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Mapping alien and native forest dynamics in Chile using Earth observation time series analysis

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ABSTRACT

Chile is a global biodiversity hotspot and hosts a large proportion of the southern hemisphere's temperate forests. The Chilean Valdivian temperate forest is a vulnerable ecosystem containing a highly ecologically valuable species assemblage. Productive forest plantations have involved deforestation of, and alien species introduction into, this ecosystem. This process has already severely impacted the western part of Chile (the Coastal Range) and is now occurring in the eastern part (the Andes), with forestry plantations promoted by government subsidies between 1974 and 2012. Archive Landsat satellite imagery classification and Google Earth Engine are used to assess land cover change over a 31-year period with a focus on alien species (Pinaceae and *Eucalyptus* spp.) spread and native (*Nothofagus* spp. and *Araucaria araucana*) deforestation. Results show a clear land cover pattern based on elevation: a higher altitude, relatively undisturbed area dominated by native forest (the Andes), and a lower altitude area where most human activity and related land covers are located (the valley area). The valley is highly dynamic because of constant land cover change due to forestry. Overall, *Araucaria araucana* cover has decreased over the study period, while *Nothofagus* spp. has remained relatively stable. Alien Pinaceae has decreased, while *Eucalyptus* spp. has remained stable. However, the results indicate that change analysis over long periods conceal dynamism. For example, *Eucalyptus* spp. sharply decreased between the 1980s and the 1990s and surged afterwards. Also, even though *Nothofagus* spp. cover dominates throughout the study period, change analysis shows a high degree of change in the valley area, indicating newly established *Nothofagus* spp. patches. Over the study period, long rotation Pinaceae plantations for timber have given way to shorter rotation forestry (alien *Eucalyptus* spp., native *Nothofagus* spp.) for pulp and local uses resulting from discontinuation of forest subsidies. In the absence of subsidies, only large-scale plantations can engage in long rotation forestry, as smallholders need the more stable income provided by shorter rotation forestry. Although higher elevations (the Andes) are dominated by native forest, several abandoned alien forest plantations may be the source of biological invasion. In addition, native forest degradation as a result of *Araucaria araucana* loss and shrub encroachment is occurring. Earth observation methods are key for forest and alien species monitoring and landscape management. They can enhance traditional, ground-based forest surveys and provide continuous and even retrospective monitoring of forest change thanks to the wide availability of current and historical satellite data.

1. Introduction

The Chilean Valdivian temperate forest is a global biodiversity hotspot, containing a highly ecologically valuable species assemblage. It is also a fragile and vulnerable ecosystem, which has recently experienced rapid rates of deforestation and habitat loss (Myers et al., 2000, Salas et al., 2016, Miranda et al., 2017), being further threatened by alien species introduction for forestry (Echeverría et al., 2006, Heilmayr et al.,

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2016, Locher-Krause et al., 2017). The rate of temperate forest loss in Chile has reached 4.5% per year in areas of the country such as the Coastal Range, which is already severely impacted by native forest deforestation, fragmentation and alien species introduction (Echeverría et al., 2006, Miranda et al., 2017). A principal factor influencing habitat loss has been the forestry industry driving large-scale expansion of non-native (alien) plantations (Miranda et al., 2015), with many of the alien tree species planted being invasive in nature (Pauchard and Alaback, 2004). Alien forest plantations have steadily increased in Chile since the 1970s (Miranda et al., 2015) due to their high profitability, coupled with national forest policy and government subsidies incentivising plantations (Niklitschek, 2007, Altamirano et al., 2013).

The native Chilean Valdivian temperate forest is especially sensitive to species introduction due to its evolution in isolation between the biogeographical barriers of the Andes and the Pacific Ocean (Alpert et al., 2000, Altamirano and Lara, 2010, Donoso and Romero, 2020). This, together with the ongoing disturbances from forestry, pose a significant risk of biological invasion to this fragile ecosystem by alien tree species (Marvier, Kareiva and Neubert, 2004), with impacts on native flora and particularly the endemic tree *Araucaria araucana* (monkey puzzle), of high concern. Classified as endangered in the IUCN Red List of Threatened Species (Premoli, Quiroga and Gardner, 2013), *A. araucana* is recognised as a Natural Monument by Chilean law due to its rarity and cultural value (Chilean Ministry of Agriculture, 1990).

Productive forestry is a highly successful economic sector in Chile, being a cornerstone of the country's rapid economic growth and its second most important export after copper mineral products (Salas et al., 2016, The observatory of economic complexity, 2021). The most widely used species in productive forest plantations are from the Pinaceae family and the *Eucalyptus* genus. Both are generally highly productive, fast growing and shade intolerant trees, characteristics that are shared by many invasive species (Dodet and Collet, 2012). The Pinaceae family contains a high proportion of invasive species (12%) compared to other families (Richardson and Rejmánek, 2004), and can colonize and expand rapidly in most environments. There is evidence of the spread of propagules from commercial plantations to native forests (Peña et al., 2008), threatening *A. araucana* regeneration due to competition from alien saplings (Pauchard et al., 2014). Once established, these alien plants are challenging to eradicate, with this only realistically possible during early stages of invasion (Schmiedel et al., 2016).

