

Forest browning trends in response to drought in a highly threatened mediterranean landscape of South America

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ABSTRACT

Deforestation is widely studied throughout the world. However, a less evident issue is the effect of climate change and drought on remnants of native forests. The objective of this work was to understand the geographic variations in resistance to drought of the Mediterranean sclerophyllous forests of central Chile. These forests have been historically reduced and fragmented and in recent years were subjected to the most prolonged drought occurred between 2010 and 2017. Using data from the MODIS satellite sensor, temporal trends in the NDVI (Normalized Difference Vegetation Index) were quantified. We related these trends with different environmental variables to understand the effects of geographical variation and forest type as indicators of resistance to drought. We observed a significant direct effect of drought, attributable to the reduced precipitation in central Chile, and a significantly reduced NDVI in near one-third of the region forests (browning). However, NDVI and therefore forest productivity were more stable in some mesic sites such as ravine bottoms, but not on south-facing slopes. This suggests that under a regime of reduced precipitations, a greater available soil humidity would be a more important factor than the fact of receiving less solar radiation. Finally, the highest degree of browning was observed in semi-arid sclerophyllous forest dominated by species tolerant to drought. Our findings emphasize the need to consider topographic site conditions to adequately assess forest productivity and vulnerability where local wet conditions could provide drought refuges. This recent drought may be analogous to forecasted warmer and drier climate conditions with more frequent and severe droughts, so our results may serve as a general framework for climate-smart decisions in highly threatened forest restoration and conservation.

1. Introduction

The greatest threat to the conservation of terrestrial biodiversity, and in particular to remnant and fragmented native forests, are land use and land cover changes (Sala et al. (2000); Pereira et al. 2010; Newbold et al. (2015)), which have affected even forests with high conservation value (Allan et al. 2017; Potapov et al. 2017; Watson et al. 2018). Less evident are the effects of climate change on natural forests change, since they may operate in different directions, gradually and

cumulatively, generating changes in forest structure, composition, dynamics and function over less perceptible time-scales, but being able to affect wide areas (Allen et al. 2010; Allen et al. 2015; Clark et al. 2016).

Extreme climate events are one of the components of climate change and, specifically, droughts have been shown to negatively impact forests with widely varying effects on different biomes and tree species (Gazol et al. 2018). As a general rule, plants react to the hydric deficit by reducing their transpiration rate and net carbon assimilation rate through the closure of the stomata and, most importantly, by

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decreasing growth (Martínez Vilalta et al., 2002; Bréda et al., 2006; Feeley et al., 2007; Gazol et al., 2018). Prolonged drought may cause dysfunctions in the trees' physiology, with water shortage causing structural alterations including leaf and shoot shedding (Rood et al. 2000) and canopy dieback and raising the water-use efficiency (Grossiord et al. 2014). In extreme cases, trees may die when their capacity for water uptake and transport has become severely damaged leading to a hydraulic failure (Anderegg et al. 2013a). At community level it has been reported that the specific effect of droughts may have a direct impact on forests canopy structure and composition (Barbosa and Asner 2017), and thus on the productivity of forests (Zhou et al. 2014; Barbosa and Asner, 2017; Sulla-Menashe et al. 2018).

In a context in which there is uncertainty about the persistence of forest ecosystems due to climate warming and increased aridification (Watson et al. 2018), it is necessary to understand the attributes which provide forest ecosystems with greater resilience. However, recovery of their attributes and functions after disturbances may be difficult even with active restoration actions and could be less predictable in a global changing context (Moreno-Mateos et al. 2017; Jones et al. 2018; Altamirano et al. 2019). Moreover, the resistance of a forest ecosystem may provide also valuable information about the ecological contexts which would enable it to persist or remain stable within certain degree of environmental variability (Scheffer et al. 2009). This is exemplified by the fact that in some areas no changes have been observed in certain attributes of the vegetation (e.g. canopy structure, greenness) in many parts of the world, despite the severe droughts and the abnormal precipitations recorded (Morton et al., 2014; Zhang et al., 2017). This suggests a combined effect of variables operating on different spatial and temporal scales which may compensate, mitigate, buffer or exacerbate the effects of climate change and droughts on forests. Among them, CO₂ fertilization (Girardin et al. 2016; Zhu et al. 2016), drought intensity, topography, soil properties (Anderegg et al. 2013a; Dorman et al. 2015), and even structural aspects of forests such as size distribution, composition, density, age or diversity, may also determine their resistance to climate change and drought (Grossiord et al. 2014, Young et al. 2017).

Vegetation indices obtained by remote sensing allow the systematic production of synoptic information all over the world and covering a relatively ample time-scale (Pettorelli et al. 2005; Zhang et al. 2017). These indices are based on high energy absorption by vegetation in the red range of the spectrum and on strong reflectivity in the near infrared. The ratio between the two gives a physiological parameter for the abundance of chlorophyll in the leaves, which is what determines the capacity for photosynthesis and the growth potential in plants (Running et al. 2004). These indices respond to physiological changes expressed in the forests as energy flows (Singh et al. 2003; Lloret et al. 2007); they are therefore a key tool for quantifying the medium-term effects of climate change and the impact of particular climate events as drought on forest productivity (Zhou et al. 2014; Barbosa and Asner, 2017). The resistance of forests to drought has been assessed mainly at a specific level, through the changes in individual growth and performance resulting from a specific event (e.g. Lloret et al. 2011; Vitali et al. 2017). Few studies have assessed the geographic variation in resistance on a large spatial scale and as a function of different environmental gradients (but see Gazol et al. 2018). Likewise, assessments tend to address a single or a few drought events, without comparing its effect with the medium-term effects on forests of reduced precipitation and rising temperatures, and how these effects also vary as a function of environmental gradients.

