each headwater stream. Measurement was carried out using two different methods in order to identify their short-term and medium-term responses: instantaneous suspended sediment (ISS) and cumulative suspended sediment (CSS). For ISS, three 1000 ml water samples were captured in the same location where nutrient samples were obtained using plastic containers. These samples were filtered in the laboratory through a Gelman Type A 47 mm glass filter, which traps sediments larger than 0.6 μm and were analysed in accordance with standard methods (APHA, 1995). For CSS, we used five fixed cylindrical PVC traps with a height of 10 cm and a height/diameter ratio of 2.85. PVC traps were located on the bed of the V-notch weir in a parallel line to the V section and perpendicular to streamflow, capturing suspended solids during 15 days, corresponding to the sample collection frequency. The lower water flow velocity in the weir allows for the deposition of sediment, which would be more difficult without the installation of the weir. Although this is an artificial manipulation of the system, it has a comparative value because all watersheds were sampled in the same way. The CSS device captures both organic and siliciclastic materials, ranging in size from gravel to clay. This CSS was separated in portions according to grain size [>2 mm (gravel), 2 mm–63 μm (sand), <63 μm (silt-clay)] using standard sieves at the sedimentology laboratory at Universidad Austral de Chile. The organic and inorganic portions were discriminated heating for 20 min or 2 h at 550 °C (for ISS and CSS, respectively) and weighing by difference (for each particle size in the case of CSS) (Folk, 1980). Our approach to capture sediment using cylindrical traps within the V-notch weir was a novel methodology in this kind of study. Captured sediment represents the accumulation over several weeks, whereas ISS was instantaneous. Moreover, the five replicates fit well within the weir and provided good spatial distribution.

Data analysis

We used R/P and the selected water quality variables in order to evaluate the hypothetical buffer function of the SNFW. First, we used simple tools including correlation matrices and variance analysis (ANOVA) in order to detect association between variables and statistical differences between all watersheds. The post-hoc tests used to compare means were the LSD and Tukey, respectively (Sokal and Rohlf, 1995).

Streamflow trends were considered as a function of precipitation trends for different temporal scales, watershed areas, and land use cover types. Correlations between run-off, nutrient, and sediment discharge with SNFW and plantation percentage cover in the entire watershed were conducted independently. Because it is difficult to find watersheds under complete experimental control, statistical control techniques were used, which allowed the isolation of the effects of single variables, while the other variables were mathematically controlled. For example, in order to discern the effect of the SNFW from the influence of watershed size, precipitation, and forest cover, a multiple linear regression was used, in which the stream variables were analysed separately as dependent factors. The independent predictor variables were SNFW, precipitation, and percent of exotic plantation cover in the watershed. In spite of the negative correlation between SNFW and plantation cover, it is not a restriction when the purpose is to evaluate the relative strength of each predictor through beta coefficients (the standardized slopes of the regression models). Original data were transformed using the function $\log_{10}(x+1)$ when the assumptions of parametric methods were not met (i.e. normal distribution of residuals), similar to the procedure followed by Miserendino *et al.* (2011). The distributions of residuals from the linear models were examined. All analyses were performed with Statistica® 6.0 software (Statsoft, Inc., Tulsa, OK, USA).

RESULTS

Rainfall–run-off monitoring

From April 2006 to March 2011, rainfall was 1465, 1623, and 2467 mm (mean values) for the gauges ADM, GRZ, and CAD, respectively (locations shown in Figure 1; inter-annual C.V. = 26.5, 32.0, and 31.9%, respectively). Precipitation is concentrated in the austral winter (July to September), representing 38% of the annual value (Figure 2). After August, rainfall decreases; summer rainfall (January to March) only represents 5 to 15% of the annual value. Monthly run-off showed a significant relationship with precipitation occurring in the same month ($r > 0.6$; $p < 0.05$). More than 75% of annual run-off was concentrated between April through September, decreasing in spring (October–December) and summer (January–March) with 20 and 5% of the total run-off, respectively (Figure 2).