The Chilean National Forest Corporation (CONAF) created the Land Use and Vegetation Resources Cadastre in 1997, and updates it roughly every 10 years, depending on location. This is a large scale and resource intensive effort, which aims to represent the vegetated land covers of the country (CONAF, 2021). However, invasion monitoring and rapid, targeted management are the most effective tools for invasive species control (Pauchard et al., 2016, Nuñez et al., 2017), with forest management at both local and landscape scales essential to control these invasive species (Salas et al., 2016, Sitzia et al., 2016). Consequently, the periodicity and scale of the CONAF cadastre may be unsuitable for targeted alien species management. This alien species monitoring is difficult in Chile using conventional ground-based surveys due to the large geographical extents involved and challenging topography in many parts of the country. In this context, Earth observation (EO) is a highly valuable tool, enabling continuous monitoring of Chilean forests over large areas, early invasion detection and modelling (Bradley, 2014), not possible using ground-based surveys alone. Additionally, repeat monitoring via EO can assess the consequences of land management decisions over time (Pettorelli et al., 2014), with the Landsat satellite imagery archive dating back as far as the 1970s for some locations (Wulder et al., 2016); this is especially useful for evaluating the historical spread of alien plantations and native forest loss.

Previous EO studies have assessed temperate forest loss in Chile using temporal series of land cover maps, but few assess the influence of alien forest plantations on landscape dynamics (Zamorano-Elgueta et al., 2015). In addition, they are generally limited in their spatial and temporal scales (Echeverría et al., 2006, Locher-Krause et al., 2017), or are focused on the intensely exploited Coastal Range such as studies by Echeverría et al. (2006) and Altamirano et al. (2013). The Andes retain a more pristine condition than the Coastal Range, however it is experiencing an increasing presence of productive forest plantations which are spreading into native forest in some areas (Peña et al., 2008, Langdon, Pauchard and Aguayo, 2010).

The Google Earth Engine (GEE) cloud computing platform (Gorelick et al., 2017) enhances the potential of EO methods, allowing users to perform resource intensive tasks using Google's computing capability cost free for non-commercial activities. This is particularly valuable for conservation activities which are often hindered by lack of resources, as GEE eliminates the need for specialised software and computing facilities to perform computationally intensive analyses. GEE also contains an ingested data catalogue of multi-source satellite imagery and ancillary datasets, overcoming common data management and accessibility challenges (Gorelick et al., 2017) when processing extensive EO datasets.

Despite its potential, EO is an underutilized tool in invasion ecology (Bradley, 2014). This study utilises the potential offered by EO and fills a gap in current knowledge by monitoring both current and historical patterns of invasion. The overarching aim of this study is to investigate the dynamics of alien tree spread and native forest loss in the understudied Araucanía region of the Chilean Andes and Andean foothills over the last four decades. To achieve this aim, the following objectives are defined: (1) determine the distribution of native and alien plantation forested areas across four time intervals from the 1980–2010s; (2) investigate the potential invasion process (spatial spread) of Pinaceae and *Eucalyptus* spp. and deforestation of the native forests using change detection analysis, and; (3) compare and explain alien spread and forest loss processes. The research outputs generated are valuable for identifying current and historical patterns of invasion to better inform forest management and conservation planning to prevent further damage to native forests.

2. Materials and methods

2.1. Study area

The study area (Fig. 1) is located to the east of the Araucanía region, in the south-central Andes, Chile. The area includes the Malalcahuello National Reserve and Malalcahuello village and surrounds. It covers approximately 14,000 km^2 across elevations ranging from 500 m to 2800 m. The temperate forests present are composed mainly of the *Nothofagus* genus and the endemic *Araucaria araucana*.

The study area contains several protected native forest areas: Malalcahuello, Nalcas, China Muerta and Malleco National Reserves, Conguillío and Tolhuaca National Parks, and parts of the Ralco and Altos de Pemehue National Reserves, and Villarrica National Park. Also, the large UNESCO Araucarias Biosphere Reserve covers part of the study area.

In the nearby Araucanía Coastal Range and Central Valley, 50% of native forest loss between 1986 and 2008 was due to the establishment of alien forest plantations (Altamirano et al., 2013), with Echeverría et al. (2006) also finding the same conversion between 1975 and 2000 in the Coastal Range in south-central Chile. Although the Andean part of the country is less exploited for commercial forestry, its profitability, plus promotional government policies starting in 1974, has encouraged the expansion of alien forest plantations throughout Chile. Between 1969 and 1970, more than ten alien coniferous species were introduced in the Malalcahuello National Reserve (Peña et al., 2008), with *Pinus contorta*, an alien species typically used in productive forest plantations, since reported as spreading into native forests here (Peña et al., 2008).