The most prolonged and intense drought experienced in central and southern central Chile since instrumental records began in the early 20th century occurred since 2010 (Garreaud et al. 2017), in the context of a general reduction in precipitation reported for the area from 1980 (Boisier et al. 2016). The aims of this research are to quantify the effects of the drought -using remote sensing- on the photosynthetic productivity in the Mediterranean forests of central Chile as indicators of

its resistance to drought, and to understand the spatial and temporal variation of forest productivity. We expected that there would be a conspicuous effect on the photosynthetic productivity expressed in the browning of the Mediterranean forests. However, we expect that this effect would vary locally as a function of the predominant topographical conditions of each forest stand, considering that south-facing slopes in the Southern Hemisphere and riparian areas along ravines and creeks could provide mesic wet conditions. This drought condition may reflect the climate of the future in the area (Rojas et al. 2019); therefore, we expect our results may orient in this particular case and act as a general framework for climate-smart decisions in ecological forest restoration and conservation.

2. Materials and methods

2.1. Study area

The study area (~13,000 km²) comprises the area of central Chile lying between the Pacific Ocean and the Andean Pre-cordillera (71.5–70.2°W, 33.1–34.1°S). This region has been cataloged as a biodiversity hotspot (Myers et al., 2000) and it has suffered a high degree of human intervention (Schulz et al. 2010; Vergara et al. 2013) (Fig. 1). The prevailing climate is mediterranean, with precipitations concentrated in winter and drought in summer. The mean annual temperature varies between 12 °C in the coastal zone and 16 °C in the central valley of Chile. The mean annual precipitation in the central valley and the rain shadow may fall as low as an annual minimum of 220 mm (MIN = 66 mm, MAX = 608 mm, SD = 126.3 mm, Huechun Embalse station), while in the coastal area it may exceed 650 mm (MIN = 137.9 mm, MAX = 1366.4 mm, SD = 302.8 mm, Lago Peñuelas station) (see Fig. 1) (data at <http://explorador.cr2.cl/>). Lately, an uninterrupted sequence of dry years has affected central Chile, starting in 2010, with a deficit in precipitations ranging between 25% and 45% and peaking at latitudes between 31° and 33°S (Garreaud et al. 2017). Although Mediterranean ecosystems may normally experience one or two years of drought, the recent “mega-drought” (2010–2016) is longer and more intense than any since records began in 1915.

The vegetation in this area is a mosaic composed principally of degraded areas covered by *Acacia caven* savannah, patches containing remnants of coastal Mediterranean sclerophyllous forest dominated mainly by *Cryptocarya alba* and *Peumus boldus*, and Andean Mediterranean sclerophyllous forest dominated by *Quillaja saponaria* and *Lithrea caustica* (Luebert and Pliscoff, 2006). Sclerophyllous forest is concentrated in more mesic and south-facing zones, forming continuous patches of vegetation; in drier and north-facing zones, however, the physiognomy is generally that of an open, multi-specific arborescent scrub vegetation (Fuentes et al. 1984).

2.2. Remote-sensing data

Data were selected from the MODIS sensor, mounted in the TERRA and AQUA satellites. They have global coverage and are acquired and processed by Land Processes Distributed Active Center (LP-DAAC, <https://lpdaac.usgs.gov/>), linked to NASA's Earth Observing System. The Normalized Difference Vegetation Index (NDVI), combining the red and infrared bands of the electromagnetic spectrum, could be used for the estimation of several vegetation parameters like the proportion of photosynthetically active radiation absorbed by the canopy (Di Bella et al. 2004), the carbon fluxes through the ecosystem (e.g. Sellers et al. 1992), or Aerial Net primary production (e.g. Paruelo et al. 2000). The NDVI data consist of temporal series obtained from two products: MOD13Q1 (start date 18/02/2000) and MYD13Q1 (start date 04/07/2002). These data, with a high temporal resolution, are produced at 16-day intervals with pixel size 250 × 250 m, providing a consistent platform for comparing vegetation over time and space (Wen and Saintilan, 2015). The data were downloaded directly from the

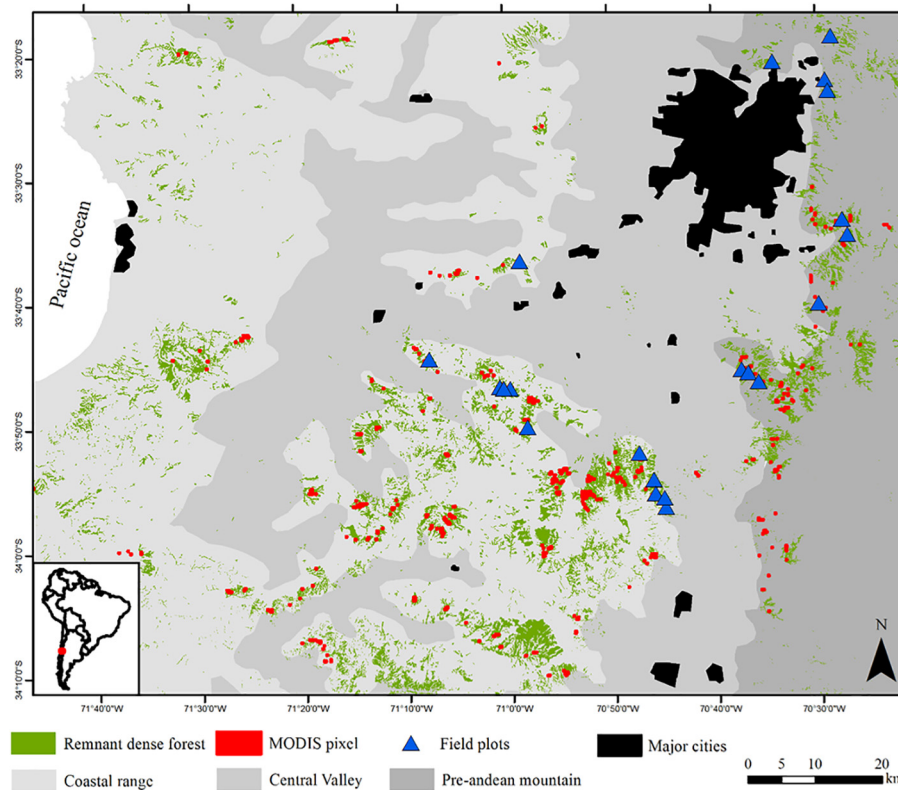


Fig. 1. Study area in central Chile. Locations of the MODIS pixels (temporal series) completely contained in dense forests according to CIREN and CONAF (2013), and the pixels with field plot.

