Forest Ecology and Management 503 (2022) 119800

Contents lists available at ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

Combining remote sensing and field data to assess recovery of the Chilean Mediterranean vegetation after fire: Effect of time elapsed and burn severity

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A R T I C L E I N F O

Keywords:
Fire severity
Forest fires
Forest resilience
Herbaceous cover
Sclerophyllous forest
Regeneration of woody species

A B S T R A C T

It has been debated if Chilean sclerophyllous vegetation can recover after fire of different severity and short and long term. We evaluated the resiliency of this vegetation type after single-occurrence fires of different severities that occurred 30, 20 and 10 years in Central Chile before the study. Two approaches were followed: satellite image analysis and vegetation sampling. Wildfires that occurred between 1985 and 2015 were identified based on Landsat images. We selected 30 sites burned by a single fire in either 1985, 1995 or 2005, and that was not converted to another land use by 2015, then determined the percentage cover by vegetation type. We recorded or estimated the independent variables of burn severity, slope, altitude, pre-fire vegetation type and latitude of the sites. Composition, richness, and abundance of adult and regeneration of woody vegetation and herbaceous cover were sampled. Generalized linear models were used to evaluate the effect of the independent variables and the time elapsed since the fire on vegetation recovery. The proportion of dense vegetation forest cover was significantly higher with more time elapsed since the fire, while semi-dense forest/shrubland and open shrubland vegetation cover returned to pre-fire levels more quickly. The richness and abundance of regenerated woody species was significantly greater with more time elapsed since the fire. However, no relationship was found between species richness and abundance of adult woody species and time elapsed post-fire. We found that vegetative recovery over time was not related to burn severity. Forested and mixed forest/shrubland cover is reached 10–20 years after the fire, if no further intervention occurs. Richness and species abundance was similar to that of unburned sites after 20 or more years post-fire. This study provided evidence that forest vegetation in the Chilean Mediterranean ecosystem is resilient to fires of at least low and medium intensities.

1. Introduction

Fires currently represent one of the main human-caused disturbances worldwide. At the same time, many ecosystems are located in areas where fire also occurs by natural causes, and as a consequence, many species have adapted to survive and/or reproduce after them (Bond and Keeley, 2005; Naveh, 1975). In Mediterranean ecosystems, many species can regenerate after fire as a result of an evolutionary history of biological adaptation to these events (Naveh, 1975). It has been reported that natural fires are common in four (California, South Africa, the

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https://doi.org/10.1016/j.foreco.2021.119800
Received 3 May 2021; Received in revised form 13 October 2021; Accepted 16 October 2021
Available online 29 October 2021
0378-1127/

Mediterranean Basin and South Australia) of the five Mediterranean-type climate ecosystems in the world of which Chile being the fifth (Naveh, 1975).

Regular occurrence of thunderstorms without rain and corresponding increased likelihood of natural fires is often cited as a factor in promoting the evolution of fire-resilient plant species in Mediterranean ecosystems (Naveh, 1975; Montenegro et al., 2004; Armesto et al., 2009). Such natural fires are not common in central Chile, probably because of the rare occurrence of thunderstorms without rain, however many woody species in this region recover after fires (Montenegro et al., 2004; Pausas et al., 2008; Armesto et al., 2009; Becerra et al., 2018). Another existing source of fires in Chile are volcanic events, however, they are uncommon. Based on carbon deposits of volcanic origin, Abarzúa et al. (2016) nevertheless proposed that Chilean vegetation could have developed mechanisms to tolerate fires, at least during the transition from the early to middle Miocene (17–15 Ma), when the climate in central Chile was warm and dry and volcanism was more frequent.

Seed production of Chilean species is not stimulated by fire as it is for example the case for the Californian chaparral species (Parker and Kelly, 1989; Muñoz and Fuentes, 1989; Armesto et al., 1992). Gómez-González and Cavieres (2009) showed that the seed bank of Chilean woody native sclerophyllous species cannot survive high intensity fires, and seed germination of Chilean Mediterranean species is not stimulated by smoke (with some exceptions, Muñoz and Fuentes, 1989), as occurs in the Californian chaparral species and fynbos of South Africa. Besides, Chilean species do not usually have lignotubers (adventitious shoots that allow regeneration after fire) (Montenegro et al., 2003, 2004). It has also been reported that there is little plant recruitment from seeds after fires in Chilean sclerophyllous forests (Gómez-González et al., 2011). In contrast, trees and shrubs in Chilean Mediterranean forests were found to regenerate through re-sprouting (Montenegro et al., 2004), with some seeds remaining viable after low intensity fires (Gómez-González and Cavieres 2009; Gómez-González et al., 2017).

Fires have been common in central Chile in the last three centuries, due to human activities (Fuentes and Muñoz 1995; González et al., 2019) and because of the marked decrease in rainfall in the last century (CONAMA, 2006; Le Quesne et al., 2006). However, the capacity of native (woody and herbaceous) vegetation to regenerate and recover after fire in this region is not yet clear.

Different factors may affect vegetation resilience after a fire, including the time elapsed since the last fire (Tessler et al., 2015) and fire severity (Meng et al., 2015). In sclerophyllous forests, high severity could imply more mortality and probably fewer remnant biological components from which vegetation can resprout, produce seeds and grow (Segura et al., 1998). Furthermore, abiotic factors such as climate and topography (e.g. slope, altitude) may affect the capacity of vegetation to recover (Hogenbirk and Wein, 1991; Keeley, 2009). The same applies for biotic factors like the pre-fire vegetation type (forest, shrubland, grassland) which can influence vegetation recovery (Cohn et al., 2015). Also, it has been proposed that in areas where human disturbances after fire are common lower recovery in terms of species composition and vegetation cover is reached (Baeza et al., 2007).

Mediterranean-type ecosystems are found in Chile between 31°S and 37°S, extending along the coast north to 23°S, and through the central valley south to 39°S (Amigo and Ramírez, 1998; Luebert and Pliscoff, 2012). This ecosystem is considered a biodiversity hotspot with a critically threatened status (Myers et al., 2000, Olson and Dinerstein, 2002). From 1975 to 2008, 42% of native forests and 22% of native shrublands disappeared in central Chile (Schulz et al., 2010), due to mainly changes in land use. The latter particularly include the intensification of agricultural production (vineyards and fruit farming), often associated with recurring and extensive human set y induced fires. In this area of Chile, the frequency of wildfires in the same site is two, followed by one fire (Smith Ramírez et al,. 2021).