The overall study area is divided into two altitudinal zones: the low elevation valley area below 600 m, and the high elevation Andes area above 600 m. A third zone contained only protected areas, with all these

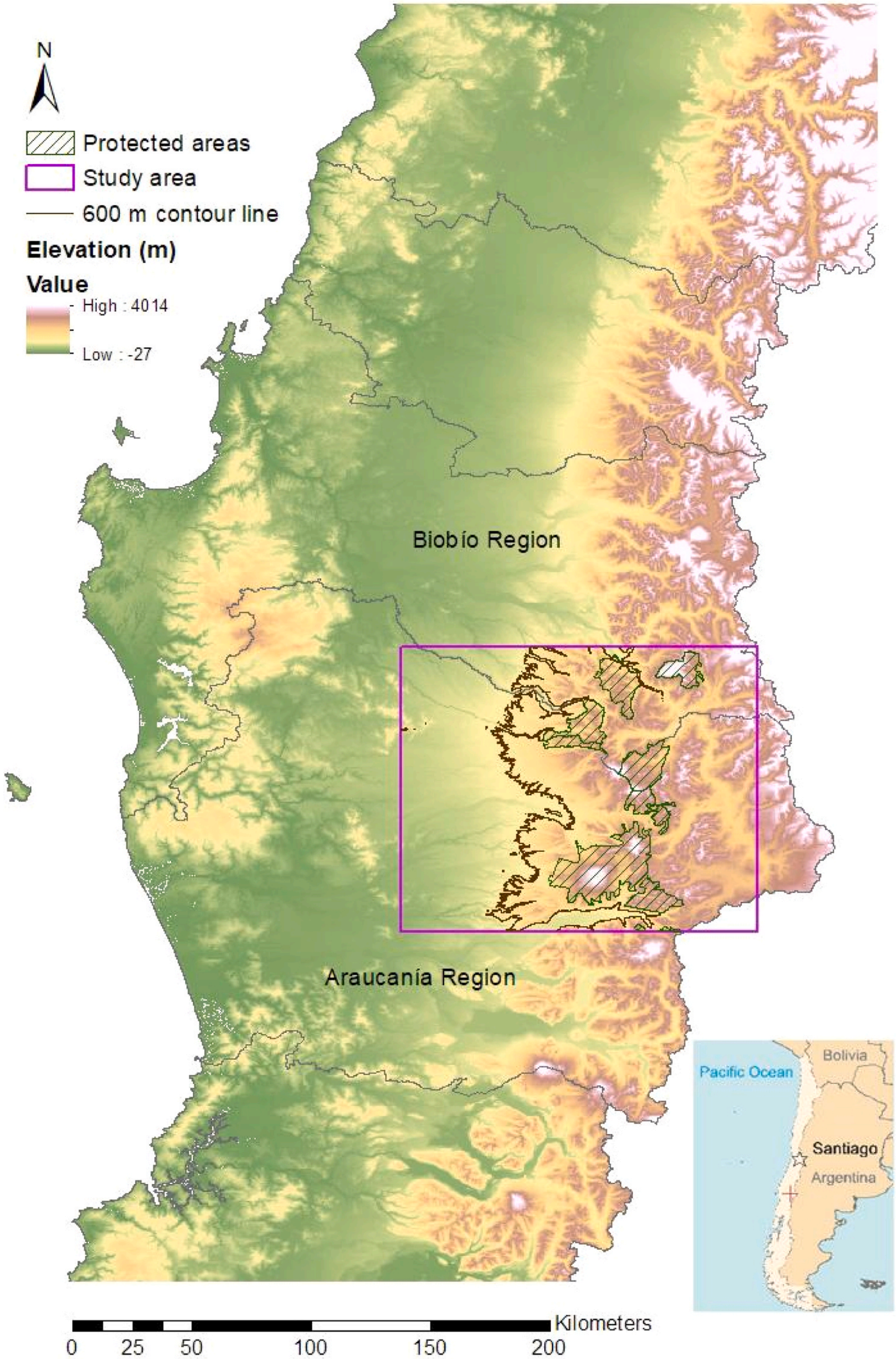

Fig. 1. Study area location. Data from the Chilean Library of National Congress (BCN), the Geospatial Data Infrastructure (IDE) from the Chilean Ministry of National Goods (BCN, 2020, Chile, 2020) and NASA (NASA JPL, 2020).

areas located at high elevation >600 m and nested within the high-elevation zone. An elevation threshold of 600 m was used to separate altitudinal zones as this is the minimum elevation at which *A. araucana* generally grows (Premoli, Quiroga and Gardner, 2013). Most anthropogenic activity is concentrated in the lower elevation valley area, where the major urbanised and agricultural areas are located.

2.2. Study species

This study focuses on the dominant native and alien species present in the Valdivian temperate forest within the Malalcahuello area. Native species include *Nothofagus* genus species (*N. dombeyi, N. pumilio, N. antarctica, N. obliqua* and *N. alpina*), both deciduous and evergreen, which form mixed forests at medium elevations. Higher elevations comprise *Araucaria araucana* and *N. dombeyi*, while highest elevations are dominated by open *A. araucana* forest with a native shrub layer.