Temporal Vegetation Analysis System (SATVeg, <https://www.satveg.cnptia.embrapa.br>) of the Brazilian Agricultural and Livestock Research Company (Embrapa).

Spatial analysis was carried out to select all the MODIS pixels which were entirely contained within a patch of dense native forest according to the official vegetation map provided by CIREN-CONAF (2013). This procedure avoided recovering a mixed spectral response. The SATVeg platform, which is linked to Google Earth, was used to confirm visually that the pixels selected were contained in patches of dense native forest. On this basis, the database consisted of 547 pixels (Fig. 1). A temporal series without clouds images were constructed for each pixel, consisting of 729 observations recorded between 18/02/2000 and 18/02/2017.

2.3. Temporal changes in NDVI

Two principal metrics were used to quantify the resistance of forest as drought response variables: i) the proportional change in the NDVI (C_p) between the pre-drought period (PreDr) and drought period (Dr) and ii) the magnitude of the NDVI trend over time β_{ndvi} . First, the ratio of the mean NDVI for the two periods was calculated at pixel level, considering the mean NDVI for PreDr between 18/02/2000 and 18/02/2009, and the mean NDVI in Dr between 18/02/2009 and 18/02/2017 (Eq. (1)):

$$C_p = -1 * \left(100 - \left(\frac{Dr * 100}{PreDr} \right) \right) \quad (1)$$

The temporal trend of the NDVI for each pixel was obtained considering three different periods: 18/02/2000 to 18/02/2017 (complete period), 18/02/2000 to 18/02/2009 (PreDr) and 18/02/2009 to 18/02/2017 (Dr). Trend magnitudes (slope of linear trend) for each period were estimated by Sens's slope test implemented in "trend" package (Pohlert 2018). To distinguish whether the temporal trend was statistically significant or due to random noise in the response, the p-value of

the estimated parameter for each time series was calculated in the "modifiednk" package (Patakamuri 2018). We use the modified Mann-Kendall test for temporal autocorrelated data, with Hamed and Ramachandra Rao (1998) variance correction approach. All the analyses were done in the R statistics software (R Development Core Team, 2015).

2.4. Environmental variables

Major environmental factors were assigned to each pixel to analyze the effect of the climate and local environmental conditions on NDVI trend. For climate data, each pixel was assigned the value of the slope in the temporal series for monthly precipitation and mean monthly temperature (period 1979–2017). This was done by fitting a simple linear regression for each temporal series as follows in Eq. (2):

$$y_i = \alpha_i + \beta_i * x_i + \varepsilon_i \quad (2)$$

where y_i is the value of the temporal climatic series for given pixel i , on date x_i , α_i is the intercept of the model, β_i is the model slope and ε_i are the residuals of the linear regression for each pixel i . In each pixel was also assigned the percentage decrease in precipitation and the percentage increase in temperature between PreDr (January 1979 to December 2009) and Dr (January 2010 to December 2016), as well as the annual precipitation and mean temperature. The temporal series for the climate variable was obtained from Boisier et al. (2016), based on georeferenced 5×5 km squares in a grid for which precipitation and mean monthly temperature were recorded for the whole country between January 1979 and December 2016.

The environmental factors considered were latitude and longitude, elevation, aspect and a topographic wetness index (TWI) defined below. The elevation and aspect were obtained from the digital elevation model (DEM) with a resolution of 90 m obtained from the Consortium for Spatial Information (CGIAR-CSI, <http://srtm.csi.cgiar.org/>). The

TWI is a relative indicator of the potential water available and is calculated as in Eq. (3):

$$\ln\left(\frac{a}{\tan b}\right) \quad (3)$$

where b is the gradient of the local surface and a is the area which may potentially contribute water to each geographical unit at each elevation isoline. It is calculated as $a = A/L$, where A (m^2) is the contributing area and L (meters) is the length of the isoline. A high TWI value is assigned to relatively flat conditions with a large potential contributing area, which may be expected to have a higher relative availability of water than areas with steeper slope and elevated areas with small contributing areas (Beven and Kirkby, 1979; O'Loughlin, 1981; Wilson and Gallant, 2000).

Each pixel was also assigned to one of the three principal forest types of the region according to CIREN-CONAF (2013), Coastal Mediterranean sclerophyllous forest of *Cryptocarya alba* and *Peumus boldus* (Ca & Pb), Andean Mediterranean sclerophyllous forest of *Quillaja saponaria* and *Lithrea caustica* (Qs & Lc) and Coastal Mediterranean thorny forest of *Acacia caven* and *Maytenus boaria* (Ac & Mb).

