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Hydric regime in Mediterranean freshwater forested wetlands and their relationship with native and non-native forest cover

Francisco Correa-Araneda[1,2,*], María Elisa Díaz[2], Manuel Olguín[3], Francisco Encina-Montoya[4], Francisco Gomez-Caponni[5] and Ricardo Figueroa[2]

[1] Department of Zoology, Faculty of Natural and Oceanographic Sciences, University of Concepción, P.O. Box 160-C, Concepción, Chile.
[2] Department of Aquatic Systems, Faculty of Environmental Sciences and Environmental Sciences Center EULA-Chile, University of Concepción, P.O. Box 160-C, Concepción, Chile.
[3] Environmental Department, Aquaculture Research Division, Fisheries Development Institute, Puerto Montt, Chile.
[4] Ecotoxicology and Environmental Monitoring Laboratory, Environmental Sciences School, Faculty of Natural Resources, Catholic University of Temuco. P.O. Box 15-D, Temuco, Chile.
[5] Institute of Environmental Sciences and Evolution, Science Faculty, Austral University of Chile, Valdivia, Chile.

* Corresponding author: franciscocorrea@udec.cl

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ABSTRACT

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In different Mediterranean zones (MZ) of the world, extensive areas of native forests have been replaced by forest plantations composed of non-native species that exceeded 264 million ha globally in 2010. In Chile, 80% of these plantations are distributed in the same zone where forested wetlands are dominant. Non-native forest plantations are inversely related to the quantity, distribution and availability of water resources. In this study, the characteristics of the water regime of the forested wetlands were assessed, and their relationship to non-native forest plantations and native forest cover was established. The results indicated that most forested wetlands present a significant decrease in both depth and water volume in the summer ($p < 0.05$), which is related to a decrease in rainfall. However, those wetlands with a higher percentage of non-native forest plantations in their basins (> 18%) presented a temporary water regime, unlike those in which native forest dominated (>21%), which presented a permanent water regime. Strong negative relations were registered ($p < 0.05$) among the percentage of forest plantation and water availability, minimum depth and minimum water volume. On the other hand, the percentage of native forest was positively related with the same variables ($p < 0.05$). This study presents a clear relationship between forest activity and water availability, especially during the summer, when the normal decline of the water resources in the wetlands worsens.

Key words: Non-native forest, hydric regime, forested wetlands, native forest, mediterranean zone.

RESUMEN

Régimen hídrico en humedales boscosos Mediterráneos de agua dulce y su relación con coberturas boscosas nativas y exóticas

En diferentes zonas Mediterráneas (MZ) del mundo, extensas áreas de bosque nativo han sido reemplazadas por plantaciones forestales de especies exóticas que superaron las 246 millones de hectáreas en el 2010. En Chile cerca del 80% de estas plantaciones se distribuyen en la misma zona donde dominan los humedales boscosos. Se ha descrito que las plantaciones forestales exóticas están inversamente relacionadas con la cantidad, distribución y disponibilidad de los recursos hídricos. En este estudio se evaluaron las características del régimen hídrico de humedales boscosos y se estableció su relación con las coberturas de plantación forestal exótica y bosque nativo. Los resultados indican que la mayoría de los humedales presentaron una disminución significativa en la profundidad y volumen de agua en época estival ($p < 0.05$), relacionada con igual disminución en las precipitaciones. Sin embargo, aquellos con un mayor porcentaje de plantaciones forestales en sus cuencas (>18%) presentaron un régimen hídrico temporal, a diferencia de aquéllos donde dominó el bosque nativo (>21%)

que presentaron un régimen hídrico permanente. Se registraron fuertes relaciones negativas ($p < 0.05$) entre el porcentaje de uso de suelo forestal exótico y disponibilidad hídrica, profundidad mínima y volumen mínimo de agua. En tanto, el porcentaje de bosque nativo se relacionó positivamente ($p < 0.05$) con la disponibilidad hídrica, volumen mínimo, profundidad media y profundidad mínima. Este estudio muestra una clara relación entre la actividad forestal y la disponibilidad de agua, sobre todo en época estival, donde se agudiza el normal descenso de los recursos hídricos.

Palabras clave: *Plantaciones forestales exóticas, régimen hídrico, humedales boscosos, bosque nativo, zona Mediterránea.*

INTRODUCTION

Changes in land cover are considered important variables in the hydrological cycle (Farley *et al.*, 2005; Huber *et al.*, 2008; Lara *et al.*, 2009). Forest plantations with non-native species (*e.g.*, *Pinus* spp., *Eucalyptus* spp.), in comparison with native forests or pasture cover, can produce a number of effects on terrestrial ecosystems because they are inversely related to the quantity, distribution and availability of water resources (Echeverría *et al.*, 2007). Among the most important effects, an increase in evaporation can be mentioned (Farley *et al.*, 2005; Jackson *et al.*, 2005; Scott & Prinsloo, 2008). This is directly related to an increase in the interception of rainfall (Huber & Trecaman, 2000). In addition, an increase in transpiration has also been reported and is related to the root uptake of these species, which tends to cover a larger area and reach a greater depth (Huber & Trecaman, 2000; Jackson *et al.*, 2005; Scott & Prinsloo, 2008). The root uptake causes a reduction in the surface runoff, a reduction in the groundwater and a lower aquifer recharge (Hofstede, 2001).