The alien species comprise several members of the Pinaceae family and *Eucalyptus* genus commonly used in productive forestry in Chile. Here, the focus is on functional groups rather than individual species, as each group (Pinaceae and *Eucalyptus* spp.) have similar ecological characteristics. The Pinaceae group mainly comprises *Pinus radiata, P. contorta, P. ponderosa, P. sylvestris and Pseudotsuga menziesii*, with *P. radiata* and *P. contorta* most widely planted. *P. radiata* is the most abundant planted alien tree in Chile, occupying a planted area of 1.46 million hectares (Pauchard et al., 2014), and reported to be invading native *Nothofagus alessandrii* forests in central Chile (Bustamante and Castor, 1998). *P. contorta* encroaches in open areas (Pauchard et al., 2016) and has spread rapidly in treeless steppes in the Chilean Patagonia (Langdon, Pauchard and Aguayo, 2010), altering microenvironmental conditions and negatively impacting native plant biodiversity (García et al., 2023). In addition, it is a competitor of the endangered *A. araucana* (Gundale et al., 2014, Pauchard et al., 2014) and its presence is correlated with that of native species *A. araucana* and *N. antarctica* which form medium to low density forests at high elevations. Most *Pinus* spp. are shade-intolerant, hence posing a similar threat to native open forests such as the high elevation *A. araucaria* forests. In fact, *P. radiata* has been unsuccessful at invading native closed *Nothofagus* spp. forests in the Coastal Range, being restricted to only forest edges (Bustamante and Simonetti, 2005).

The most widely planted *Eucalyptus* spp. trees in the area are *Eucalyptus globulus* and *E. nitens*. These species are not formally listed as invasive, but exhibit characteristics common among invasive plants, such as fast growth and low shadow tolerance (Dodet and Collet, 2012). They are extensively planted in Chile, playing a role in landscape change and native forest loss (Echeverría et al., 2012, Altamirano et al., 2013, Heilmayr et al., 2016).

2.3. Data sources

This study utilises the historical archives of Landsat 5 and 8 satellite imagery hosted in the GEE image catalogue. The GEE cloud computing platform enables users to process collections of imagery, rather than just individual images. This has the benefit of enabling generation of cloud-free or cloud reduced composite images from a collection of input images which may, individually, all be cloud affected. It also enables the use of reducers to transform an image collection into a single composite image, for example producing a median pixel value output from a series of individual input images acquired over a user-defined time period.

A series of Landsat image collections spanning four decades, from the 1980s to the 2010s, are used to assess the invasion process of Pinaceae and *Eucalyptus* spp. The image collections for the 1980s, 1990s and 2000s were captured by the Landsat 5 Thematic Mapper (TM) sensor, and the 2010s collection was captured by the Landsat 8 Operational Land Imager (OLI) and Thermal Infrared Sensor (TIRS) sensors, with sensor specifications presented in Table 1.

Within each decade, the years 1986/87, 1998/99, 2006/07 and 2016/17 were selected, with imagery collections for each generated for both summer and winter seasons (to characterise seasonal differences in landscape characteristics) for each period. The specific years targeted depended on the availability of cloud-free imagery. Different plant species exhibit varied phenological profiles throughout the year. Consequently, two vegetation types that are spectrally similar at one time of the year can be spectrally dissimilar at other times. Using multi-seasonal imagery exploits this difference, enabling improved discrimination of land cover and vegetation types. The date range for each decade was selected to be approximately every ten years (2016–2017, 2006–2007, 1996–1997, 1986–1987). For each winter and summer season respectively, imagery collections were selected over four month periods (Table 2). The earlier collections (1980 s, 1990 s) used images acquired over a wider time period due to the lower number of image acquisitions during these periods (Table 2). In addition, limited cloud free image availability for 1996 and 1997 necessitated the selection of 1998/99 as the period to represent the 1990 s decade. All images used were atmospherically corrected tier 1 surface reflectance data, which reach the highest radiometric and geometric quality standard, enabling time series comparison between different Landsat sensors (USGS, 2019). Table 2 shows a summary of the image collections used in this study. Image processing was performed using GEE (Gorelick et al., 2017).

Shuttle Radar Topography Mission (SRTM) Version 3 30 m resolution digital elevation model (DEM) data (Farr et al., 2007) was used as an additional ancillary data source in the classifications, as the large altitudinal gradient across the study area is expected to impact plant species distribution.

Multiple data sources were used for training and validating the land cover classifications. Field data were collected during two land cover survey campaigns in June-July 2017 and January-February 2018, with a total of 146 points (locations) visited. At each point, the following information was collected: land cover, qualitative description of the site as observed from the point; list of tree and shrub species (native and alien) present in a 5 m radius circular plot from the point; cardinal direction photographs (facing north, east, south and west); and, for some locations, supplementary upwards photos showing forest canopy cover/ stand density. Additional photos of features of interest in and around the survey points were also taken when appropriate.

To complement the field data, further reference information was obtained from: (1) geolocated field photos; (2) a pre-existing high quality land cover map of Chile created in 2014 at 30 m spatial resolution and with an overall accuracy of 80% (Zhao et al., 2016); (3) the Land Use and Vegetation Resources Cadastre created by the CONAF for management and national statistics purposes (CONAF, 2017) with 0.5 ha minimum mapping unit dating from 2013, 2008, 2007 and 1997; and (4) very high resolution (VHR) WorldView-2 satellite imagery, plus other VHR imagery accessible through Google Earth (Heilmayr et al.,

Table 1
Spectral bands of Landsat 5 and 8 used in this study (NASA, 2020).