Post-fire vegetation resilience was traditionally studied using field data (Trabaud et al., 1987, Purdie and Slatyer, 1976, Altamirano et al., 2019), but in the last fifteen years, satellite analysis has been increasingly used to assess post-fire vegetation recovery (Díaz-Delgado and Pons, 2001; Inbar et al., 1997; Malkinson et al., 2011). However, we could only find one study that combined the two methods (Röder et al, 2008). Combining the two methods is important to understand how the composition of species changes after fire, and whether recovered species are native or alien and endangered or not. These are important issues for post-fire ecosystem management and restoration.

Our goal in this study was to compare if both, time elapsed since a wildfire and burn severity influence in the recovery of native cover, species richness and abundance in sites burned by a single fire which took place 10, 20 and 30 years ago. We tested the hypothesis that vegetation cover, and species richness and abundance in the Chilean sclerophyllous forest recovery at similar habitat type and species composition after fire is positively related to the time elapsed since the fire and negatively to its severity. We tested these hypotheses using two approaches: (1) we estimated the percentage of recovered woody vegetation cover using satellite images from 1985, 1995 and 2005, and (2) we conducted field surveys of species richness and abundance of adult and regenerated woody species in the present (2015). We also tested the influence of characteristics of burned area, including slope, altitude, pre-fire vegetation type (shrubland or forest) on vegetation recovery.

2. Methods

2.1. Study area

The study was conducted in the Valparaíso, Metropolitan and O'Higgins Regions in central Chile, which cover the region between 32° 02′ to 34° 45′S, and 69° 40′ to 72° 10′ W (Fig. 1). This area is part of the north-central distribution of Chilean Mediterranean forests. Native vegetation in this area has been strongly modified by economic activities such as intensive agriculture, mining, and industry, as well as the establishment of settlements. The climate of this area is Mediterranean (di Castri and Hajek, 1976) with cold wet winters and warm dry summers (Aschmann, 1984). Together, the three regions have almost 300,000 ha of Mediterranean forests and shrublands (UACh et al., 1999). This area is the most populated in the country, with close to 9.7 million people (INE, 2018). The woody dominant species are endemics, these are *Cryptocarya alba* (Lauraceae), *Peumus boldus* (Monimiaceae), *Quillaja saponaria* (rosaceae) and *Schinus polygamus* (Anacardiaceae), among others. The most common shrubs (that some time have tree habit) are *Lithrea caustica* (Anacardaceae), *Kageneckia oblonga* (Rosaceae) and *Acacia caven* (Fabaceae), this last species is not endemic to sclerophyllous forests. There are not open areas composed by native species in Chilean sclerophyllous forests, then the recovery analysis of sclerophyllous forest does not include open sites that recover to open sites; however, we show it in figures to compare with forest and shrubland recover.

2.2. Satellite image processing

All the fires that occurred in the Chilean Mediterranean forests between 1985 and 2015 were identified and delimited using 60 Landsat images of summer season (30 m of spatial resolution). All images were corrected geometrically, radiometrically and atmospherically (Chuvieco et al., 2002; Chander et al., 2009) using ENVI 4.5 software (ITTVIS 2008). The oldest image we found without clouds in our study area stemmed from 1985, and defined the start of the examined time-window. A maximum likelihood statistic of the supervised classification method (Chuvieco et al., 2002) was used in ENVI 4.5 to classify native forest, schrubland, pasture/cropland, urban areas, exotic plantations, water bodies, bare soil and burned sites. We used 500 training points, stratified by surface of landcover, to classify each Landsat image.

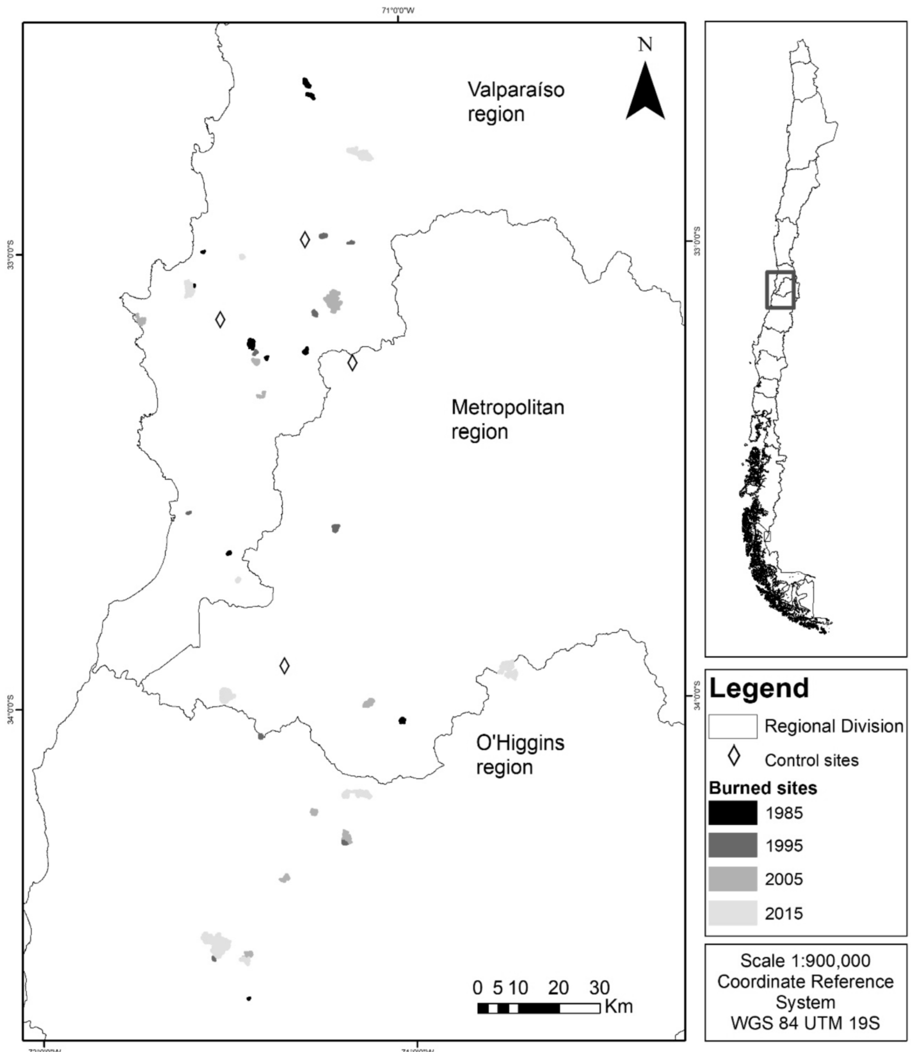

Fig. 1. Study area in central Chile. The size of burned sites was doubled in order to see them better.