Freshwater ecosystems are susceptible to direct effects of forest activity because the combination of the described processes (*e.g.*, transpiration, interception) largely determines the contribution of water to these ecosystems (Swank & Douglas, 1974). This is particularly relevant in MZ because the absence of rainfall during the summer (Di Castri & Hajek, 1976; Barry & Chorley, 1985) implies that the groundwater reserves of the surrounding terrestrial ecosystems would be the sources of the main contributions in order to maintain a permanent water regime (Ward & Trimble, 2004; Zimmermann *et al.*, 2006).

In different MZ of the world (*e.g.*, Iberian Peninsula, Chile, California), extensive areas of native forests have been replaced by forest plantations composed of non-native species. These species, due to their fast growth rate, are very profitable for the pulp and paper industry (Ferreira *et al.*, 2006). Globally, forest plantations have increased from 178.3 million ha in 1994 to 264 million in 2010, and it is expected that forest plantations will exceed 300 million ha by 2020 (FAO, 2010). In Portugal, 21% of the forested area is occupied by non-native plantations (DGF, 2005), while in Spain, they represent more than 12% of the total forest cover (MMA, 2003) and in Chile, it has been reported that non-native plantations cover more than 17% of the forest in the country (CONAF, 2011).

In Chile, in the same zone in which forest plantations have been established (MZ), a large amount of unique forested wetlands are also distributed. These wetlands are dominated by native arboreal species from the Myrtaceae family, including the following genera: *Myrceugenia*, *Blepharocalyx*, *Luma* and *Tepualia* (Correa-Araneda *et al.*, 2011, 2012). These environments have been recently studied from a limnological perspective (*e.g.*, Correa-Araneda *et al.*, 2011), although their functioning is still unknown (Correa-Araneda *et al.*, 2012) and only potential effects of forest activity on the water resource have been proposed, noting that they can produce changes in the variables that determine the hydroperiod (Echeverría *et al.*, 2007; Angeler *et al.*, 2010). These are the main factors that determine the functioning of this type of ecosystems, and we can infer that there is a more extreme and prolonged summer drought by the increased water consumption by exotic forest plantations in the studied watersheds. On this basis, this study assessed the characteristics of the water regime of forested wetlands with the aim of establishing their relationship with non-native forest plantations and native forest cover.

METHODOLOGY

Study area

The study area is located in southern Chile (Araucanía Region: 37°-40° S), where the exotic forest activity is more intensive (80%) (CONAF, 2011), and there is a greater concentration of forested wetlands (Correa-Araneda *et al*., 2011). The predominant climate in this zone is the wet-Mediterranean type, characterized by dry summers and wet winters with annual precipitation in the range of 1200 mm to 1600 mm. Summer temperatures fluctuate between 14 and 23 °C, and winter temperatures fluctuate between 7 and 13 °C (Di Castri & Hajek, 1976, DGA, 2013).

Site selection was based on a classification obtained by a modification of the methodology proposed by the Water Framework Directive from the European Union (EC, 2000). This methodology uses a series of edaphological, climatic and morphological variables at the basin scale to identify statistically similar groups of wetlands, from which the formation of four types of wetlands is indicated. The group with the highest number of wetlands was chosen (n = 12), whose edaphic characteristics (silt-loam) and morphology (low slope) would indicate the presence of permanent hydric regimes (Fuentes-Junco, 2004); those wetlands formed an intervention gradient in relation to land cover with non-native forest plantations and native forest conservation were also identified.

The percentage of land cover was determined by means of the delimitation of the basins that drain towards their respective wetlands using a digital elevation model (DEM) based on topographic maps (1:25 000) prepared by the Chilean Military Geographic Institute (IGM, 1968). In each basin, the surface covered by forest plantations and native forests was calculated by means