Band	Landsat 5 (TM)		Landsat 8 (OLI + TIRS)	
	Wavelength (μm)	Spatial resolution (m)	Wavelength (μm)	Spatial resolution (m)
Coastal	-		0.44–0.45	30
Blue	0.45–0.52	30	0.45–0.51	30
Green	0.52–0.60	30	0.53–0.59	30
Red	0.63–0.69	30	0.64–0.67	30
NIR	0.76–0.90	30	0.85–0.88	30
SWIR 1	1.55–1.75	30	1.57–165	30
SWIR 2	2.08–2.35	30	2.11–2.29	60
TIRS 1	10.40–12.50	120	10.60–11.19	100
TIRS 2	-		11.50–12.51	100

Table 2
Landsat data collections used in this study.

Landsat sensor	Decade	Season	Date range	Number of images
Landsat 5 (TM)	1980 s	Winter	01/04/1986 – 30/09/1986	20
		Summer	01/11/1986 – 28/02/1987	23
	1990 s	Winter	01/04/1998 – 31/08/1998	12
		Summer	01/10/1998 – 15/03/1999	25
	2000 s	Winter	01/05/2006 – 31/08/2006	23
		Summer	01/11/2006 – 28/02/2007	25
Landsat 8 (OLI + TIRS)	2010 s	Winter	01/05/2016 – 31/08/2016	27
		Summer	01/11/2016 – 28/02/2017	29

2016). Using higher resolution imagery as a source of reference data for classification of coarser resolution imagery is an established technique (Duro, Franklin and Dubé, 2012). However, to ensure robustness, these VHR imagery were only used where there was no significant temporal change in landscape condition between the dates of the VHR imagery acquisition (information about the dates of acquisition is available in portals such as Google Earth) and the Landsat imagery analysed.

The CONAF's Land Use and Vegetation Resources Cadastre is created with a method based on the objective representation of vegetation, especially forests, through the identification of its life form (tree, vegetation or grass cover), structural properties, density, height and dominant species. This vegetation cadastre was first conducted in 1997 and should be, according to Law 20.283, article 4, publicly available and regularly updated. For the Araucanía region, where the study area is located, the cadastre was updated in 2007–2008 and 2014. For each update, a suite of cartographic data from governmental sources and remote sensing data is collated, analysed and compared with the previous version of the cadastre. Then, field campaigns to refine and validate the cadastre are carried out (CONAF, 2021).

The 2010 s decade classification was trained and validated using the field data, 2017 WorldView-3 imagery, the 2014 Chile land cover map and the most recent CONAF forest survey, dating from 2014. Reference data for the earlier classifications was extracted from historic VHR imagery via Google Earth (2003 onwards) and previous versions of the CONAF forest survey (2007–2008 and 1997). These ancillary datasets were scarce for the 1990 s decade and not available for the 1980 s. In this case, present-day field observation of mature forest (i.e. a mature undisturbed native forest present in the 2010 s must have been present in earlier decades) combined with expert visual assessment of Landsat imagery were used for training and validation purposes. This lack of reference data is a common issue in temporal studies such as this one. In these cases, visual interpretation is often used (Gómez, White and Wulder, 2016, Camacho Olmedo et al., 2022)

2.4. Image pre-processing

To minimize the topographical shadows in the satellite imagery that can negatively impact the classification quality, each image within each image collection was topographically corrected in GEE. The correction was based on the sun-canopy-sensor geometry, which is especially useful in forested areas (Soenen, Peddle and Coburn, 2005), and the illumination correction algorithm developed by Tan et al. (2013), which has been shown to improve the accuracy of forest change detection studies. Each image was then cloud masked using the Landsat pixel quality assessment band (pixel_qa). A median value composite image was then generated for each of the summer and winter image collections for each decade, with any remaining cloud masked manually.

Finally, a two-season composite image was created for each decade by stacking all bands from both winter and summer median composite images and the SRTM DEM. The final Landsat 5 composites (1980 s, 1990 s and 2000 s) comprised 15 bands (elevation and 14 spectral bands: 7 winter and 7 summer). The Landsat 8 composite (2010 s) had 19 bands (elevation and 18 spectral bands: 9 winter and 9 summer). For Landsat 8 the cirrus and panchromatic bands were removed from further assessment.

2.5. Land cover classification

Land cover classification was performed on the 1980 s, 1990 s, 2000 s and 2010 s composite images using a random forest classifier in GEE. Random forests are a machine learning algorithm (Breiman, 2001) widely used for land cover classification of multispectral imagery yielding high accuracies (Rodriguez-Galiano et al., 2012). The training dataset was constructed using the reference data summarized in Section 2.3. On average, 40 training polygons were created for each class, except for the more spatially constrained classes such as permanent snow, bare soil, urban or *Eucalyptus* spp., which had around 20 training polygons each.