These training points were obtained through two sources, a) cadastre of the native forest resources of Chile (CONAF et al., 1999) to classify images prior to 2002, and b) Google Earth (specifically its "time slider") to obtain input to classify images after 2002. The cadastre of native forest was used to obtain the training point for extensive land use/cover (for example, forest, water bodies, agricultural areas). The validation of the classifications was carried out by confusion matrices through 250 points for each image. These validation points were obtained from the same sources of information as the training points. After obtaining land use/cover classifications for each of the considered time-steps we identified the burned sites. We distinguish between areas with vegetation loss due to fires and areas where loss was due to other activities. Usually, the other uses were agriculture and exotic forest plantations. The step followed were: First) false-color images were created from spectral bands responses (R: mid-infrared band, G: infrared band, B: red band), which helped to validate the sites classified as burned sites. Second) we identified the loss of vegetation (e.g., forest or shrubland to burned site) using supervised classification maps through of the spatial analysis tools of ArcGis 10.4 (ESRI 2016). Finally, we evaluated temporally the changes (decrease) in the pixel values of two spectral indices, a) NDVI (Normalized Difference Vegetation Index) (Rouse et al., 1974; Chuvieco et al., 2002; Heilmayr et al., 2016) and b) NDWI (Normalized Difference Water Index) (Gao, 1996), as proxy to spatially discriminate the sites with vegetation loss by wildfires. Subsequent regeneration of the vegetation to shrubland allowed the evaluation of subsequent fires that occurred in that same area through the processes explained above. Additionally, we identified native forest areas which were not affected by fire since 1985 in the land use/cover obtained from the supervised classification. These areas were used to identify control or unburned sites.

We found 1,586 fires that occurred in the Chilean Mediterranean forests and shrublands of the study area during this period. The criteria to select burned sites to be comparable were as follows: (First) we excluded burned sites that had been converted to other land use obtained from the supervised classification in the periods studied (30, 20 and 10 years before 2015) to avoid confusion between land use change and absence of recovery after fire. This greatly reduced the number of sites since most of them that were burned during these periods had changed to other land use, mainly agricultural, (Second) we included only the burned sites located on slopes with west, south, and southwest exposure because sclerophyllous forest is dominant in these habitats (slopes with other exposures are dominated by xerophytic species, which were not considered), and (Third) we included only burned sites located on hills of the Coastal Range. For these activities, a digital terrain model with 30 m of spatial resolution was used. With the resulting set of burned sites, we randomly selected 10 sites burned only once in 1985, 1995, or 2005 (30 sites in total). It was very difficult to find sites not transformed for other uses and where native vegetation remained. In addition, we studied 10 sites burned in 2015 to make a descriptive comparison of these sites with those burned in previous years. These sites had a fire one to two months before we visited the area to sample the species composition. In total we studied forty sites (Fig. 1).

We determined the percentage of vegetation cover in 30 selected sites using the "time slider tool" of Google Earth Pro and ArcGis (10.2), and classified vegetation coverage as dense (75–100% of woody vegetation cover), semi-dense (50–75%), and open (1–50%), burned vegetation, bare soil and other land cover (roads, houses, agricultural crops, bodies of water, among others) obtained from supervised classification of Landsat images (from 1985, 1995 and 2005). Each burned site contains areas of dense, semi-dense and/or open vegetation or a mix of these. We used Google Earth Engine (Gorelick et al., 2017) to download satellite images of each burned site taken one month before the fire, to examine through NDVI if the pre-fire vegetation had been forest (values between 0,35 and 0,50), shrubland (values between 0,1 to 0,3), open vegetation (values ≤ 1), or a mix of these.

The studied sites ranged in size from 11 to 832 ha. Burn severity, altitude, latitude, slope, pre-fire cover (forest or shrubland) and distance to the closest populated centers were determined for all sites (Tables 1 and 2). We used latitude as a proxy of precipitation, because in Chile latitude decreases in almost the same way that precipitation increases (Luebert and Pliscoff, 2006), and there is not complete data for precipitation from 1985 to 1995 period in part of the study area. Average, altitude, latitude and slope were determined by each polygon per site (also minimum and maximum altitude and slope is showed in Table 1) We considered as populated center those with more than 10,000 people (INE, 2017). We also explored whether the size of burned site could be related to vegetation recovery, but decided to not use this parameter, because it correlated positively with altitude, and altitude has been argued to be one of the best descriptors of vegetation recovery (Hogenbirk and Wein, 1991; Keeley, 2009). Distance to the closest populated centers was relevant because gathering firewood from native forests is an important activity among people in small cities and rural areas (INFOR-CONAMA, 2005). Sandoval (2016) found stumps cut by axes or chainsaws in our study sites. However, the analysis showed that the deviance and AIC values are the same with or without this variable.

2.3. Calculating burn severity

Burn severity was calculated using the differenced Normalized Burn Ratio (dNBR) index, which is based on the analysis of two satellite images, one month or less before the fire and the other one month or less after fire. The Normalized Burn Ratio (NBR, Equation (1)) is determined for each satellite image, and the difference between the two indices is the dNBR (Equation (2)). Calculating the NBR and dNBR requires the near-infrared (NIR), short-wave infrared band (SWIR) and thermal band (TB) (Equation (1)), measured by Landsat TM (band 4 and 7, thermal 6)

Table 1
Characteristics of the sites included in the study.