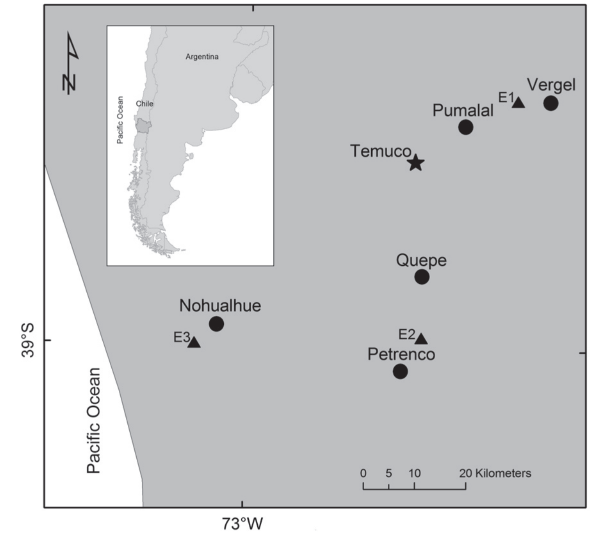

Figure 1. Location of the studied forested wetlands (●; Vergel, Pumalal, Nohualhue, Quepe, Petrenco), meteorological stations used for precipitation data (▲; E1, E2, E3) and the mayor city (★). *Ubicación de los humedales boscosos estudiados (●; Vergel, Pumalal, Nohualhue, Quepe, Petrenco) y las estaciones meteorológicas utilizadas (▲; E1, E2, E3).*

of the Chilean Vegetation Census (CONAF-CONAMA-BIRF, 2007). Finally, wetlands with a range of non-native cover of 3-25% and a range of native forest of 6-36% were selected (Table 1), corresponding to the highest and lowest percentages of both soil covers registered in the identified wetlands. Others soil covers are detailed in Table 1. All analyses were carried out using ArcGis 9.3 software.

Sampling methods

The study was carried out by means of sampling from April 2011 to March 2012. Daily rainfall (mm) data were used in order to determine the pattern of rainfall in the zone under study and its relationship with the water variables of the wetlands. These data were collected from the databases of the Chilean General Water Commission (DGA, 2013). The selected stations (E1: 38°31′07″S-72°27′08″W; E2: 38°57′33″S-72°36′30″W; E3: 39°00′52″S-73°04′58″W) are the closest to the studied wetlands (Fig. 1).

The hydric regime and temporal variation of water level (cm) were determined by means

Table 1. Characterisation of forested wetlands based on geographic, morphometric, climatic, hydrological and land cover aspects. CS = Coordinate South; CW = Coordinate West; WS = Wetland surface (Ha); BS = Basin surface (Ha); Elev. = Elevation (m.a.s.l.); AP = Annual precipitation (mm); HP = Hydroperiod (days); MV = Maximum volume (m^3); MinV = Minimum volume (m^3); MaxD = Maximum depth (cm); MinD = Minimum depth (cm); FL = Flooded (% year); SAT = Saturated (% year); NNP = Non-native plantation (%); NF = Native forest (%); AGR = Agricultural (%); NDS = Native dominant species; NNDS = Non-native dominant species. *Caracterización de los humedales boscosos basada en aspectos geográficos, morfométricos, climáticos, hidrológicos y de cobertura de suelo. CS = Coordenada Sur; CW = Coordenada Este; WS = Superficie Humedal (Ha); BS = Superficie cuenca (Ha); Elev. = Elevación (m.s.n.m.); AP = Precipitación anual (mm); HP = Hidroperíodo (días); MV = Volumen máximo (m^3); MinV = Volumen mínimo (m^3); MaxD = Profundidad máxima (cm); MinD = Profundidad mínima (cm); FL = Inundado (% año); SAT = Saturado (% año); NNP = Plantación forestal exótica (%); NF = Bosque nativo (%); AGR = Agricultura (%); NDS = Especies nativas dominantes; NNDS = Especies exóticas dominantes.*

	Petrenco	Quepe	Nohualhue	Vergel	Pumalal
CS	39°9′20.29″	38°52′26.36″	38°58′0.72″	38°33′43.64″	38°35′55.34″
CW	72°39′45.27″	72°36′56.02″	73°4′34.59″	72°19′27.93″	72°30′59.57″
WS	269	346	107	138	192
BS	3630	1642	2778	2840	3344
Elev.	94	95	26	182	158
eAP	1186	1186	1250	1229	1229
WR	Permanent	Permanent	Permanent	Temporary	Temporary
HP	365	365	365	199	215
MaxV	5 826 864	4 090 151	1 216 511	2 463 471	4 800 075
MinV	4 853 046	2 822 733	592 758	0	0
MaxD	69.7	59.9	83.5	82.4	51.5
MinD	30.8	15.7	22.1	0	0
FL	100	100	100	54.5	58.9
SAT	0	0	0	45.5	41.1
NNP	3	4	8	18	25
NF	36	27	21	17	6
AGR	54	48	67	60	63
Other	10	25	12	23	31
NDS	*Nothofagus dombeyi, Drimys winteri, Lomatia hirsuta, L. ferriginea, Maytenus boaria, Sophora microphylla, Amomyrtus luma*	*Nothofagus obliqua*	*Nothofagus obliqua, Aristotelia chilensis,*	*Nothofagus obliqua, Persea lingue*	*Nothofagus obliqua*
NNDS	*Eucalyptus globulus*	*Eucalyptus nitens, Pinus radiata*	*Eucalyptus globulus, Pinus radiata*	*Eucalyptus nitens*	*Eucalyptus globulus, Pinus radiata*