Table 3 displays the ten-class classification nomenclature used for the classifications. This includes six vegetation classes and separates the different alien and native forest types. This system was built on the Food and Agriculture Organization (FAO) (Di Gregorio and Jansen, 2005), and 2014 Chile land cover map (Zhao et al., 2016) land cover classification systems, with adaptations for this study. It also includes a simplified classification system used subsequently in the temporal study of land cover change.

2.6. Land cover change analysis

Land cover change analysis initially calculated the area of each land cover in each subset of the study area (low-elevation valley, high-elevation Andes, and protected areas) to assess the broader variation in class coverage between different time periods. Secondly, land cover class change trajectories were investigated to assess overall change between the first and the last period (1980–2010 s), and each decadal-scale change (1980–1990 s, 1990–2000 s and 2000–2010 s).

Examination of the change trajectories involved construction of change detection matrices for each pair of classifications. As protected area extents have expanded over the time period covered by this study, the land cover area inside protected areas was calculated using the protected area extent in the earlier of the pair of classifications being compared (1980 s protected area extent for the 1980–1990 s comparison, 1990 s for 1990–2000 s, etc.). Finally, change maps were created for the vegetated land cover classes showing specifically where alien and native forest gain and loss has occurred. A simplified classification system was used for the change detection matrix and change maps to limit

Table 3
Simplified and detailed land cover class nomenclatures and class descriptions.

Simplified classes	Detailed classes	Class description
Alien forests	Coniferous plantations	Pinaceae family.
	Broadleaved plantations	*Eucalyptus* genus (*E. globulus* and *E. nitens*).
Native forests	*Nothofagus* spp. (broadleaved) forests	*Nothofagus* spp. or laurel forests (Laureliopsis philippiana, Aextoxicon punctatum, Eucryphia cordifolia, Caldcluvia paniculata, Weinmannia trichosperma, etc.).
	Araucaria araucana (mixed coniferous-broadleaved) forest	Patagonian forests (*Nothofagus* spp. mixed with *A. araucana* and high mountain shrubs).
Shrubs		*Chusquea culeou* mixed with *Holcus lanatus*, *Rosa moschata*, *Rubus ulmifolius* or other, less common shrubs. Also, large shrubs (or very small trees) such as *Aristotelia chilensis*, *Ovidia pillopillo.* Sometimes including smaller size, stunted *Nothofagus* spp. individuals.
Grassland		Agricultural grasslands or livestock grazing plots dominated by *Holcus lanatus*, *Poa nemoralis*, *Nothofagus obliqua* and *Luma apiculata*. Also, high areas dominated by mountain flora.
Water		Permanent water bodies including lakes and rivers.
Bare		Rocky outcrops, bare soils in rotation agricultural grassland, sandy lake beaches or high mountain areas above the limit of vegetation.
Snow		Permanent and seasonal snow.
Urban		Cities, smaller urban settlements, or impervious surfaces such as asphalt, concrete and roof materials.

error that could arise through confusion between forest sub-classes (Table 3). The simplified vegetated land cover classes used are alien forest plantations (including both coniferous and broadleaved), native woody species (encompassing *A. araucana* and *Nothofagus* spp. native forests and shrublands) and grasslands.

3. Results

3.1. Land cover classifications

Fig. 2 displays the land cover classifications for the four time periods. The classifications show that the area coverages of the land cover classes for the 1980 s were 2433.84 km^2 native *A, araucana*, 3266.84 km^2 native *Nothofagus* spp., 1430.77 km^2 shrubs, 786.99 km^2 coniferous alien plantations, and 981.29 km^2 alien broadleaved alien plantations. By the 2010 s, area coverage for *A. araucana* had decreased to 499.89 km^2, *Nothofagus* spp. had increased to 5426.29 km^2, shrubs had increased to 3426.52 km^2; and both alien coniferous and broadleaved alien plantations had decreased to 375.60 km^2 and 876.37 km^2, respectively. Table 4 summarises the class areas over the whole study period. Clear patterns in land cover distributions are observed in all periods: most plantations are located at lower elevation, in the valley, to