Native forest cover (NF) was positively correlated with the annual R/P ($r = 0.75$), being a linear relationship ($R/P = 0.53 + 0.0045 \times NF$; $R^2 = 0.57$; $p = 0.0012$). Seasonal run-off distribution showed positive correlations with the percentage of NF cover in summer ($r^2 = 0.28$; $p = 0.052$) and fall ($r^2 = 0.35$; $p = 0.025$) periods. However, there was no detectable association between NF cover and R/P in winter or spring (data not shown). Annual R/P was negatively correlated with percent plantation cover ($R^2 = 0.57$; $p < 0.005$; Figure 3a). The highest and lowest values for annual R/P were observed in the reference watershed (RC13) containing 100% cover of native forest and RC1 watershed containing 98.7% plantation cover, respectively. Consequently, in these watersheds as a study system, a positive trend between SNFW and the residual R/P coefficient ($p = 0.054$) was observed (Figure 3b), and an R/P rate of 0.0065 was estimated for each metre of change of SNFW. Using the equations developed in Figures 3a and b, with the minimum value of SNFW and maximum of plantation cover (Table I), the estimated R/P rate of 0.0065 per metre of SNFW represents 1.4% of variation of the total R/P.

Nutrient and sediment concentrations

Mean concentrations of nutrients and suspended solids showed high fluctuations among the watersheds (Table II). The maximum total nitrogen (TN) value of 144.6 $\mu g\,l^{-1}$ was measured in the RC5 watershed, and the minimum of 63.1 $\mu g\,l^{-1}$ was observed in the reference watershed (CMT) (Table II). Dissolved inorganic N (DIN) varied from 15.5 to 63.2 $\mu g\,l^{-1}$ and in some cases corresponded to values close to 50% of TN (Table II). With the exception of CMT, NO_3^-–N represented the dominant form of DIN, with proportions between 50 and 87%. Total N, DIN, and NO_3^-–N (54.8–3.9 $\mu g\,l^{-1}$) concentrations showed significant differences among watersheds. Total P (TP) (7.8–6.3 $\mu g\,l^{-1}$) and phosphate-P (3.4–2.7 $\mu g\,l^{-1}$) concentrations were not different among all watersheds (Table II).

The response of suspended sediment concentration was different between instantaneous (ISS) and cumulative (CSS) sediment sample monitoring. All watersheds showed a positive and significant correlation coefficient between the organic and inorganic fractions ($p < 0.05$). ISSs did not show significant differences among watersheds in the concentration of the organic portion ($F_{6,288} = 1.74$, $p = 0.111$). CSSs were statistically different between watersheds, showing differences in organic and inorganic fractions contained in gravel, sand, and silt-clay. We observed that in RC12 and RC13, the concentration values of CSS were generally higher than in RC11, RC10, RC5, RC6, and RC1 (Table II).

Determining streamside native forest buffer effects

In addition to the SNFW effect on stream water, we also analysed nutrient and suspended sediment concentrations as a function of precipitation for the same day that water samples were collected and for cumulative precipitation 5, 10, and 15 days before collection of the water sample. This analysis was necessary because of the very important role of rainfall on stream properties (Graham 1999, Tesař *et al.*, 2008). We observed that some water nutrient and sediment variables were related to precipitation, with 15 days of cumulative precipitation resulting in the correlation containing significant coefficients ($p < 0.05$) [DIN, $r = 0.32$; NO_3^-–N, $r = 0.34$; particulate organic matter-gravel (POM-gravel), $r = 0.34$; inorganic gravel (IG), $r = 0.24$; POM-sand, $r = 0.51$; inorganic sand, $r = 0.28$; POM-silt/clay, $r = 0.55$; and inorganic silt/clay, $r = 0.38$]. Because watershed areas and CSS concentrations were positively correlated ($p < 0.05$, data not shown), we standardized CSS by watershed area to further investigate the SNFW buffer effect.