Fire year	Site	Surface (ha)	Minimum altitude (m)	Maximum altitude (m)	Average altitude (m)	Minimum slope (%)	Maximum slope (%)	Average slope (%)	Latitude
1985	1	17.26	141	204	170	0	26	11	−33.004497
	2	235.70	316	560	432	3	45	24	−33.208351
	3	40.85	200	310	255	1	42	22	−33.667749
	4	11.14	406	547	477	0	55	28	−34.648692
	5	27.86	305	408	357	8	35	22	−33.240503
	6	108.78	270	650	460	0	5	25	−32.666036
	7	73.65	640	795	710	5	20	14	−33.227260
	8	152.56	315	850	583	15	45	30	−32.639164
	9	97.83	340	700	520	9	55	29	−34.044993
	10	11.29	312	378	340	0	10	5	−33.076862
1995	1	42.62	495	320	658	9	45	27	−34.55695
	2	102	430	1 180	805	6	58	32	−34.308581
	3	56.85	220	461	312	8	60	30	−34.072881
	4	154.10	195	475	330	12	55	24	−33.617913
	5	20.02	210	220	214	0	10	5	−33.576799
	6	36.46	324	460	392	10	30	21	−32.974816
	7	106.22	340	770	555	35	60	48	−33.228362
	8	46.98	286	757	522	9	57	33	−32.991432
	9	86.94	832	1000	902	0	32	16	−33.144401
	10	84.04	430	1130	780	12	55	34	−34.306991
2005	1	37.13	209	513	361	0	48	24	−33.324541
	2	20.05	295	1256	776	0	63	32	−33.316955
	3	210.43	351	544	448	0	67	34	−33.150952
	4	924.68	285	413	351	0	62	31	−33.117810
	5	206.46	803	1 843	1323	19	78	49	−34.004756
	6	127	330	930	630	0	65	33	−34.241425
	7	196.81	47	456	252	0	52	22	−34.552384
	8	240.39	300	885	593	0	60	30	−34.296670
	9	1331.78	483	825	654	0	61	34	−33.247436
	10	336.27	310	1 140	725	0	60	32	−34.385959
2015	1	614.33	176	645	411	0	44	22	−34.202020
	2	1932.64	187	602	395	0	47	24	−34.524510
	3	162.63	173	489	331	0	36	18	−34.563514
	4	738.08	202	334	261	0	31	16	−33.983437
	5	299.44	450	787	619	0	53	27	−33.951663
	6	555.89	440	1004	725	0	71	36	−33.930514
	7	514.26	195	425	310	0	34	17	−33.08660
	8	68.65	180	227	204	7	29	18	−33.017251
	9	696.11	383	977	681	3	54	28	−32.794016
	10	58.71	245	326	286	2	58	30	−33.727865
Control	1	18.1	233	392	313	6	48	27	−32.981354
	2	75.47	405	453	423	9	20	14	−33.148283
	3	67.05	330	429	383	2	69	35	−33.255328
	4	26.14	199	310	255	5	35	20	−33.913476

Table 2
Distance to closest populated center and dNBR of each study site.

Year	N°	Dist pop. (m)	Burned area (ha)	Burn severity (Mean)	Burn severity (stdDev)	Burn severity (max)	Burn severity (min)
1985	1	0	17	0.047	0.020	0.092	−0.001
	2	2000	236	−0.007	0.009	0.032	−0.036
	3	3500	41	0.063	0.015	0.094	0.009
	4	6325	11	0.032	0.009	0.051	0.009
	5	742	28	0.003	0.006	0.020	−0.012
	6	468	109	0.036	0.015	0.077	−0.006
	7	15,100	74	0.027	0.017	0.065	−0.011
	8	3396	153	0.026	0.016	0.078	−0.023
	9	300	98	0.032	0.014	0.082	−0.008
	10	313	11	0.049	0.027	0.100	−0.029
1995	1	7350	43	−0.009	0.020	0.038	−0.073
	2	1340	102	−0.046	0.023	0.007	−0.108
	3	2796	57	−0.019	0.012	0.018	−0.066
	4	2458	154	−0.017	0.020	0.032	−0.085
	5	889	20	0.010	0.007	0.029	−0.014
	6	2300	36	−0.023	0.017	0.030	−0.078
	7	1500	106	−0.028	0.021	0.025	−0.087
	8	0	47	−0.005	0.017	0.043	−0.050
	9	9871	87	−0.044	0.025	0.005	−0.107
	10	1491	84	−0.036	0.026	0.031	−0.096
2005	1	598	37	−0.034	0.018	0.004	−0.092
	2	716	20	−0.054	0.023	−0.004	−0.100
	3	387	210	−0.045	0.025	0.009	−0.116
	4	0	925	−0.025	0.017	0.027	−0.110
	5	5,845	206	−0.013	0.009	0.009	−0.047
	6	1000	127	−0.039	0.019	0.007	−0.093
	7	60	197	−0.033	0.019	0.006	−0.085
	8	1200	240	−0.046	0.029	0.010	−0.107
	9	603	1332	−0.002	0.017	0.044	−0.068
	10	500	336	−0.022	0.013	0.007	−0.084
2015	1	260	614	−0.053	0.025	0.011	−0.128
	2	0	1934	−0.045	0.025	0.010	−0.133
	3	0	163	−0.037	0.021	0.006	−0.104
	4	200	738	−0.041	0.018	0.021	−0.116
	5	0	299	−0.021	0.014	0.010	−0.120
	6	280	556	−0.032	0.018	0.014	−0.106
	7	0	515	−0.050	0.028	0.021	−0.127
	8	238	69	−0.033	0.016	−0.001	−0.102
	9	4800	696	−0.042	0.030	0.019	−0.134
	10	330	59	−0.046	0.029	0.026	−0.124

and Landsat OLI (band 5 and 7, thermal 11). We used Google Earth Engine tools (Gorelick et al., 2017) to calculate the index processing all available Landsat images for the study area.