2.5. Modelling strategy

We assessed the relationships between predictors and the following response variables: i) NDVI proportion change between periods (Cp), and the magnitude of the NDVI trend between ii) 2000–2009, iii) 2009–2017 and iv) 2000–2017. The effect of the environmental variables on the response variables were assessed using a multiple linear regression model considering several predictors (latitude, longitude, elevation, aspect, TWI, mean annual precipitation, mean temperature and trends in the climatic variables). The candidate models were selected by minimizing the Akaike Information Criterion (AIC) (Burnham and Anderson, 2002). Where more than one model was found with a difference of < 2 units in the AIC, the model with the smaller number of predictor variables was selected. To verify the assumption of the spatial independence of the observations, we assessed the spatial self-correlation of the residuals using Moran's correlogram (Moran, 1950). The correlograms were constructed at a constant distance (250 m). For each distance class we carried out 1000 permutations to verify whether the level of self-correlation differed significantly from a random distribution, tested with the Bonferroni sequential correction (Legendre and Legendre, 1998). This analysis produced spatial autocorrelation of the observations for the first 500 m. With this information, in cases where the points were located at a smaller distance, a random point was selected for each 500×500 m grid square by spatial sampling. The analyses were repeated until a database was found with no evidence of spatial self-correlation and which retained the greatest number of observations. The final database for the multivariate analysis included the temporal series of 353 pixels. The relative importance of each regressor and the variance explained by the model were calculated using the "relaimpo" package in the R statistics software (Grömping, 2006; R Development Core Team, 2015). This value represents the weighted contribution ordered between the regressors to R^2 (Chevan and Sutherland, 1991).

2.6. Field validation and data analysis

Field work was carried out with the two purposes; i) relate the satellite information with the field information, discarding a different effect than drought on the NDVI as logging and fires and ii) validate the official cartographic information about tree species composition. Previous selection of pixels was done by a random-stratified selection considering the two predominant forest types. However for logistical reasons the most accessible pixels, located at a mean distance of 2.5 km from the nearest road, were selected. We visited 21 pixels in the field between August and October 2017. We established 20×20 m

(400 m^2) plots in the center of each pixel where we identified and measured the diameter at breast height (DBH) of all trees with DBH greater than 5 cm. For the purposes of analysis of the field data, each plot was assigned to one of the two predominant forest types: coastal Mediterranean sclerophyllous forest of *Cryptocarya alba* and *Peumus boldus* or Andean Mediterranean sclerophyllous forest of *Quillaja saponaria* and *Lithrea caustica* (Luebert and Pliscoff, 2006). Composition and basal area of the stands were registered, and the plot mean canopy loss was estimated through a visual and individual estimate of crown transparency of each tree. The canopy loss was taken to be all that part of the crown of each tree with dry leaves or leafless branches and it was estimated using binoculars. The percentage was quantified in 10% intervals. In addition, boxplot visual analysis was used to compare the canopy loss measurements taken on the field with the percentage change in the NDVI in the same pixel during the drought period.

3. Results

3.1. NDVI temporal trend and their predictors

In the percentage change estimates of NDVI (Cp), a decrease over the 2000–2017 period was observed in 91% of the cases. Of the 536 temporal series analyzed, 74% presented a negative trend since 2000, with a significant decrease (p -value < 0.05) in 39% of decreasing series. However, Cp exceeded 5% in only 12% of cases, reaching a maximum decrease of 11.29%. Positive change in NDVI value was found in 8.8% of the temporal series, but only 13% (18 pixels) showed a significant increase (p -value < 0.05), with a maximum increase of 4.49%.

The percent change in NDVI (Cp) between 2000 and 2009 and 2009–2017 periods was positively related to latitude, longitude, temperature increase and annual precipitation. These results imply that a higher net decrease in forest NDVI was higher toward the north pre-Andean area, in the driest part of the study area, where also the increase in temperature was higher. These variables together explained 30.8% of the change in NDVI (Table 1; Fig. 2). The best model for NDVI trend slope from 2000 to 2017, retained the same variables that Cp model, maintaining the direction of their effect (Table 1).

The variance in the model of the trend in the PreDr years was explained to a smaller degree (28%) with latitude presenting higher relative importance of 63% (Table 1). The model for NDVI trend in the Dr period (2009–2017) presented an increase in the variance explained to 50%; however latitude was maintained with a higher relative importance (57%). Two new variables were also retained by the model, including the TWI and the percent decrease in precipitation for the period. These produced a more pronounced decreasing trend in NDVI in areas with a lower accumulation of topographic wetness, and also in zones where a lower proportional decrease in precipitation was observed in the north pre-Andean area (Fig. 2).

3.2. NDVI temporal trends in the main forest types

The MODIS pixels were distributed among three forest types: 46% in coastal Mediterranean sclerophyllous forest of *Cryptocarya alba* and *Peumus boldus* (Ca & Pb), 50% in Andean Mediterranean sclerophyllous forest of *Quillaja saponaria* and *Lithrea caustica* (Qs & Lc) and 4% in coastal Mediterranean thorny forest of *Acacia caven* and *Maytenus boaria* (Ac & Mb). For the whole period, we found that 75% of Ca & Pb pixels showed no temporal NDVI trend, while in 21% of them there was a significant negative trend (browning) and in only 4% of the pixels a significant ($p < 0.05$) positive trend (greening) was observed (Fig. 3). In Qs & Lc , 38% of the series presented a significant negative trend (browning), and we did not observe positive trends in any pixel (Fig. 3). For Ac & Mb we found that 65% of the pixels presented no trend, whereas only 10% of them showed negative and significant trends.

Table 1

Linear models of the relationship observed between NDVI and environmental variables for different periods. SE = standard error. Relative importance = relative contribution of each regressor to the variance explained by the model.