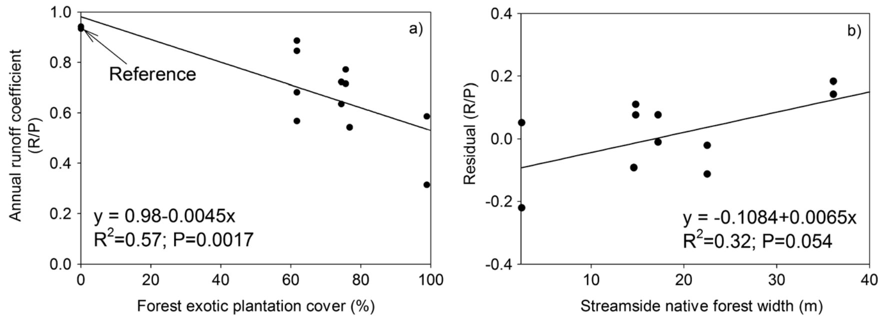

Figure 3. (a) Annual run-off coefficient (R/P) as a function of exotic forest plantation cover in each watershed. Each point (14) represents an annual value for each of the seven watersheds for two years. (b) Residuals of annual run-off coefficient (R/P) as a function of streamside native forest width.

Table II. Mean concentrations for water quality variables.

Watersheds code	RC1	RC5	RC6	RC10	RC11	RC12	RC13	CMT	ANOVA among watersheds
Streamside native forest width (m, each side)	2.5	14.8	14.6	22.5	17.2	36.1	151	109	
Nutrient concentration ($\mu g\,l^{-1}$)									
Total nitrogen	112.7^{a} (11.9)	144.6^{b} (21.2)	122.2^{ab} (14.3)	76.4^{c} (9.3)	74.3^{c} (7.5)	n.d.	n.d.	63.1^{c} (7.4)	$F_{5,148} = 12.4$ $p < 0.001$
NO_3^{-}–N	35.1^{a} (8.9)	51.2^{b} (7.1)	54.8^{b} (6.1)	12.5^{a} (2.6)	16.1^{a} (2.3)	n.d.	n.d.	3.9^{c} (0.2)	$F_{5,148} = 18.1$ $p < 0.001$
DIN	46.2^{a} (9.2)	$61.\ 6^{b}$ (7.2)	63.2^{b} (6.1)	23.9^{ac} (3.0)	26.7^{a} (2.5)	n.d.	n.d.	15.5^{c} (2.4)	$F_{5,148} = 13.3$ $p < 0.001$
Total P	6.3 (0.7)	7.4 (8.9)	6.4 (0.7)	6.5 (0.5)	7.8 (1.3)	n.d.	n.d.	7.8 (1.0)	$F_{5,148} = 0.81$ $p = 0.5453$
PO_4^{3-}–P	2.8 (0.1)	2.7 (0.1)	3.2 (0.3)	2.7 (0.1)	3.4 (0.5)	n.d.	n.d.	3.0 (0.3)	$F_{5,148} = 0.79$ $p = 0.5521$
ISS^{Ψ} ($mg\,l^{-1}$)									
Inorganic	1.98^{bc} (0.53)	1.43^{bc} (0.31)	1.27^{b} (0.40)	2.01^{b} (0.89)	1.21^{bc} (0.28)	2.71^{a} (0.30)	1.63^{c} (0.27)	n.d.	$F_{6,288} = 5.88$ $p < 0.001$
Organic	2.68 (0.63)	2.76 (0.49)	2.38 (0.56)	3.81 (1.53)	1.61 (0.25)	2.14 (0.16)	1.39 (0.19)	n.d.	$F_{6,288} = 1.74$ $p = 0.1113$
CSS^{Π} ($g\,l^{-1} 15\,days^{-1}$)									
Inorganic gravel	0.00^{a} (0.00)	0.03^{a} (0.01)	0.04^{a} (0.01)	0.08^{a} (0.04)	0.16^{a} (0.06)	1.29^{a} (0.78)	31.49^{b} (10.82)	n.d.	$F_{6,221} = 10.71$ $p < 0.001$
$POM^{£}$-gravel	0.01^{a} (0.01)	0.48^{ac} (0.16)	0.32^{ac} (0.11)	1.04^{abc} (0.68)	1.31^{bc} (0.54)	1.90^{b} (0.58)	1.69^{b} (0.43)	n.d.	$F_{6,221} = 4.10$ $p < 0.001$
Inorganic sand	0.21^{a} (0.06)	2.45^{a} (0.72)	3.55^{a} (0.79)	1.6^{a} (0.50)	6.7^{a} (2.02)	61.93^{b} (21.88)	131.53^{b} (40.66)	n.d.	$F_{6,221} = 15.32$ $p < 0.001$
POM-sand	0.24^{a} (0.07)	2.52^{bd} (0.53)	3.34^{bd} (0.61)	1.98^{b} (0.51)	2.41^{bd} (0.56)	9.97^{c} (1.87)	3.97^{d} (0.67)	n.d.	$F_{6,221} = 16.60$ $p < 0.001$
Inorganic silt/clay	0.18^{a} (0.03)	0.85^{ac} (0.19)	1.18^{bc} (0.23)	0.57^{ac} (0.15)	1.28^{bc} (0.25)	6.27^{d} (0.90)	3.08^{e} (0.48)	n.d.	$F_{6,217} = 31.95$ $p < 0.001$
POM-silt/clay	0.15^{a} (0.02)	0.64^{b} (0.12)	0.73^{b} (0.12)	0.41^{ab} (0.11)	0.59^{b} (0.11)	1.55^{c} (0.21)	0.70^{b} (0.11)	n.d.	$F_{6,217} = 12.84$ $p < 0.001$