The scale of the dNBR index proposed by Key and Benson (2006) was used, which classifies burn severity as low when the dNBR is between 0.1 and 0.27, medium–low between 0.27 and 0.44, medium–high between 0.44 and 0.66, and high over 0.66. We defined burn severity following Keeley (2009). We worked with low and medium–high severity fires in this study because we were not able to completely subdivide the severity of fires into medium or high (Table 2).

$$NBR = \frac{(NIR + SWIR(\frac{TB}{1000})}{(NIR - SWIR(\frac{TB}{1000})} \quad (1)$$

$$dNBR = PrefireNBR - PostfireNBR \quad (2)$$

2.4. Field work

The composition and abundance of plant species were identified in four burned sites for each year 1985, 1995, 2005 and 2015 (16 of the 40 sites studied with satellite images). Four unburned (control) sites were also sampled, these sites had not been burned in more than 30 years (between 1985 and 2015). We chose the closest unburned site to each the burned site, and the average distance between burned and unburned sites was 48.5 km. The most important variables modulating the plant species composition and abundance in the Chilean sclerophyllous forest are the topographic position (aspect and distance to streams) (Armesto & Martínez 1978), climate (determined by latitude and elevation) (Becerra 2016) and disturbances (Fuentes et al. 1984). For that reason, all unburned sites chose had dense vegetation, indicating a low or non-existent recent anthropogenic disturbance, and a similar aspect latitude and altitude than at burned sites. For logistical reasons, the sampled burned sites were the smallest selected in the preceding analysis (between 11 and 32 ha), because the largest burned sites that maintained vegetation until 2015 were inaccessible to field work.

The sites burned in November-December 2015 were sampled one or two months after fire (January 2016). These last sites and unburned sites were studied to compare species composition with the sites burned 10, 20 and 30 years ago. The average slope and altitude and geographical coordinates (using a GPS) of all sites were registered and the linear distance to the nearest populated center was calculated using Google Earth. We checked by charcoal and burned logs before making the transects. In total we found 65 burned logs in transects of all burned sites, but also eight small traces of burned wood at unburned sites.

Within each site, we walked the area and chose a representative place (physionomically similar in species composition and cover) then installed four parallel transects 20 m apart with similar slope (following Smith Ramírez et al 2021). Four parallel transects perpendicular to the slope were established. Each transect was 50 m long by 2 m wide. All individuals of woody species two meters tall or taller in the transects were counted, as well as woody regeneration (individuals less than 2 m tall) in ten plots of 0.25 m^2 (0.5 × 0.5 m) separated by 10 m from each other along each transect. Four transects and 10 plots per transects resulted in 40 plots per site, which was twice the number established by Gómez-González et al. (2011) by Chilean sclerophyllous forests. The composition of herb species and the percentage cover per species in every plot was also registered. The composition of herbaceous species in sites burned in 1985 and 1995, and in unburned sites were registered between October to November 2015. Unfortunately, the sites burned in 2005 and 2015 were sampled late spring (December 2015 and January 2016) when many herbaceous species were unrecognizable because drought. Samples of unidentified woody species in the field were collected and identified in the lab. The composition and abundance of woody and herbaceous species in the burned and unburned sites are shown in Appendix S1 and S2. Visiting sites allow us confirm visually that dense vegetation was comprised mainly by trees in overstory, open vegetation by scattered shrubs, and semi-dense vegetation was composed of a mix of both but less scattered than in open areas.

2.5. Statistical analysis

2.5.1. Analysis of satellite image

Generalized linear models (GLM) were used to evaluate the effect of time elapsed (T) since the last fire (10, 20, 30 years), burn severity (Se) as measured by dNBR, slope (pronounced, moderate and slight) (Sl), altitude (high, intermediate and low) (A), pre-fire vegetation type (shrubland or forest) (V) and latitude by every 0.5° (L) (Tables 1 and 3), on the percentage of vegetation cover. The response variables were the percentage of dense vegetation (in relation to total area of the burned site) (Dv), the percentage of semi-dense vegetation (Sdv) and the percentage of open vegetation (Ov). A binomial error distribution and a logit link function in GLM were used because these are recommended when the response variable is a percentage (Cayuela 2015). Since the burned sites were not the same size, we used percentage vegetation cover as a response variable of total area of dense, semi-dense and open vegetation of each burned site, in order to make a comparison among sites. The Akaike Information Criterion (AIC) was determined, which is an index that indicates the degree of fit of the model and allows comparing the different models. We chose the model with the lowest AIC, which indicates the best fit. We also calculated the deviance (D^2), which indicates the percentage of variance explained by the model. The GLM models used (Table 3) were:

Model a: $Y = B_0 + B_1 * T + B_2 * Se + B_3 * Sl(mean) + B_4 * A(mean) + B_5 * V + B_6 * L$

Model b: $Y = B_0 + B_1 * T + B_2 * Se + B_3 * Sl(min) + B_4 * A(min) + B_5 * V + B_6 * L$

Model c: $Y = B_0 + B_1 * T + B_2 * Se + B_3 * Sl(max) + B_4 * A(max) + B_5 * V + B_6 * L$

Where $B_{0,1,2,3,4,5}$: estimated coefficients, Y: percentage of Dv or DSv or Ov, min: minimum and max: maximum.

2.5.2. Field work

A second set of generalized linear models (GLM) were used to evaluate the effect of time elapsed since the fire (10, 20 and 30 years) (T), altitude (A), and latitude (L) (Table 3) on: (a) richness of adult woody species, (b) abundance of adult woody species, (c) richness of regenerated woody species, (d) abundance of regenerated woody species, (e) richness of native herbaceous species, (f) richness of alien herbaceous species, (g) herbaceous cover of native species (%), (h) herbaceous cover of alien species and. For e - h, we analyzed the effect of time elapsed since the fire 20 and 30 years ago, as there was no information from 2005. In the case of the richness and abundance variables, we used a Poisson error distribution and a logarithmic link function in GLM, as these variables were counted totals. A binomial error distribution and a logit link function were used for herbaceous cover (Cayuela 2015). The models were:

Model a: $Y = B_0 + B_1 * T + B_2 * A + B_3 * L$

Model b: $Y = B_0 + B_1 * T$

Where $B_{0,1,2,3}$: estimated coefficients, Y: richness or abundance of adults and regenerated woody species, or herbaceous cover.

The Jaccard similarity index was calculated to assess the similarity in species composition between burned and unburned sites in the composition of woody species (adults and seedlings) and herbaceous species. A value 1 means more similarity in species composition (Real and Vargas, 1996). The analysis of variance test (ANOVA) or Kruskal-Wallis test (if the assumptions of the ANOVA were not met) were applied to identify significant differences between unburned and burned sites in richness and cover of native and exotic species. All statistical analyses were performed with RStudio Team (2020).