Response variable	Predictor variables	Coefficients	SE	Relative importance	p-value
NDVI change (Cp)	(Intercept)	-2.16E + 02	3.54E + 01		2.71E-09
	~ latitude	1.33E + 00	3.87E-01	39%	0.000673
	+ longitude	3.41E + 00	6.77E-01	19%	7.61E-07
	+ PAnnual	2.64E-03	7.06E-04	22%	0.00022
	+ Tincrease	9.67E-01	4.40E-01	20%	0.028654
NDVI trend 2000–2017	(Intercept)	-1.75E-02	2.25E-03		6.82E-14
	~ latitude	2.88E-04	4.29E-05	44%	8.51E-11
	+ longitude	1.06E-04	2.46E-05	20%	2.27E-05
	+ PAnnual	1.59E-07	4.48E-08	19%	0.000452
	+ Tincrease	5.00E-05	2.79E-05	17%	0.07392
NDVI trend 2000–2009	(Intercept)	-2.47E-02	2.21E-03		< 2.00E-16
	~ latitude	3.94E-04	4.69E-05	63%	1.21E-15
	+ longitude	1.62E-04	2.77E-05	37%	1.11E-08
NDVI trend 2009–2017	(Intercept)	-3.83E-02	2.77E-03		< 2e-16
	~ latitude	7.14E-04	5.84E-05	57%	< 2e-16
	+ longitude	1.84E-04	3.57E-05	17%	4.37E-07
	+ TWI	4.00E-05	1.48E-05	5%	0.00719
	+ PPdecrease	-8.14E-05	1.22E-05	21%	1.14E-10

3.3. Forest structure and composition

The field sampling was carried out in such a way as to check the composition of the two principal forest types in the region (*Qs* & *Lc* and *Ca* & *Pb*) and to relate remotely sensed and field browning measure. A total of 14 tree species were found during sampling. Eleven of these species, representing 98% of the total basal area, were found in the plots, and belong to late successional stages according to the classification of Hoffmann (1989). The *Ca* & *Pb* plots were dominated by less drought-tolerant species like *Cryptocarya alba* (57.8%) and *Peumus boldus* (14.8%), whilst *Qs* & *Lc* plots presented a greater representation of more drought-tolerant species like *Lithrea caustica* (22.6%), *Kagenckia oblonga* (20.7%) and *Quillaja saponaria* (5%), although there was still a strong presence of *Cryptocarya alba* (36%) (Table 2).

On average, *Qs* & *Lc* presented greater canopy loss than *Ca* & *Pb* (11.0% vs 5.7%), which agrees with the difference in NDVI decrease measured through remote sensing (Fig. 4). In both cases, we found a significant ($p < 0.05$) difference between forest types.

4. Discussion

4.1. Climatic and NDVI geographic variation and trends

The results found reflect an effect of the 2010–2017 “mega-drought” on the productivity and canopy cover of the study forests, expressed in decreasing NDVI, particularly affecting the pre-Andean mountain range (Fig. 3). This phenomenon of browning has been reported in different studies around the world as an effect of climate change, and particularly of drought (Vicente-Serrano and Heredia-Laclaustra, 2004; Samanta et al. 2010; Asefi-Najafabady and Saatchi, 2013; Zhou et al. 2014; Hilker et al. 2014; Girardin et al. 2016; Barbosa and Asner, 2017; Garreaud et al. 2017; Sulla-Menashe et al., 2018). However, this effect is not generalized, since greening has been also reported in extensive areas of the world (Zhu et al. 2016), attributed principally to large-scale CO₂ fertilization (Zhang et al. 2017).

This variable pattern of greening and browning – and even absence of NDVI change – under abnormal precipitation regimes observed at the global scale (Morton et al. 2014; Zhang et al. 2017) has also been documented here in south-central Chile (30°–42° S) after the 2010–2016 “mega-drought”. Different underlying factors may determine this variation. In the study region, climatic conditions have become drier since the 1980 s, this tendency being stronger in the Andean Pre-cordillera, particularly between latitudes 33° and 37° S (Boisier et al. 2016). In addition, temperature increases have been observed since the 1970 s

(Falvey and Garreaud, 2007; Vuille et al. 2015). According to Garreaud et al. (2017), the greater intensity of the 2010–2017 “mega-drought” was associated with pronounced vegetation browning, with changes in precipitation explaining more than 50% of the Enhanced Vegetation Index variance. These results, however, comprise the variations in all vegetation cover types, including crops, meadows, forestry plantations and scrubs over the whole latitudinal range and at a spatial resolution of 5 × 5 km. Our results, covering a smaller range of latitude (33.1°–34.1°S) and considering only dense natural forests, show an opposite trend, in which the greatest decrease of NDVI is associated with a smaller percentage decrease in precipitation for the drought period (Fig. 2). We also found greater browning towards the Andes Mountains than in the forests of coastal zones. This agrees with a stronger intensity of the mega-drought towards the inland valleys of the Andes (Garreaud et al. 2017).

For the percentage NDVI difference between the periods, the mean annual precipitation is the determining factor, with the greatest percentage decreases in NDVI found in the areas of the region with drier climate conditions. However, there is a local effect of topography, where the higher the TWI values, the lower the browning rate. This result suggests the softened effect of topographical moisture during the period of the “mega-drought”; nevertheless, the TWI is not retained by the model in the analysis of the trend for the whole period. The results reveal that although the structure and composition of Mediterranean forests is highly influenced by their exposure (Fuentes et al., 1986), in conditions of extreme drought the effect of water accumulation as a result of the topography – as a proxy for the availability of water to the forests – would have a greater effect on maintaining forest productivity than the aspect (Table 1). Ravine bottoms are characterized by very dense forests favored by wetter conditions, and lower solar radiation than north-facing slopes.