The standard error is presented in parentheses (SE). Different superscripts indicate differences at $p < 0.05$ (LSD test), analyzing each variable separately.
n.d., not determined; DIN, dissolved inorganic nitrogen; ISS^{Ψ}, instantaneous suspended sediment and CSS^{Π}, cumulative suspended sediment. $^{£}$POM, particulate organic matter of a specific granulometry.
CMT is the reference watershed for water nutrients, while RC13 is the reference for sediments and run-off coefficient.

Multiple regression models showed that some water nutrient and sediment variables were a function of SNFW, cumulative precipitation, and exotic plantation cover (Table III). Total N, DIN, and NO_3^-–N produced statistically significant results, with SNFW having a statistically significant negative beta coefficient for DIN and NO_3^-–N, indicating that increased SNFW is associated with reduced concentrations of these nutrients in stream water. Concentrations of DIN and NO_3^-–N also increased with increasing cumulative precipitation (Table III). Moreover, decreased NO_3^-–N concentrations were significantly associated with increased exotic plantation cover (Table III). No effects were detected for SNFW, cumulative precipitation, or percentage of exotic plantation cover on TP or phosphate-P.

It was observed that precipitation was significantly associated with all organic and inorganic particulate fractions (Table III). Multiple regression models showed that SNFW had a significantly negative association with POM-gravel and POM-sand and a significantly positive association with IG. Percentage of exotic plantation cover was negatively associated only with POM-sand. There was also a significant positive association between SNFW and the inorganic fraction of ISS ($p = 0.054$). It is noteworthy that the SNFW effect was stronger than the precipitation effect for nutrients, whereas precipitation generally had a stronger impact in the case of CSS (Table III). Despite the fact that multiple regressions are linear models, the logarithmic transformations determined that nutrient/sediment relationships with SNFW are non-linear. Therefore, a two-parameter rate of change relating both variables could not be derived.