Table 3
Predictor variables included in GLM analyses (satellite imagen analysis and field work).

Predictor variable	Number of classes	Range	Unit	Spatial analysis	Field work
Time elapsed since the last fire	3	10 to 30	Years	X	X
Burn severity	2	Low or Medium-high	–	X	
Pre-fire vegetation type	2	Shrubland or Forest	–	X	
Slope	–	0 to 78	Percentage	X	
Altitude	–	47 to 1843	Meters	X	X
Latitude	–	34.6 to 32.6	Decimal Degrees	X	X

3. Results

3.1. Satellite image processing

The results of the validation of the all-land use/cover maps indicated an average of 87.8% classificatiońs accuracy. The land use/cover that obtained the highest 3.1.st accuracy was the water bodies with 97.3%, followed by crop areas with 93.8%. Conversely, the coverage that obtained a lower precision was shrubland with 81.4%, followed by urban areas with 86.9%.

The coverage recovered after fire from 0% to semi dense and dense vegetation was 3090.9 ha (95.7% of the total burned area) in the total of 30 sites burned in 1985, 1995 and 2005. The largest percentage recovered vegetation area was in the sites burned in 1985, followed by sites burned in 1995 (Fig. 2). Approximately 65% of the recovered area of dense vegetation corresponds to sites that had a pre-fire forest cover, while the remaining 35% corresponds to pre-fire shrubland cover. As well, 43.5% of burned vegetation classified as shrublands before the fire had recovered as dense vegetation (mainly forest) by 2015. Close to 50% of burned shrubland recovered to semi-dense vegetation (a mix of trees and shrubs), while some sites did not recover at all.

The percentage of dense vegetation was significantly higher with

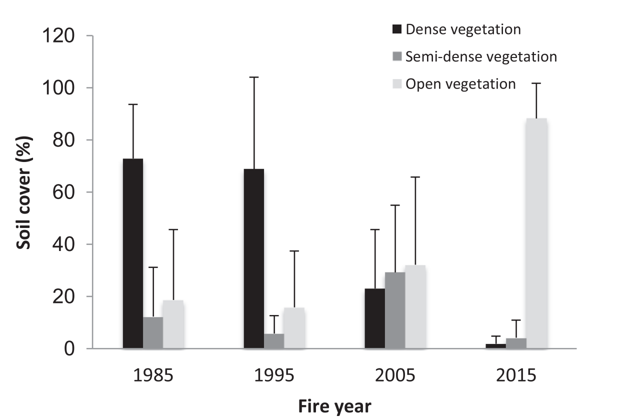

Fig. 2. Percentage of vegetation cover recovered in 2015 (Mean ± Standard deviation) in sites burned once in 1985, 1995, 2005 and 2015.

more time elapsed since the fire, while semi-dense and open vegetation cover was found more frequently in the more recently burned sites (Fig. 2, Table 4). No significant relationship was found between burn severity and the current cover of dense and semi-dense vegetation. However, the proportion of open vegetation cover was greater with more severe fires (Table 4). The percentage of dense vegetation cover was significantly higher on slighter slopes, at higher altitudes and latitudes, and at sites with forest as pre-fire cover. The percentage of semi-dense vegetation cover was significantly higher on more pronounced slopes, lower altitude and more northern latitudes. The percentage of open vegetation cover was significantly higher at higher altitudes and latitudes, and at sites with shrubland as the pre-fire cover (Table 4).

3.2. Field work

A total of 51 adult native woody species were found in the transects of the four classes of burned sites (1985, 1995, 2005 and 2015). A total of 842 trees, 365 shrubs and 173 subarboreal individuals were found (Appendix S1). One alien tree species and two alien shrub species were found, represented by 21 individuals in total. The most abundant tree species were *Peumus boldus,* followed by *Cryptocarya alba*. The most abundant shrub species were *Baccharis linearis* followed by *Cestrum parqui.* The most abundant subarboreal species were *A. celastrina,* followed by *Luma chequen.* Only one individual was registered representing the species *A. petiolaris*, *Baccharis concava*, *Blepharocalyx crukshanskii*, *Calceolaria thyrsiflora*, *Eriosyce* sp., *Myrceugenia correifolia*, *Retanilla ephedra* and *Teucrium bicolor*. At unburned sites, 27 woody adult species, 217 trees, 76 shrubs and 16 subarboreal were registered (Appendix S1) and no exotic woody species were found. The most tree abundant species as burned sites were *C. alba,* followed *P. boldus*. The shrubs were mainly represented by *C. odorifera* followed by *Eupatorium salvium.* The only two subarboreal species were *A. celastrina* and *Escallonia pulverulenta,* with eight individuals each.

With native woody seedlings, 35 species and 255 individuals of tree species, 31 individuals of shrub species and 39 individuals of subarboreal species were found in the 16 burned sites (Appendix S2). One alien seedling was found (*Pinus radiata*). The most abundant seedlings were *C. alba* followed by *P. boldus*. The most abundant subarboreal species were *Luma chequeen* followed by *Escallonia pulverulenta* (Appendix S1). At unburned sites, 15 species of woody seedlings were registered, 70 of tree species, 16 of shrub species and two of subarboreal species (Appendix S2). Only one seedling of an alien tree species was found (*Crataegus monogyna*). The most abundant tree species were *C. alba* followed by *P. boldus.* The most abundant shrub species was *Colliguaja odorifera.*

The abundance of adult native woody species in sites burned 30, 20, 10, 0 years ago and unburned sites was 30 individuals (Mean ± Standard Deviation: 123.3 ± 47.5), 35 (87 ± 29.1), 30 (82 ± 18.1), 29 (66.3 ± 7.4) and 29 (79.3 ± 22), respectively. The numbers (abundance) of woody seedlings in sites burned 30, 20, 10, 0 years ago and unburned sites were 20 (95 ± 50.5), 21 (67.5 ± 63.6), 9 (23.8 ± 6.1), 11 (14.3 ± 14.2) and 16 seedlings (60.5 ± 47), respectively.