When we modeled the NDVI trend during the drought period, we observed a counter-intuitive trend in which the greater the decrease in precipitation, the lower the rate of decrease in the NDVI. The trend in NDVI for the drought period is found to be inversely related to the intensity of the drought. This gives substance to the idea that certain areas of forest ecosystems in its current state may be reaching a threshold in their capacity to resist prevailing precipitation levels, and may suffer a change in their state through the reorganization of forest structure and composition. However, this response may be also an effect of leaf shedding, a strategy to reduce water loss through the canopy as occurs in other Mediterranean ecosystems (Lo Gullo and Salleo, 1988; Peñuelas et al. 2001). This is reflected in the NDVI due to the loss of the photosynthetic capacity which is concentrated in the leaves. Such

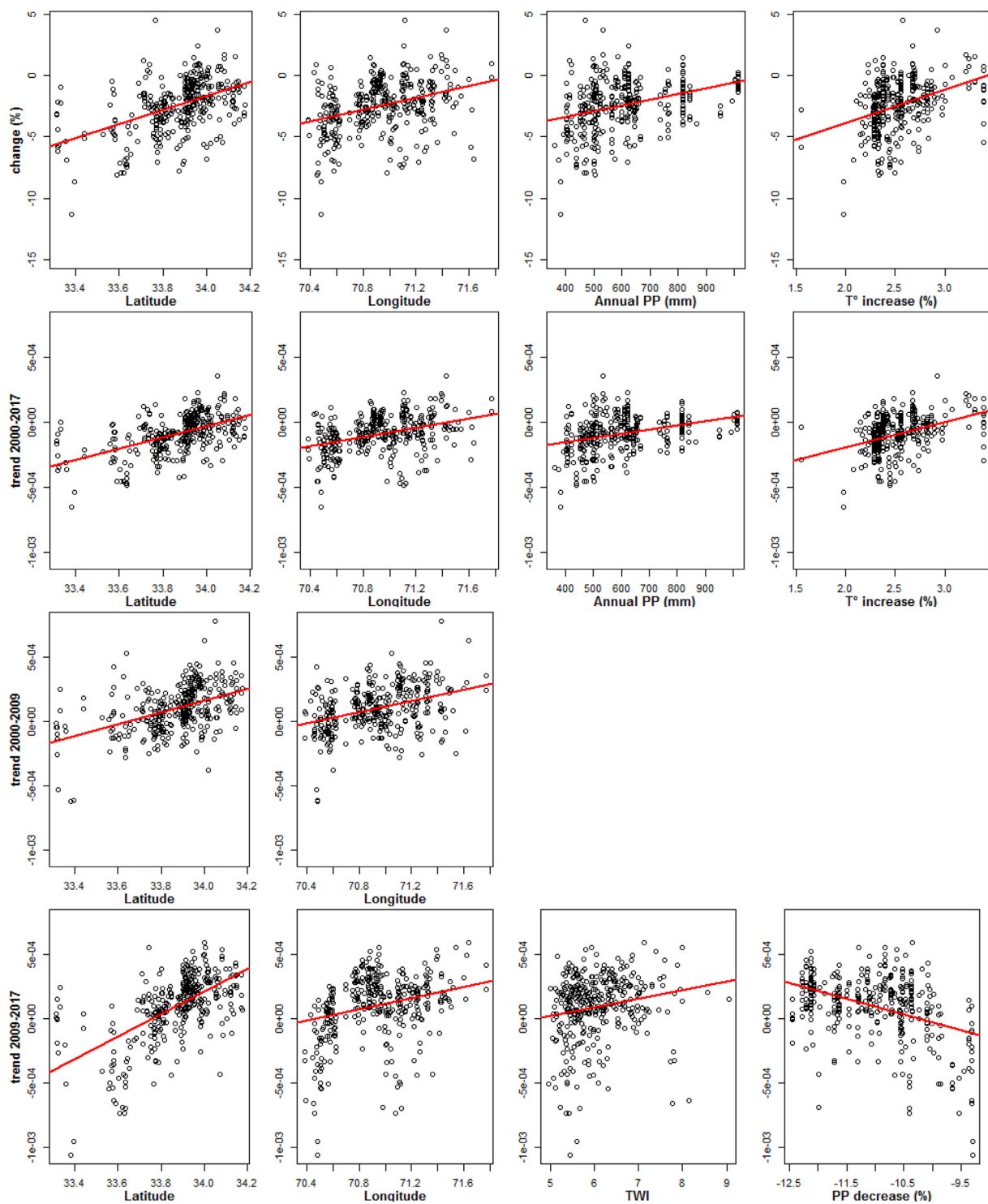


Fig. 2. Relationships between NDVI mean change and trend slope per period with different environmental variables retained by the models. Latitude and Longitude in decimal degrees. TWI is a relative indicator, high values are assigned to relatively flat conditions with a large potential humidity contributing area.

drop in productivity does not necessarily imply the death of the individuals or abrupt changes in forest structure, since – if the forest has maintained its resilience – this response may be reversed (e.g. through resprouting) when the precipitation regime recovers. However, considering that the trend is for precipitation to decrease (Garreaud et al., 2017), the recovery of historical precipitation regimes is uncertain. A

continued aridification trend could indeed lead to the death of tree individuals when physiological limits are overstepped, implying changes in the species composition and the structure of the forests (Lewis, 2006; Enquist and Enquist, 2011; Fauset et al. 2012; Clark et al. 2016; Barbosa and Asner, 2017).