The CMT reference watershed had significantly lower DIN and NO_3^-–N concentration values than most watersheds (LSD test, $p < 0.05$), (Table II). Post-hoc tests showed that watersheds with SNFW ≥17 m were comparable with the reference watershed (CMT) for total N. Cumulative sediment components POM-sand and POM-silt/clay generally decreased with respect to increasing SNFW (Figure 4), and in the range of SNFWs studied (2.5–36 m), only POM-silt/clay was comparable with the reference watershed (RC13) at 36 m. In general, Figure 4 also confirms that these relationships are not linear.

Table III. Beta coefficients, partial correlations, adjusted coefficient of multiple determinations (R^2adj.), Fisher (F) and probability (p) values of the multiple regression models among the different nutrient forms as a function of streamside forest width, cumulative precipitation, and exotic plantation cover.

Water quality variables	SNFW£ β (partial r)	Precipitation previous 15 days β (partial r)	% exotic plantation β (partial r)	R^2 (adj.)	F	Degrees of freedom	p
Nutrient concentration ($\mu g\, l^{-1}$)							
Total N	−0.31 (−0.08)	0.11 (0.12)	−0.01 (−0.00)	0.09	5.97	3,150	0.0007
NO_3^-–N	−1.06* (−0.31*)	0.32* (0.36*)	−0.68* (−0.20*)	0.29	21.88	3,150	<0.0001
DIN	−0.86* (−0.24*)	0.31*) (0.33*)	−0.53 (−0.15)	0.22	15.52	3,150	<0.0001
Total P	0.10 (0.03)	−0.11 (−0.11)	−0.02 (−0.00)	0.01	1.41	3,150	0.2408
PO_4^{3-}–P	−0.14 (−0.04)	−0.15 (−0.16)	−0.17 (−0.04)	0.01	1.33	3,150	0.2652
CSS$^{\Pi}$ ($g\, l^{-1}\, ha^{-1}$ 15 days^{-1})							
Inorganic gravel	0.66* (0.21*)	0.29* (0.32*)	0.20 (0.06)	0.29	29.46	3,209	<0.0001
POM$^{\varphi}$-gravel	−0.45* (−0.14*)	0.45* (0.45*)	−0.42 (−0.13)	0.21	19.43	3,209	<0.0001
Inorganic sand	0.21 (0.07)	0.54* (0.57*)	−0.11 (−0.04)	0.38	44.04	3,209	<0.0001
POM-sand	−0.82*(−0.31*)	0.68* (0.71*)	−0.66* (−0.26*)	0.54	83.52	3,209	<0.0001
Inorganic silt/clay	−0.25 (−0.10)	0.68* (0.69*)	−0.09 (−0.03)	0.49	68.49	3,206	<0.0001
POM-silt/clay	−0.01 (−0.00)	0.62* (0.66*)	0.30 (0.11)	0.48	65.39	3,206	<0.0001
ISS$^{\Psi}$ ($mg\, l^{-1}$)							
Inorganic	0.41* (0.12*)	0.09 (0.09)	0.32 (0.09)	0.02	2.57	3,276	0.0545
Organic	−0.06 (−0.02)	0.05 (0.05)	0.02 (0.01)	−0.00	0.76	3,276	0.5146

All dependent variables were logarithmically transformed.
SNFW£, streamside native forest width; CSS$^{\Pi}$, cumulative suspended solids; ISS$^{\Psi}$, instantaneous suspended solids; POM$^{\varphi}$, particulate organic matter of a specific granulometry. Organic sand and organic silt/clay portions included detritus.
*Beta coefficients and partial correlations that are significant at $p < 0.05$.