of autonomous water level sensors (HOBO U20-001-02) with a measurement range of 0 to 30 m in depth. These sensors were calibrated for measurement intervals of one hour over the course of one year (April 2011-March 2012). From these analyses, the maximum depth (cm), minimum depth (cm), average depth (cm) and water availability (%; difference between highest and lowest volumes determined) was calculated.

In order to determine annual variations in the water volume (m^3), the volume was calculated bimonthly in 100 m^2 quadrants ($n = 3$), using an adapted version of the equation for lake systems (Schneider, 2000), which includes the area occupied by the islets. This equation is based on the volumetric measurement of horizontal sections of the water column, which are then aggregated according to their number of sections (bottom contour) to obtain the total volume (Equation 1).

$$V_{\text{total}} = \frac{H\left((A_1 - I_1) + (A_2 - I_2) + \sqrt{(A_1 - I_1)\cdot(A_2 - I_2)}\right)}{3} \quad (1)$$

Where H is the average depth (cm) of the water column in those sectors where the islets are not present. A_1 and A_2 represent the area (cm^2) of the upper and lower profiles of the water column, respectively, whereas I_1 and I_2 are the sum of the upper and lower areas (cm^2), respectively, of the present islets through the water column. The islets consist of the submerged portions of the arboreal vegetation, defined by the adventitious and horizontal roots systems that emerge from the main trunk, and accumulate fallen leaves and sediment around them (Correa-Araneda *et al.*, 2012).

The height of the water table (WT) was determined bimonthly by means of a scuttled perforation that determined the percentage of time for which wetlands remained flooded (depth > 0 cm), saturated ($0 > WT > -30$ cm) and dry ($WT < -30$ cm), according to the description by Cole *et al.* (2006).

Statistical analysis

In order to determine if there were differences at the temporal scale for the depth and water volume variables, a one way ANOSIM variance analysis was used (Clarke & Warwick, 2001), using the season as a factor (summer and winter) on a distance matrix (Euclidian). Regression and correlation (Pearson) analyses between hydrological variables and land covers were carried out. The same analysis was performed between the rainfall data and hydrological variables of the wetlands. Relationships were considered significant when $p < 0.05$. All the analyses and graphics were carried out by means of Primer v.6 (Clarke & Gorley, 2006) and SigmaPlot v.11.0 (Systat Software, Inc.) software.

RESULTS

The studied wetlands ranged in area from 107 to 346 hectares, with drainage basins between 1642 and 3630 ha and altitudes between 26 and 182 m.a.s.l. The land cover from non-native forest plantation varied between 3% and 25% (*Pinus radiata-Eucalyptus globulus*), whereas the land cover from native forest (*Nothofagus* spp.) varied between 6% and 36% (Table 1).

Annual rainfall records varied between 1186 and 1250 mm, with a clear seasonality, with between 70% and 77% concentrated in the winter (April-September) (Table 1). Monthly rainfall was closely correlated with the depth and volume of most of the wetlands ($r > 0.6$; $p < 0.001$), except for the water volume in Nohualhue ($r = 0.2$; $p > 0.05$) (Fig. 2).

Regarding the water regime, it was observed that the Petrenco, Quepe and Nohualhue wetlands presented a permanent surface cover, with depths that varied between 15.7 and 83.5 cm and a water volume of 592 758-5 826 864 m^3 (Table 1). On the other hand, the Pumalal and Vergel wetlands presented temporary regimes, which were flooded 199 (54.5%) and 215 (58.9%) days, respectively, in a period of one year, with depths that varied between 0 and 82.4 cm and a water volume that varied between 0 and 4 800 075 m^3. These wetlands remained without a visible water column during the entire summer season, specifically from December 2011 until April 2012 (Fig. 2). Statistical analyses support this variation because both the water volume and depth in these sites presented highly significant differences over time ($p < 0.001$; Table 2). However, when the water cover was not present, the soil in these sites remained saturated (Table 1).