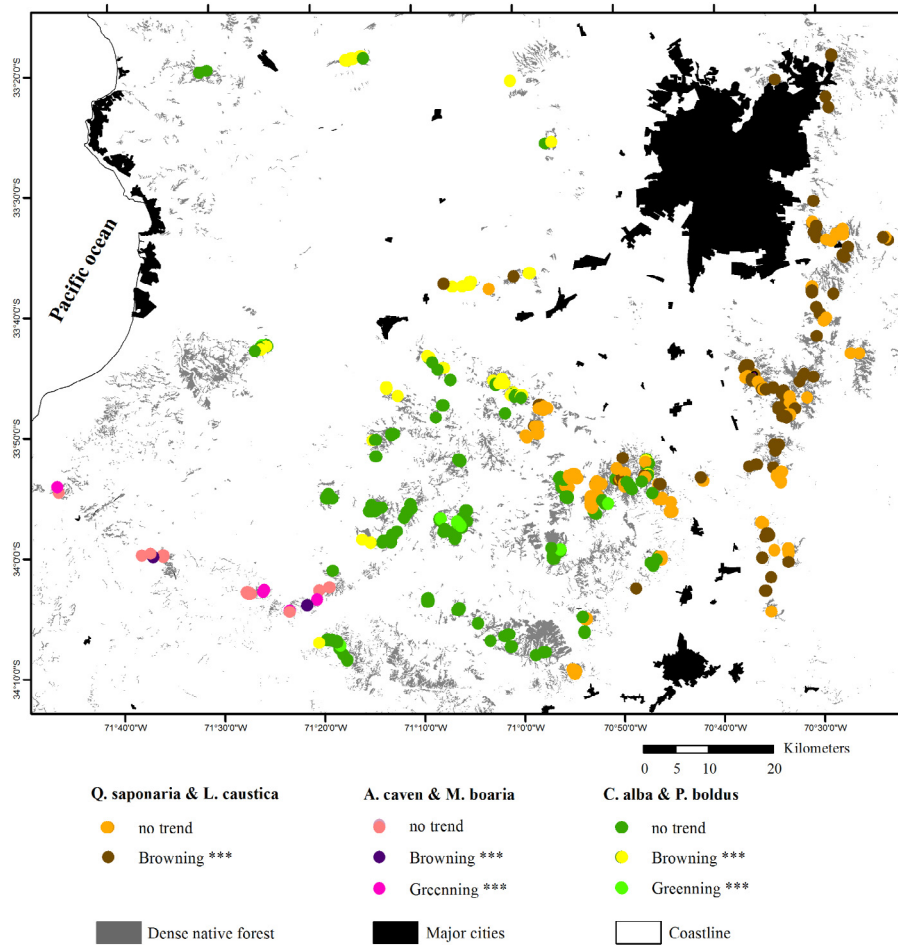


Fig. 3. Temporal trends in NDVI for the different forest types. Asterisks indicate significance levels: *** p -value < 0.05.

Table 2

Composition and dominance measured on the field for the principal forest types in the region (Q_s & L_c and Ca & P_b). The percent basal area is indicated for early- and late-successional species.

Forest types	Q_s & L_c		Ca & P_b			
	Mean	Range	Mean	Range		
Basal area ($m^2 ha^{-1}$)	20.7	3.2	47.5	22	9.6	26.9
Density (trees ha^{-1})	970	375	2750	1368	500	2250
Canopy loss (%)	11	0	34.5	5.7	0.7	14
Late-successional species						
<i>Lithrea caustica</i>	22.6	1.5	78.1	12.5	0	46.4
<i>Quillaja saponaria</i>	15	0	50	4.7	0	30.6
<i>Kageneckia oblonga</i>	20.7	0	78.2	1.3	0	10.2
<i>Cryptocarya alba</i>	36	0	93.8	57.8	25	88
<i>Peumus boldus</i>	3.7	0	21.1	14.8	0	49.7
<i>Schinus latifolius</i>	0	0	0	3.5	0	29.2
<i>Escallonia pulverulenta</i>	1.1	0	11.3	1.4	0	5.9
<i>Azara celastrina</i>	0	0	0	2.7	0	18.1
<i>Persea lingue</i>	0	0	0	0.8	0	8.2
<i>Azara petiolaris</i>	0	0	0	0	0	0.2
<i>Maytenus boaria</i>	0	0	0.1	0	0	0
Early-successional species						
<i>Baccharis linearis</i>	0.8	0	6.9	0	0	0
<i>Acacia caven</i>	0	0	0.4	0.2	0	2
<i>Trevoa trinervis</i>	0.2	0	2	0.4	0	3.2

4.2. NDVI temporal trends vary as a function of forest type

Marked differences exist in the browning observed in different types of forests. In Ca & P_b stands there is generally no temporal trend in the NDVI, and an increase in the NDVI may even be observed in a small percentage of the pixels; the opposite occurs in Q_s & L_c stands, where areas with significant browning predominate. Q_s & L_c forests are distributed throughout the latitude range studied, but mainly restricted to the pre-Andean Range, whilst Ca & P_b forests are concentrated mainly in the southern part of the study area, in the “Altos de Cantillana” sector, which forms part of the Coastal range.

In general, studies which have measured trends in photosynthetic activity and forest productivity do not include field validations based on exploration of forest composition and structure on the ground, perhaps due to the high cost of analyzing large areas (Samanta et al., 2010; Saatchi et al., 2013; Zhou et al., 2014; Hilker et al., 2014; Garreaud et al., 2017). This may lead to confusion in the interpretation of the results, considering the differences in the composition of plant communities (Barbosa and Asner, 2017); the more so in semi-arid or drought-prone Mediterranean ecosystems where differences are found in the species compositions best adapted to drought conditions. For example, in central Chile the variation in species composition is largely due to the influence of the Pacific Ocean, since the temperature range is narrower close to the coast, with more frequent morning fogs. These conditions reduce the stress suffered by plants during summer droughts, leading to a different species composition than pre-Andean forests (Fuentes et al. 1984, 1986; Holmgren et al. 2000, Fuentes-Castillo et al. 2012).