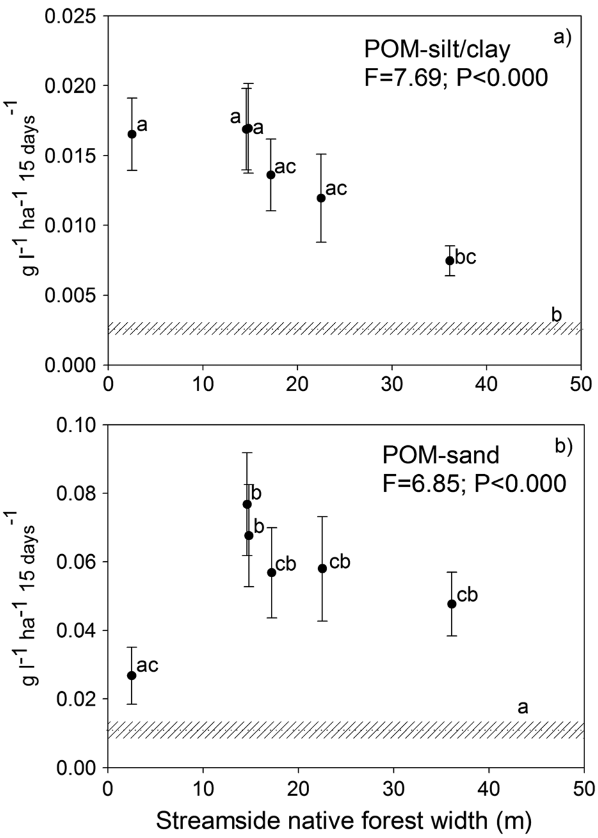

Figure 4. Scatterplots between streamside native forest width and stream water quality. Cumulative precipitation for the previous 15 days was the covariable for ANCOVA analyses. Hatched lines correspond to mean values ± one standard error for the reference watershed (RC13). Distinct superscripts indicate differences among means at $p < 0.05$ (Tukey test). POM, particulate organic matter.

DISCUSSION

Our results show an increase in water supply and a decrease in some nutrient and sediment concentrations in streamflow associated with native forest buffer width in watersheds dominated by exotic *Eucalyptus* plantations. This buffer effect supports our prediction regarding a positive relationship between the SNFW and run-off, as well as a negative effect on nutrient and sediment discharge in watersheds dominated by exotic plantations. We explored this issue in six watersheds dominated by industrial fast-growing exotic forest plantations considering reference watersheds completely covered by native forest. With the exception of the reference watersheds, we did not find watersheds with >40% of native forest cover as a percentage of total surface area. This is a typical limitation in this type of studies, in which it is desirable to have a wide range of situations along a gradient. Furthermore, biophysical variables such as watershed size, climate, and soil condition need to be controlled. Similar restrictions were reported by Lara *et al.* (2009), who evaluated the effect of native forests and exotic plantations on water yield in watersheds where the latter did not cover >60% of the watershed area.

Streamside native forest width and run-off

Important interactions may exist between forest cover and run-off. In this study system, both *E. globulus* plantations and native forests had their greatest relative impact on run-off. It was determined that R/P ratio in each study watershed was significantly influenced by both the percentage of exotic forest plantations over the entire watershed and SNFW. A method was developed to distinguish these two factors as independent drivers; after removing the effect of the exotic forest plantation cover on R/P, a positive trend was related to the increase in SNFW and a rate of 1.4% of increase in R/P for each metre of SNFW increase was estimated. Previous studies performed in other watersheds located in the Valdivian Coastal Range report an increase of summer run-off of 14% for every 10% increase in native forest cover (Lara *et al.*, 2009). Our results are also well-supported by studies reporting effect of plantations with high evapotranspiration rates on run-off in different countries (Calder, 1992; Farley *et al.*, 2005; Scott and Prinsloo, 2008) and for exotic forest plantations in Chile with similar canopy cover, age and development stage (Huber *et al.*, 2008), and their effect on run-off reduction (Little *et al.*, 2009). Nevertheless, the significance value of the relationship that we found between SNFW and the residual of R/P ($R^2 = 0.32$, $p = 0.054$) indicates that this result should be interpreted with caution. The non-linear effects of different land uses, forest structure, climate condition, and other factors on run-off processes that have previously been described (e.g. Farley *et al.*, 2005) might partially explain these limitations.