In wetlands with a permanent water regime, the volume decreases from August (No) and September (Que) until March (No, Que). In Petrenco, this decrease was between September and December. The depth exhibits a clear de-

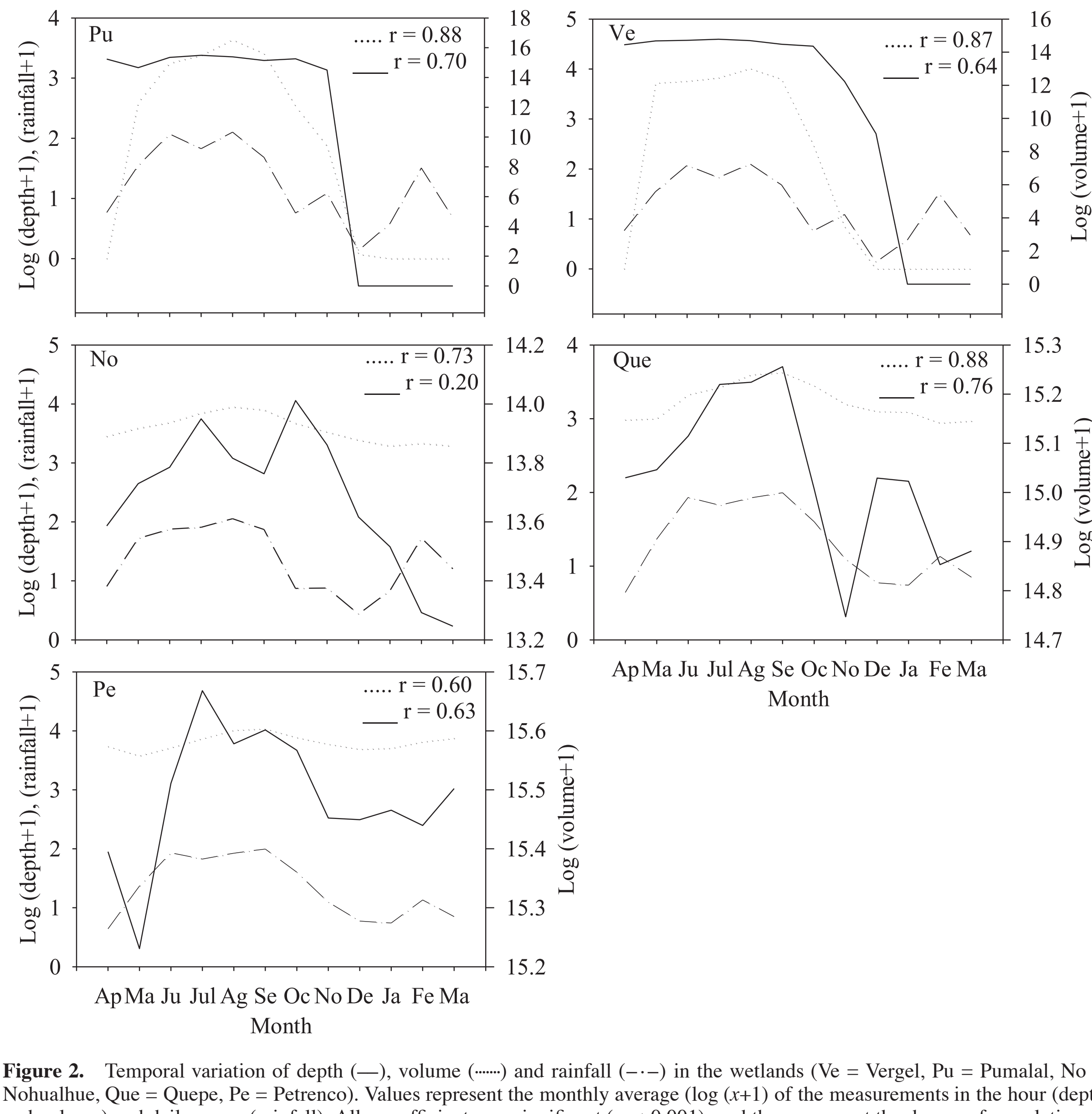

Figure 2. Temporal variation of depth (—), volume (······) and rainfall (–·–) in the wetlands (Ve = Vergel, Pu = Pumalal, No = Nohualhue, Que = Quepe, Pe = Petrenco). Values represent the monthly average (log (x+1) of the measurements in the hour (depth and volume) and daily range (rainfall). All r coefficients are significant ($p < 0.001$), and they represent the degree of correlation of the depth and volume of rainfall (except for volume in Nohualhue $p > 0.05$). *Variación temporal de la profundidad (—), volumen (······) y precipitaciones (–·–) en los humedales (Ve = Vergel, Pu = Pumalal, No = Nohualhue, Que = Quepe, Pe = Petrenco). Los valores representan el promedio mensual (log (*x*+1) de mediciones en un rango diario (precipitaciones) y horario (profundidad y volumen). Todos los coeficientes* r *son significativos (*p *< 0.001) y representan el grado de correlación entre las precipitaciones y la profundidad y volumen (excepto para el volumen en Nohualhue (*p *> 0.05).*

crease between October and March (Nohualhue), September and February (Que) with an important increase in December, and in Petrenco, from July to February with a small increase in September (Fig. 2). The variation in the water level was significant among the different seasons (winter and summer) with the exception of the Quepe wetland, which presented stable depths over time ($p > 0.05$). However, the volume of water presented no statistically significant variation over time ($p > 0.05$; Table 2).