In our results, forests with higher decrease in productivity coincide

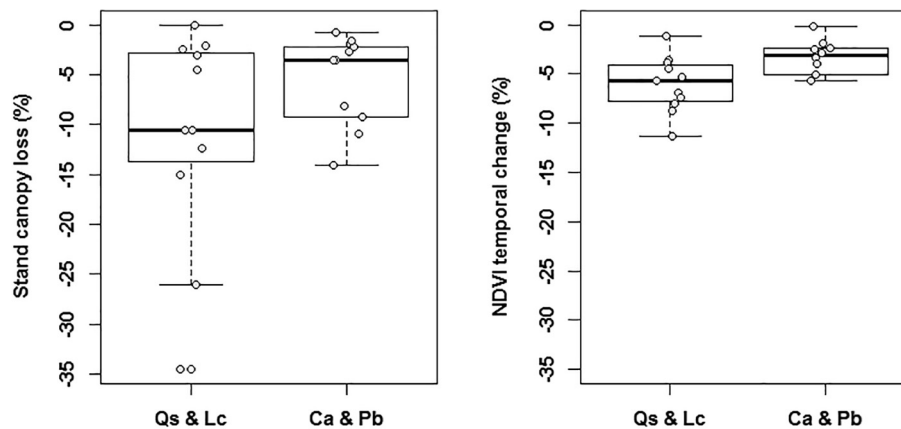


Fig. 4. Comparison of canopy loss data obtained through field sampling and NDVI data obtained from remote sensing. Left: average percentage of individual canopy loss per stand. Right: percentage NDVI change between periods of the pixels measured on the field.

with Qs & Lc stands, dominated by species from xeric habitats which are adapted to drought conditions. This occurs principally in the northern part of the study area and in the Andes mountains. Nevertheless, the species with the largest number of individuals with greater than 30% crown loss was *Cryptocarya alba*, a species generally found in moist sites; note that in the field, it was found to be the dominant species in both forest types. The severe historical use and intervention suffered by the forests in central Chile (Schulz et al., 2010; Vergara et al., 2013) relegate the remnants of dense forests dominated by native species mainly to protected zones in ravine bottoms. This could cause a confusion factor, because *Cryptocarya alba* would also dominate a large part of the area covered by Qs & Lc stands in areas with higher humidity, together with *Lithraea caustica*. Independently of the above, it has been reported in Mediterranean ecosystems that forests with lower resistance would have greater post-drought resilience based on the adaptive measures adopted by the tree species, and that more resistant forests may have problems in recovering their productivity and growth once the precipitation regime is re-established (Gazol et al. 2018).

In the field measurements we found concordance between the browning detected by satellite and the percentage of canopy loss observed at the stand level. This implies that Qs & Lc forests suffer a greater decrease in the photosynthetic canopy area (Fig. 4), reducing the uncertainty surrounding other predominant factors that might act to confuse the large-scale trends in NDVI, such as illegal felling, cattle-farming and fires. All the sites sampled are dominated by species considered late-successional. It may therefore be assumed that these forests established before 1980, the date from which the general decrease in precipitations has been recorded in central Chile (Boisier et al. 2016), since their composition and structure are comparable to those found by Fuentes-Castillo et al. (2012) in forests established before the 1950 s. However, our analysis does not reveal whether there have been temporal changes in the structure and composition of the forests. According to climate change scenarios, the areas where the greatest decrease in NDVI is projected would be highly vulnerable to a change of state from dense Andean Mediterranean forest to more open forest types, with the displacement of the sclerophyllous forest (Martinez-Harms et al. 2017). A loss of resistance of this ecosystem by increased aridification may open the way to indirect effects of climate change, such as changes in soil cover (Martinez-Harms et al. 2017), biological invasion, increased fires and deforestation, and increased vulnerability to new extreme climate disturbances (Seidl et al. 2017).

The changes observed, if they are continued in the long term, may modify the composition, distribution, dynamics and structure of the dominant tree species, and the composition of the undergrowth, with an indirect cascade effect on the associated communities through changes to the structure and quality of the habitat and related ecosystem services (Barbosa and Asner, 2017). All these effects may occur

simultaneously and in synergy and exacerbated by other processes (e.g. fire), affecting the structure and ecology of forest communities as well as their functions through changes in the transpiration, interception and infiltration of rainwater, modifying the hydric regime in the ecosystem and the nutrient cycle in the soils (Anderegg et al. 2013b). Finally, the understanding of the thresholds of forest resilience and resistance to climate change and variability, i.e. climate extremes as droughts, and their effects on drought-prone Mediterranean communities are key for conservation and ecological restoration of the remnants of these natural forests (Holl and Aide, 2011).

5. Conclusions

A significant negative effect of the “mega-drought” was observed on the productivity of Mediterranean Chilean forests, but local environmental conditions providing wet conditions and forest composition enhancing forest resistance buffered or alleviated the negative effects of water shortage. In particular, the effect of moist “refuges” was an important factor; in areas intensively affected by the “mega-drought”. This represents an opportunity and a criterion for selecting restoration sites for Mediterranean forests under climate change scenarios. At the same time, although a differential effect is observed on different forest types, it is still not possible to distinguish in which of the considered forest types the drought had the most intense effect, due to potential adaptive responses. Future analyses should evaluate the post-drought recovery period to quantify multiple processes of forest resistance and resilience including productivity, growth, resprouting, mortality, regeneration, and shifts in species compositions.

CRedit authorship contribution statement

Alejandro Miranda: Conceptualization, Data curation, Formal analysis, Funding acquisition, Investigation, Methodology, Project administration, Resources, Software, Validation, Visualization, Writing - original draft, Writing - review & editing. **Antonio Lara:** Conceptualization, Funding acquisition, Resources, Writing - review & editing. **Adison Altamirano:** Conceptualization, Funding acquisition, Resources, Writing - review & editing. **Carlos Di Bella:** Data curation, Writing - review & editing. **Mauro E. González:** Conceptualization, Writing - review & editing. **Jesus Julio Camarero:** Writing - review & editing.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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