Overall, at burned and unburned sites, there were 114 herbaceous taxa (species and subspecies), of which 64 were native (56.14%) and 50 were alien (43.86%). We found 35 (29.5 ± 5.9), 47 (30.5 ± 3.7) and 39 (31 ± 12.2) native herbaceous taxa at the sites burned 30 and 20 years ago and at unburned sites, respectively, as well as 27 (25 ± 2.71), 36 (25 ± 19.2) and 34 (27.33 ± 19.5) alien herbaceous taxa at the sites burned 30 and 20 years ago and at the unburned sites, respectively (Appendix S2). Native and alien herb species richness did not differ significantly between unburned and burned sites (ANOVA, F = 1.1596, df = 2, p = 0.32 and F = 0.062, df = 2, p = 0.94, for native and alien species, respectively). The cover of native herbaceous species also did not differ significantly among sites burned at 1985 (25.8 ± 7.4 %), 1995 (22.8 ± 5.2%) and unburned sites (28.6 ± 5.9 %) (ANOVA F = 0.88, df = 2, p = 0.45). Similarly, the cover of alien herbaceous species did not differ significantly among sites burned in 1985 (23.1 ± 6.1 %), 1995 (22.3 ± 14.4 %) and unburned sites (30.3 ± 11.6%) (Kruskal-Wallis chi-squared = 2.58, df = 2, p = 0.28) (Appendix S2).

No significant relationships were found between species richness of adult woody plants and time elapsed since the last fire, altitude and latitude (Table 5). However, the richness of woody species regeneration was greater in sites burned in 1985 and at higher latitudes (Table 5). The abundance of adult woody plants was significantly and positively related to altitude, while the abundance of the regeneration of woody species was significantly and positively related to the time elapsed since the last fire, altitude and latitude (Table 5). No significant relationships were found between herbaceous richness of native and alien species and time elapsed since the last fire, altitude and latitude (Table 5). Herbaceous cover of native species was greater at higher altitudes and lower latitudes. Herbaceous cover of alien species was greater with less time elapsed since the fire, and at lower altitudes and latitudes (Table 5).

The Jaccard's Index of Similarity of adult woody species, woody seedlings and herbaceous seedlings shows, that the similarity in species composition among burned sites with unburned or control sites was higher as more time had passed since the fire (Table 6).

4. Discussion

Our results show that forest/shrubland and forested vegetation was able to recover after one fire over a period between 10 and 20 years post-fire, in Chilean sclerophyllous forests. After a period of this length, there

Table 4
Significant variables of the GLM analysis (p < 0.05) on the percentage of vegetation recovery after fire. T: time elapsed since the fire disturbance, Se: Burn severity, Sl: slope, A: altitude, V: pre-fire vegetation type (shrubland or forest), L: latitude, AIC: Akaike Information Criterion, D^2: deviance, min: minimum and max: maximum.

Percentage of vegetation cover	Model	Significant terms	AIC	D^2
Dense vegetation	b	(+0.1380) T + (-0.0166) Sl (min) + (+0.0028) A (min) + (-1.192) V + (+0.0038) L	4482.1	39.12
Semi-dense vegetation	b	(-0.0466) T + (+0.0483) Sl (min) + (-0.0023) A (min) + (-0.0069) L	3108.8	19.39
Open vegetation	c	(-0.1363) T + (+0.3634) Se + (+0.0011) A (max) + (+1.339) V + (+0.0039) L	2899.9	43.85

Table 5
Significant variables of the GLM analysis (p < 0.05) of the woody adult and seedling richness and abundance, and herbaceous richness and cover. T: time elapsed since the fire disturbance, A: altitude, L: latitude, AIC: Akaike Information Criterion, D^2: deviance

Response variable	Model	Significant terms	AIC	D^2
Richness of adult woody species	–	–	–	–
Richness of woody species regeneration	a	(+0.0426) T + (+0.0035) L	154.22	44.88
Abundance of adult woody species	a	(+0.001) A	321.55	19.91
Abundance of woody species regeneration	a	(+0.0723) T + (+0.0018) A + (+0.0082) L	541.46	46.01
Richness of native herbaceous species	–	–	–	–
Richness of alien herbaceous species	–	–	–	–
Herbaceous cover of native species	a	(+0.0012) A + (-0.0025) L	265.71	20.59
Herbaceous cover of alien species	a	(-0.0303) T + (-0.0018) A + (-0.0054) L	336.87	24.77

Table 6
Jaccard similarity index (*j*) in species richness between burned and unburned sites. Vs = versus

Species	Jaccard similarity index (%)			
	1985 vs unburned	1995 vs unburned	2005 vs unburned	2015 vs unburned
Woody adults	54.05	52.38	46.15	40.48
Woody seedlings	45.83	25.00	26.32	23.81
Herbaceous cover	52.08	40.98	–	–

was also a decrease in shrubland and forest/shrubland vegetation in burned sites, which were replaced by forested cover. However, richness and abundance of adult woody species at burned sites were not related to the time elapsed since the fire, probably because many species and individuals re-sprout rapidly after a fire and the time scale analyzed (every 10 years) was too wide to detect differences at this level. Nevertheless, the Jaccard Index showed more similarity in the species composition of burned sites as compared to unburned sites, in both adult individuals and regeneration of woody species. Sites burned 30 years ago had more species richness than unburned sites, and similar species richness to sites burned 20 years ago, as is expected by intermediate disturbance hypothesis (Connell 1978).

Contrary to what we expected, regeneration of vegetation cover was not related to the severity of fires. In a literature review, Keeley (2009) showed that there is not a general pattern of the effect of fire and burn severity on vegetation recovery; this relation depends mainly on the species and ecosystem rather than on the severity or intensity of the fire. We found that other variables had a greater influence on a site's percentage of recovery after fire; for example, sites where pre-fire vegetation had had forested vegetation were able to recover in higher proportion than sites that were previously shrublands. Probably this occur because forested vegetation areas are less disturbed than shrublands and may be soil, biological and microclimate characteristics are different. The re-sprouting of vegetation has been shown in several species within Chilean sclerophyllous forests by Montenegro et al., (1983, 2003, 2004).