The correlation analyses among the studied variables indicated a negative relationship ($p < 0.05$) between the percentage of land cover of non-native forest plantation in the basin and the water availability ($r = -0.86$), minimum depth ($r = -0.81$), minimum water volume ($r = -0.78$) and mean depth ($r = -0.58$) (Fig. 3). Significant and positive relationships ($p < 0.05$) were also found between the percentage of land cover in native forest and water availability ($r = 0.84$), minimum volume ($r = 0.83$), minimum depth ($r = 0.86$) and mean depth ($r = 0.75$) (Fig. 3).

DISCUSSION

By the late 1990s, approximately 50 % of the Earth surface had been directly affected by human action (Vitousek *et al*., 1997), and the forest industry was one of the economic activities with a greater contribution to this type of change (Ferreira *et al*., 2006), replacing a large amount of the forest cover in many countries with plantations of fast-growing non-native species. In Chile, these species were successfully established primarily between 32° and 41° south, covering approximately 17% of the forestland of the country (CONAF, 2011). Pizarro *et al*. (2006) and Little *et al*. (2009) recorded significant increases in non-native forest cover in central southern Chile. These authors indicate that in the period

Table 2. Volume (m^3) and depth (cm) values (mean ± standard deviation) from wetlands in the winter and summer seasons. *R*-values are the result of a one-way ANOSIM. Significance levels: ns = $p > 0.05$, *$p < 0.05$, **$p < 0.01$, ***$p < 0.001$. *Valores (promedio ± desviación estándar) de volumen (m^3) y profundidad (cm) de los humedales en las épocas de invierno y verano . Valores de* R *son el resultado de test de ANOSIM de una vía. Niveles de significancia: ns =* p *> 0.05, **p *< 0.05, ***p *< 0.01, ****p *< 0.001.*

	Summer	Winter	
Volume	Mean ± SD	Mean ± SD	*R*
Temporary			
Vergel	627 625 ± 1 079 473	2 201 119 ± 414 375	0.366 *
Pumalal	1 389 325 ± 2 406 381	4 555 887 ± 308 945	0.558 ***
Permanent			
Quepe	3 185 375 ± 314 059	3 682 939 ± 399 310	−0.075 ns
Petrenco	5 016 904 ± 144 134	5 679 684 ± 202 385	0.132 ns
Nohualhue	735 818 ± 124 486	1 062 379 ± 134 343	0.030 ns
Depth	Mean ± SD	Mean ± SD	*R*
Temporary			
Vergel	13.4 ± 13.3	42.5 ± 23.3	0.358 *
Pumalal	3.1 ± 4.6	20.3 ± 8.7	0.621 **
Permanent			
Quepe	15.2 ± 6.3	22.9 ± 8.5	0.156 ns
Petrenco	31.6 ± 8.6	48.6 ± 10.4	0.311 *
Nohualhue	12.0 ± 6.8	38.7 ± 24.7	0.372 **