Streamside native forest width and nutrient and sediment concentrations

Numerous studies have shown that variations in nutrient and sediment concentration and export in stream water are a function of changes in forest land cover, watershed properties, and precipitation (e.g. Graham, 1999; Strayer *et al.*, 2003; Cuevas *et al.*, 2006; Little *et al.*, 2008). The importance of forest stream buffers to conserve stream water quality has been recognized in North America and Europe for several decades, but much of the work on dissolved N and forest buffers has been performed in N-saturated systems (Osborne and Kovacic, 1993). Fewer studies documenting similar ecosystem function of forest buffers have been carried out in the tropics (e.g. Heartsill-Scalley and Aide, 2003; Meli *et al.*, 2013). In pristine native forests in Southern Chile, nutrient exports are extremely low compared with some places of the Northern Hemisphere (Perakis and Hedin, 2002), characterized by chronic deposition of nutrients and pollutants. Studies performed in Southern Chile that consider the land use in the entire watershed dominated by forest exotic plantations indicate a retention of an additional 4% for total N and 31% for total P for every 10% increase in native forest cover; however, the effect of native forest was not explicitly related to streamside native forest (Oyarzún *et al.*, 2007). \Our results suggest that SNFW plays an important role in the regulation of stream water DIN, particularly NO_3^-–N, in low-nutrient discharge conditions. Trends described for the Northern Hemisphere related to reduction of NO_3^-–N concentration and sediment from forest uplands to riparian environment (i.e. ~15 to ~1 mg l^{-1} total N for overland flow, Osborne and Kovacic, 1993), can be compared with the effect of SNFW on the present study system. However, magnitudes of the stream variables are different between systems.

The reference watersheds in this study, dominated by native forests, showed that total N, DIN, and NO_3^-–N concentrations in stream water were lower compared with watersheds with exotic forest plantations, even though the latter sites included streamside native forest comprising buffers as wide as 36 m. However, multiple regression analysis indicated that NO_3^-–N concentration in stream water also was negatively associated with the proportion of exotic plantations cover. This negative association between exotic plantation cover and NO_3^-–N concentration in stream water is supported by documentation of high NO_3^-–N demand of young, fast-growing plantations (Schlatter, 1997; Schlatter *et al.*, 2001), especially under the low nutrient status typical of Ultisols. Furthermore, the inverse correlation that we observed between SNFW and N concentrations in stream water suggests the importance of the native forest buffer for denitrification processes such as those reported by Yamada *et al.* (2007), supported by the increased soil moisture accumulation in riparian strips (Naiman *et al.*, 1998), and increased N absorption under native forest cover (Lowrance, 1998). More recently, nitrate immobilization by soil microbes as well as intensified conversion to ammonium (dissimilatory nitrate reduction to ammonium, DNRA) has been reported in riparian zones (Dhondt *et al.*, 2006). (Cuevas *et al.* 2014; Huygens *et al.*, 2008), recently showed that groundwater nitrate-N decreases from 120 to 5 µg l^{-1} in a 75 m transect going from grasslands to a riparian forest.

No significant differences between watersheds with exotic forests versus reference native forests for total P or phosphate in stream water were detected in our study. Phosphorus is strongly adsorbed by volcanic soils (Morgan, 1997), such as those found in our study area, and this may explain the lack of effect of SNFW on P.

For most of our study watersheds, no significant differences were found in inorganic or organic ISS concentrations among SNFWs. Conversely, significant negative effects of SNFW on CSS were identified. Variations in ISS among watersheds may not have been detected because the method (i.e. frequency interval of grab samples) was probably not efficient for signal detection in small watersheds, in which rapid movement of suspended sediments following intense precipitation and resultant increase in ISS concentration can be detected only during the limited hours after peak flow following storms (Rodgers *et al.*, 2011). Nevertheless, CSS traps were able to integrate these effects and to detect the combined signal of precipitation and SNFW as a buffer. Use of sediment traps to capture particles has been applied, tested, and discussed in a series of studies in lakes, estuaries, and marine environments to determine rates of sedimentation of particles, pollen, plankton, and nutrients (Kirchner, 1975; Hardgrave and Burns, 1979; Blomqvist and Hakanson, 1981; Lorenzen *et al.*, 1981; Bale, 1998).