Burned sites at higher altitudes presented more abundance of both tree species and seedlings, which was also found by Smith-Ramírez et al. (2011) in the same area, however, it was unclear if the sites were burned or. Higher altitude sites typically have higher soil moisture and less human disturbances, and thus may recover more rapidly than low elevation sites exposed to more human disturbance (Becerra et al. 2016). This trend is expected to increase worldwide with climate change (Mondoni et al. 2020). On the other hand, abundance and species richness in burned sites were also positively related to latitude; since precipitation is greater at higher-latitude sites (www.ine.cl), this could suggest the positive impact of precipitation on dryland regeneration after fire (Hinojosa et al. 2019). This pattern has been observed in other studies of sclerophyllous forests in central Chile, but without specification of whether the sites had been burned or not (Fuentes et al. 1989, Becerra, 2016). Also, forested vegetation at burned sites recovered better after disturbance on slight slopes than on steep ones, which could relate to soil-moisture water loss through runoff, which is more rapid on steeper slopes.

Some studies have documented in central Chile that native forests either do not recover after fire or the process is very slow (Fuentes et al., 1989; Fuentes-Castillo et al., 2012). However, according to our results, burned sites seem to recover between 10 and 20 years after one fire and reach high levels of forest cover, when there was indeed a forest before the fire. We observed sprouts even two months after a fire at sites evaluated in 2015 (authors' personal observation). On the other hand, most of the burned sites studied had unburned remnant vegetation found throughout the burned vegetation, which would have served as a source of seeds. However, recovery by seeding and sexual regeneration seems to be much poorer than by re-sprouting (Becerra et al., 2018). Other factors that can also reduce the ability of vegetation in Chilean sclerophyllous forests to recover after fire are the occurrence of more than two successive and short-interval fires (Smith-Ramírez et al. 2021). Herbivory by exotic species like cattle, rabbits, and horses can also reduce post-fire regeneration (Moreno and Oechel, 1991, Fuentes et al., 1984, Holmgren, 2002, Becerra et al., 2018).

Although our results suggest Mediterranean vegetation was relatively resilient after single fire, it is unclear if the recovery will continue to occur at the same rate in the future. Our study spans a period with three strong El Niño events, called super-Niños, that struck the Pacific coast of South America, resulting in considerably more rain that during La Niña dry periods (see the Climate Prediction Center, link: https://www.cpc.ncep.noaa.gov/products/precip/CWlink/MJO/enso.shtml). This important input of water could have facilitated recovery during the post-fire period, as has been well documented for European Mediterranean forests (Gouveia et al., 2012). However, the last decade was the driest in central Chile in the past century (González-Reyes, 2016), and if this condition continues as a consequence of climate change, it is probable that forest regeneration in this region will be slower.

We found that woody species cover after fire in our study sites was mainly due to native species, while exotic woody species were uncommon. In contrast, native and alien herbaceous species recovered their cover and richness with similar rapidity after fires. Gomez-Gonzalez et al.'s (2011) results concur with our results. Those authors compared the percentage of native and alien herbaceous species in recently burned versus unburned plots in the Valparaiso Region (which includes the northern part of our study area) and found that native and alien species both maintained the same percentages of total cover over time after a fire. It is notable that alien herbaceous species were highly dominant even in unburned sites, which suggests previous disturbances in these sites that facilitated the invasion of exotic herbs.

Almost all related articles that have appeared in the last fifteen years have used satellite images to analyze post-fire vegetation recovery, without determining in the field what species are recovering (Bastos et al., 2011, Tessler et al., 2015; Paci et al., 2017; Gouveia et al., 2018 among others), with the one exception of a study by Röder *et al.* (2008). We argue that it is entirely necessary to include field study in any multitemporal spatial analysis of post-fire vegetation recovery. Field work allowed us to identify seven threatened species (among them *Beilschmiedia miersii*, *Blepharoclayx cruckshanksii*, *Citronella mucronata* and *Drimys winteri* var. *chilensis*), in sites burned 30 and 20 years ago and in the unburned areas but none in sites recently burned in spite of our observation that these species can sprout after fire in other areas (authors pers. observation). Focal studies are necessary to know how to recover these species or whether it is even possible in some areas, given recent climate trends.

We found that the resiliency of Chilean Mediterranean vegetation is similar to that of other Mediterranean-type regions in California (US), Europe, South Africa and Australia. Although we found that Chilean Mediterranean woody vegetation is resilient to fire, this is not commonly observed in the landscape of central Chile because of rapid changes in land use. In fact, we found very few burned sites with no change in land use after a fire.

In conclusion, the woody Mediterranean vegetation in central Chile is returning to pre-fire levels, unrelated to the severity of the fire. The time elapsed after fire, pre-fire cover, and altitude and latitude, but not burn severity, are the most important drivers in this recovery. Recovery of forest/shrubland to forested vegetation occurs between 10 and 20 years after a fire, and species richness and abundance recover around 20 years post-fire.

CRediT authorship contribution statement

Cecilia Smith-Ramírez: Conceptualization, Methodology, Supervision, Investigation, Validation, Writing – original draft. **Jessica Castillo-Mandujano:** Conceptualization, Methodology, Data curation, Writing – original draft. **Pablo Becerra:** Conceptualization, Methodology, Investigation, Validation, Writing – original draft. **Nicole Sandoval:** Investigation. **Rodrigo Fuentes:** Data curation, Writing – original draft. **Rosario Allende:** Investigation, Writing – original draft. **María Paz Acuña:** Data curation, Writing – original draft.

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.

Acknowledgments

We thank the owners of the studied sites and the field assistants. The English was edited by Dr. Lafayette Eaton, George Montgomery, Denise Fliegel and Emma Gleeman. This study is part of "Centro de Investigación de Fuego y Resiliencia de Sistemas Socioecológicos (FireSes)" of the Universidad Austral de Chile. I am grateful for comments from anonymous reviewers and the editor of this journal.

Funding

This work was supported by National Forestry Corporation (CONAF) [project FBN 007/2013], the Institute of Ecology and Biodiversity (IEB) [grant numbers PFB-23, AFB 170008, AFB 210006 and ACE210006). and CAPES FB [0002 – 2014 P.B]. Pablo Becerra also thanks ANID [PIA/BASAL FB0002].

Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.foreco.2021.119800.

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