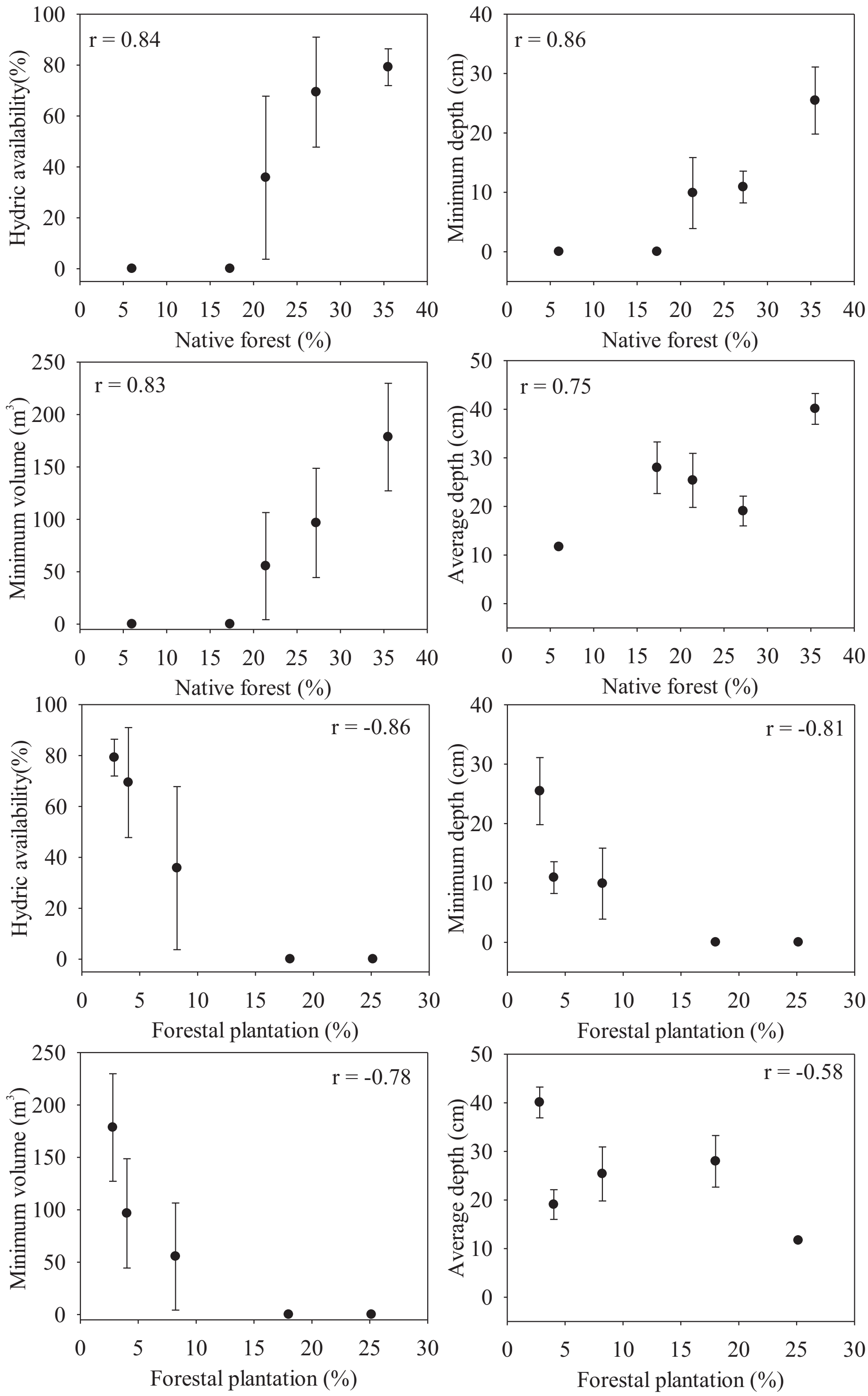

Figure 3. Regression analysis between the percentage of native forest soil cover and forest plantation, with water availability, minimum volume, average depth and minimum depth. *Análisis de regresión entre el porcentaje de cobertura de bosque nativo y plantación forestal exótica, con la disponibilidad hídrica, volumen mínimo, profundidad media y profundidad mínima.*

1975-2000 an increase in forest plantations of between 30% and 53% was produced, with an estimated annual rate that varied between 4.5% and 10.5%. Even between 1995 and 2009, Chile presented one of the highest annual deforestation rates of native forest (49 020 ha) and forestation (53 610 ha) with forest plantations in South America (FAO, 2010; INFOR, 2010). Some of the basins considered in this study show the same trend, reaching up to 25% of forest cover that is composed of non-native species. However, other basins present a low intervention (3%) from exotic species, which allows them to be used as reference sites in comparative studies.

The studied zone is characterised by rainfall ranging between 1200 and 1600 mm per year (Di Castri & Hajek, 1976, DGA, 2013). The results indicate that the studied sites had rainfall in the lower limit described for such zone, and its distribution was typical of a Mediterranean climate, characterised by a marked seasonality. Seventy to seventy-seven percent of the rainfall is concentrated between April and September, with a decrease in the summer. This is consistent with what has been described for the zone, where an even more noticeable seasonality has been registered, with 85% of the rainfall concentrated in the winter period (Little *et al*., 2009). Rainfall is the main influence on the monthly and seasonal variability of the water resources (Ward & Trimble, 2004; Pizarro *et al*., 2006; Little *et al*., 2009), which is reflected in this study with a clear seasonality in the water regime. It was observed that principally in the summer season, there was a significant decrease in the water depth and volume in all wetlands.