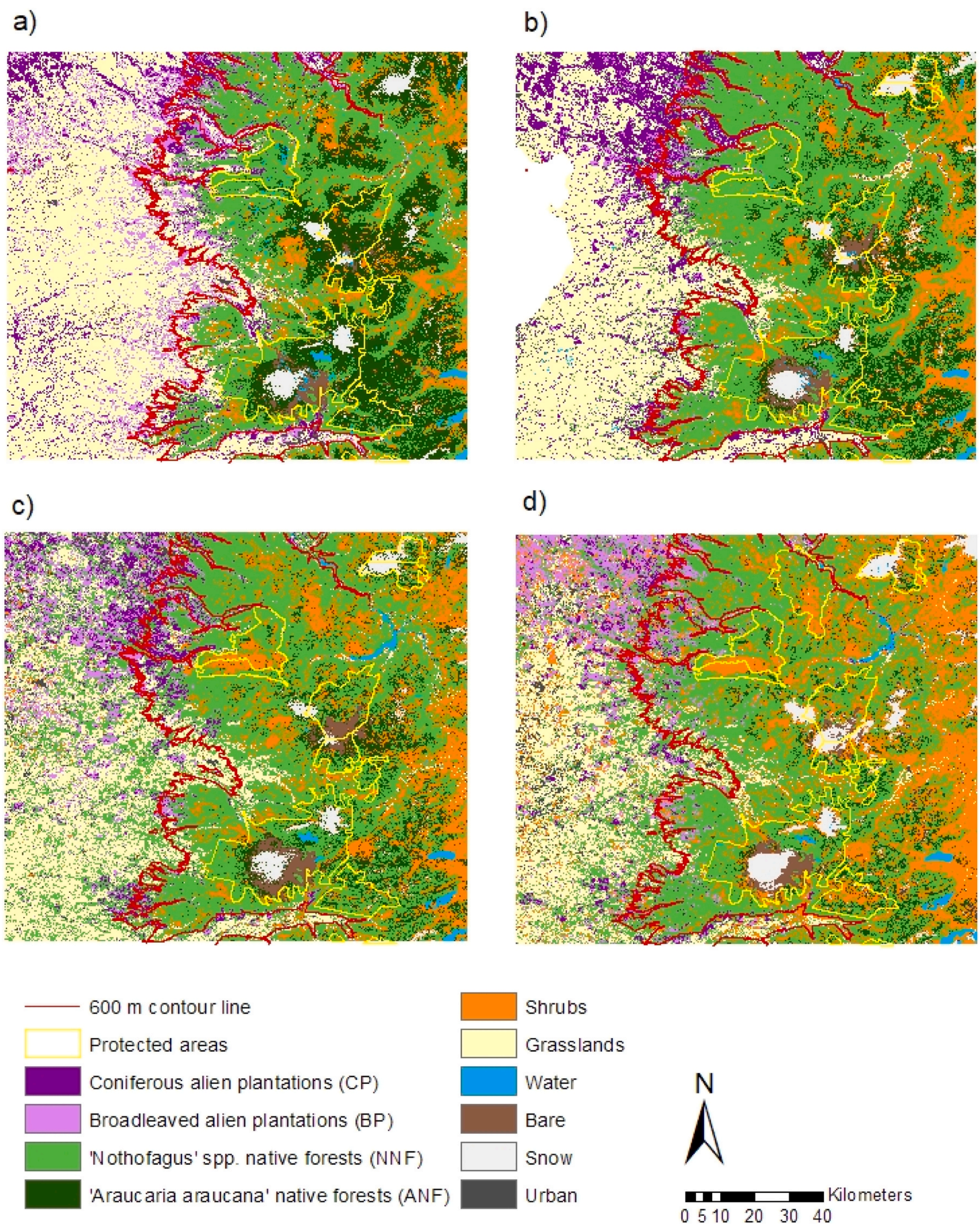

Fig. 2. Land cover classifications for: a) 1980 s, b) 1990 s, c) 2000 s, and d) 2010 s decades.

Table 4
Vegetated class areas in km^2 and percentages of the whole study area in the four decades.

Land cover class	1980 s	1990 s	2000 s	2010 s
Coniferous alien plantations	786.99 (5.74%)	1186.86 (8.65%)	574 (4.18%)	375.6 (2.74%)
Broadleaved alien plantations	981.29 (7.15%)	197.71 (1.44%)	713.6 (5.2%)	876.37 (6.39%)
Nothofagus spp. native forests	3266.84 (23.81%)	4627.85 (33.72%)	5716.51 (41.66%)	5426.29 (39.54%)
A. Araucana native forests	2433.84 (17.74%)	1509.64 (11%)	1064.18 (7.76%)	499.89 (3.64%)
Shrubs	1430.77 (10.43%)	1735.71 (12.65%)	2355.78 (17.17%)	3426.52 (24.97%)
Grassland	4139.66 (30.17%)	3654.32 (26.63%)	2469.35 (17.99%)	2067.39 (15.07%)

the west of the study area, while the Andes (to the east and containing the protected areas) and are dominated by native vegetation classes. This is unsurprising, as most urban settlements, agriculture and silviculture are located in the valley area, which is more easily accessible due to lower elevations and gentler topography.

The overall classification accuracies are 86%, 80%, 85% and 85% for the 1980 s, 1990 s, 200 s and 2010 s classifications respectively, with full accuracy assessment matrices presented in supplementary information (Tables S1 to S4). For the simplified class structure (Table 3), classification accuracies of 91%, 92%, 94% and 94% for the 1980 s, 1990 s, 200 s and 2010 s classifications respectively are achieved. A simplified classification system (Table 3) was used to overcome these confusions (Tables S5 to S8).