Sediment traps can underestimate or overestimate particle fluxes considerably; however, cylindrical traps are more reliable than the angular and asymmetrical ones, and to avoid re-suspension of the trap content, the height/diameter ratio must be >3 (Blomqvist, 1981; Blomqvist and Hakanson, 1981; Valeur and Pejrup, 1998). Despite this limitation, the approach used here has a comparative value, because all watersheds were sampled with the same devices placed into each weir bottom.

Streamside native forest width for watershed management

Water provision (quantity and quality) has been reported as an important ecosystem service provided by remainder native forests in watersheds dominated by intensively managed exotic tree plantations (Lara *et al.*, 2009; Oyarzún *et al.*, 2007). At an international level, the most commonly recommended widths for stream buffers to maintain water quality in forested areas are between 5 to 30 m, but it depends on specific site conditions and ecosystem function of interest (Barling and Moore, 1994; Hawes and Smith, 2005). A review presented by Osborne and Kovacic (1993) summarized that 9–45 m of vegetated buffer strips maintain stream temperature and removed a substantial portion of sediments in overland flow from a variety of disturbances and geographic locations in North America. Our results show that dissolved N variables stabilized at 17–22 m SNFW, resulting in concentrations comparable with those observed in the reference watersheds. This result corroborates expectations for a threshold in which these ecosystem functions will show no significant differences compared with watersheds covered completely by native forests. However, organic sediment concentrations indicate that for the maintenance of water nutrient and sediment loads comparable with standard reference levels in forested watersheds of Southern Chile (i.e. near pristine streams), SNFW should be ≥36 m on each side of a stream. This recommendation is coherent with

NO_3^-–N behaviour, because this variable was not comparable with the reference watershed, at least under 17–22 m SNFW.

The system analysed in the present study showed that an increase of SNFW is efficient to increase R/P, at the same time Nitrogen and some types of sediments are removed. While there is variation in different forested regions, generalizations regarding the effects of SNFW on run-off should be useful for planning forest management to provide multiple goods and ecosystem services (Paquette and Messier, 2010).

The importance of this approach and methods for identifying widths, type, and structure of vegetation to maintain adequate flow and nutrients in industrial forest landscapes are the basis for sustainable timber production. These results have policy implications regarding watershed management schemes that point towards the combined and compatible production of timber and water provision. Large-scale streamside native forest protection and restoration programs, such as retaining native streamside buffers with adequate widths to meet management objectives, are strongly associated with maintaining ecosystem services providing abundant water of high quality.

Results from this study are valid for watersheds in which the land use cover beyond the streamside buffer strips is exotic *Eucalyptus* plantations in a closed-canopy development stage, which is a typical forested industrial landscape in Southern Chile and other countries with intensive industrial forest land use. Additional studies are needed to improve the understanding of the non-linear effects and other factors that might be limiting the percentage of variance explained ($R^2 = 0.32$) of the effect of the SNFW on R/P after removing the effect of exotic forest plantation cover and to understand the complexity of this relationship. Future research should also assess the effects of native forest buffer widths when harvesting exotic forest plantations of different age and density classes, as well as the potential to increase run-off through the restoration of streamside native forests.

To our knowledge, this is the first study to quantify these effects under low non-point pollution and little anthropogenic deposition. This study describes the results of detailed measurements of hydrology and stream water chemistry in eight headwater catchments in south central Chile, subjected to replacement of temperate rainforest and establishment of *Eucalyptus* plantations. It represents a baseline for future comparative studies in other countries for watersheds dominated by industrial planted forests.

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