Although precipitation can clearly explain the water regimes of aquatic ecosystems, when analysing systems similar in geology, morphology and precipitation, it is interesting to evaluate the different land covers as well as other sources of variability in the water availability. This analysis showed a negative relationship between the non-native forest land cover and the water availability, minimum volume and minimum depth of wetlands. In particular, when there was more than approximately 18% of non-native forest land cover in the basins, these variables tend to zero, presenting shorter hydroperiods. In contrast, positive relationships between the percentage of native forest in the basins and the same variables was observed. This indicates that when more than 21% of the land cover is native forest, a permanent water regime was fostered in the wetlands. Similar results were obtained by Mbano *et al*. (2009) and Garmendia *et al*. (2012), who found a negative relationship among the percentage cover of non-native plantations, the water flow and the water productivity, respectively. These results indicate that an increase in the percentage of land intended for forest plantations would be adversely affecting the water inputs to the wetlands, especially when the normal distribution of the volumes in the summer season become critical. This is in agreement with similar studies carried out in Chile where it has been stated that the replacement of native forests by plantations of *Pinus* and *Eucalyptus* spp. produces a decrease in the water flows of small basins, especially during the summer season (Lara *et al*., 2009; Little *et al*., 2009), reaching 20.4% for every 10% increase in the area of non-native forest plantations (Lara *et al*., 2009). All this leads to the presence of large volumes of water in the winter, which is the product of rapid surface runoff due to the lack of an understory that generates interception, as well as the lack of the water resources in summer season, just as found in the present study.

Additionally, it has been indicated that changes in volume or minimum depth have always been considered more relevant than a variation in the annual average (Little *et al*., 2009) because they can directly affect the users of the water resource (Farley *et al*., 2005). In addition, these variables define the characteristics of the hydroperiod, which is one of the greatest influences on the structure of communities of both plants and animals in wetlands (Farley *et al*., 2005). Therefore, a change in this variable can generate modifications in the temporal distribution patterns of aquatic communities (Angeler *et al*., 2010).

These results are in agreement with the water losses associated with non-native forest plantations. Huber *et al*. (2008) estimated that in a plantation of *Pinus radiata* located in a similar zone to the study area, up to 40 % of the annual rainfall is lost, a product of the interception generated by the canopy and up to 55 % as a result of evapotranspiration. The latter can be explained by the high water demands, which have been ascribed by several authors to *Pinus radiata* plantations as well as *Eucalyptus* spp. (Farley *et al*., 2005; Jackson *et al*., 2005; Scott & Prinsloo, 2008).

It has been estimated that *Eucalyptus globulus* can achieve a transpiration volume five times larger than the amount transpired by slow growth trees such as *Podocarpus falcatus* or *Cupressus lusitanica* (Fetene & Beck, 2004). Likewise, Licata *et al*. (2008) estimated that *Pinus ponderosa* plantations had average water consumptions 33% to 64% larger than that of an *Austrocedrus chilensis* forest. Similarly, it has been observed that plantations of *P. radiata* in central southern Chile show only 5% of the percolation or infiltration of rainfall (Huber *et al*., 2008), producing a depletion of soil moisture reserves that can reach a depth of three metres (Hofstede, 2001). However, it has been determined that the conservation of riparian native forests may significantly reduce the adverse effects on water provision from forestry plantations of exotic species (Little *et al*., 2015).

It is important to mention that given the forest characteristics of the studied wetlands, there are processes that can differ from other lentic ecosystems. The forest component would be generating a larger interception of the direct precipitation in comparison with lacustrine ecosystems or freshwater marshes, which implies a lower water contribution during rainy periods. However, the same characteristics produce a lower penetration of solar energy to the water, resulting in lower evaporation of the water resource; this could explain the fact that those wetlands that presented a temporary regime maintained a continuously saturated soil.

In conclusion, the current study suggests that forest plantations of non-native species and native forest species are related to the water behaviour of forested wetlands, and the main factor is the percentage of the basins that is intended for this use. The results show the need to carry out better management of the basins that drain towards wetlands, especially in those in which the forest activity is not yet intensive. Better management can reduce the increase in the area planted with fast-growing non-native species by means of finding and fostering new economic alternatives for the owners. The conservation or restoration of buffer areas with native vegetation in areas surrounding wetlands is also an important issue. These zones have been recognized as areas of ecological importance (Lemly & Hilderbrand, 2000), because of their roles in the ecosystem processes regulation, water storage and aquifer loadings, among others, thus favouring water availability during drought (CHS, 2008, Little et al., 2015).

Given that the effects of these environments have been identified at the hydrographic basin scale, the protection of these ecosystems by means of the Chilean National System of Protected Wild Areas (SNASPE) does not fully ensure their conservation. However, this can be the first step for future integrated management, especially for those ecosystems that still present a low level of human intervention. In Chile, there are no models for wetlands protection, except for bogs (D.F.L. Nº 1122) or those wetlands that have been considered for an international protection model, such as Ramsar. For this reason, it is important to note that the conservation efforts through governmental tools should have to at least consider the protection of as many different ecosystems as possible. However, this does not happen with this type of wetlands because by being unknown or even not recognized as wetlands, they have not been considered within the Chilean protected areas to date. This is even more relevant if it is considered that the forested wetlands are unique ecosystems with global significance by being located in a biodiversity hotspot (Smith-Ramírez, 2004).

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