Generally, alien plantations are well separated from native forested areas. There is some confusion between the two alien plantation classes (Tables S1 to S4). The lowest accuracies in alien plantation classes are found for broadleaved alien plantations in the 1980 s and 1990 s decades (Tables S1 and S2). Also, shrubs, *Nothofagus* spp. and *A. araucana* forests sometimes experience confusion. This is unsurprising, as *A. araucana* forests are rarely monospecific, often being intermixed with *Nothofagus* spp. trees and, at high elevations, appearing as open forest with an understorey of shrubs. Additionally, the shrubs class includes some large shrubs belonging to the *Nothofagus* genus. These similarities among the three native woody species classes make them difficult to

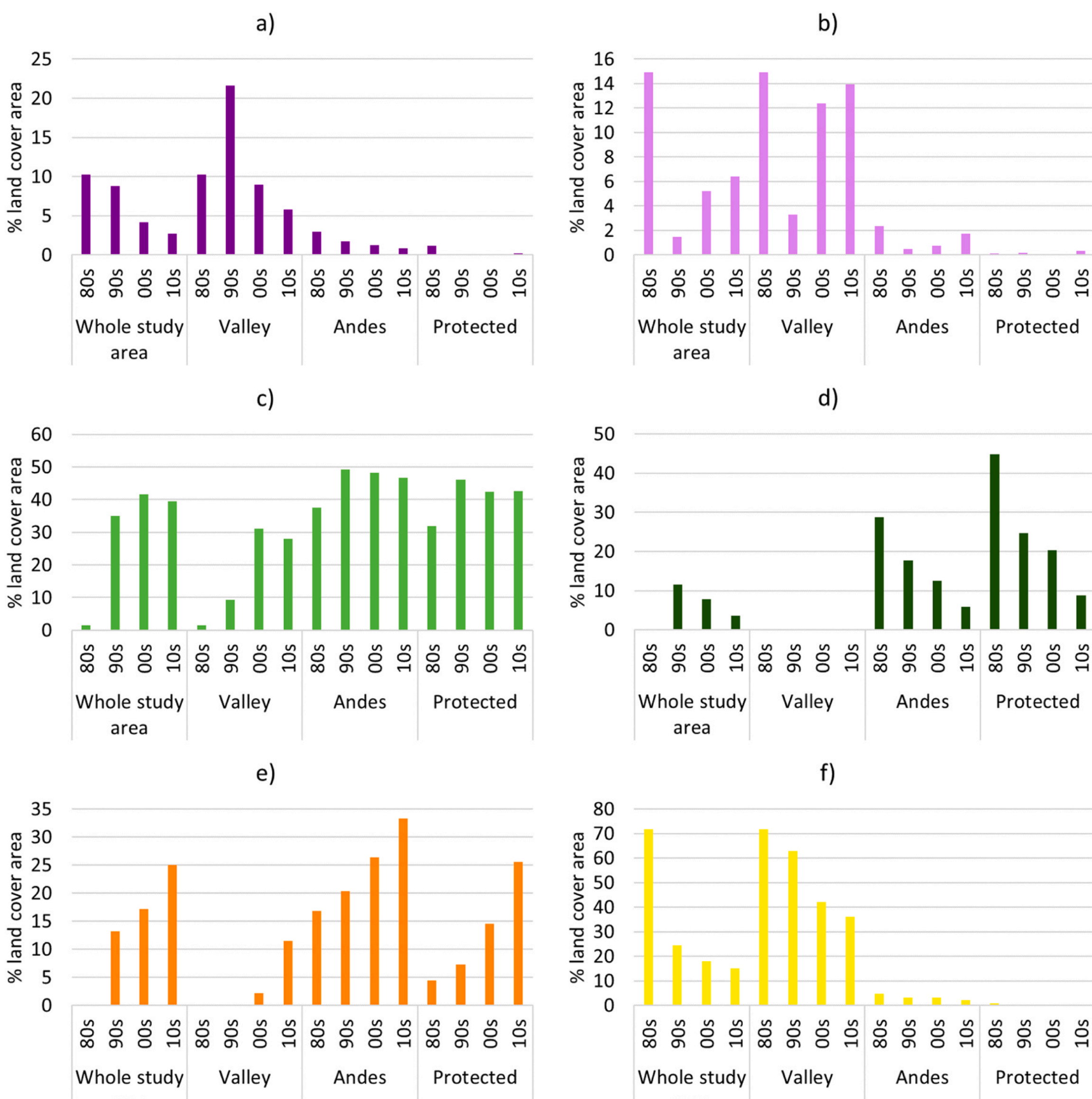

Fig. 3. Overall vegetation land cover change over the study period for a) Pinaceae, b) *Eucalyptus* spp., c) *Nothofagus* spp., d) *Araucaria araucana*, e) shrubs, and f) grasslands.

separate using 30 m resolution imagery. For this reason, the change detection analysis was performed using the simplified classification system mentioned earlier (Table 3), which achieved higher user's and producer's accuracies (Tables S5 to S8).

Fig. 3 presents the percentages of vegetated land cover areas for both the full study area and subset areas, for each decadal period across the full study period. In general, coniferous alien plantations have been steadily decreasing since the 1980 s (Fig. 3a). Broadleaved alien plantations, however, experienced a sharp decrease in the first period (1980–1990 s) followed by an increasing trend until the 2010 s (Fig. 3b). In 1976 the Chilean Forestry development law, which partially subsidized new pine and eucalyptus plantations, was issued, promoting the establishment of new alien plantations (Chilean Ministry of Agriculture, 1998). *Eucalyptus* spp. have a silvicultural rotation of 10–20

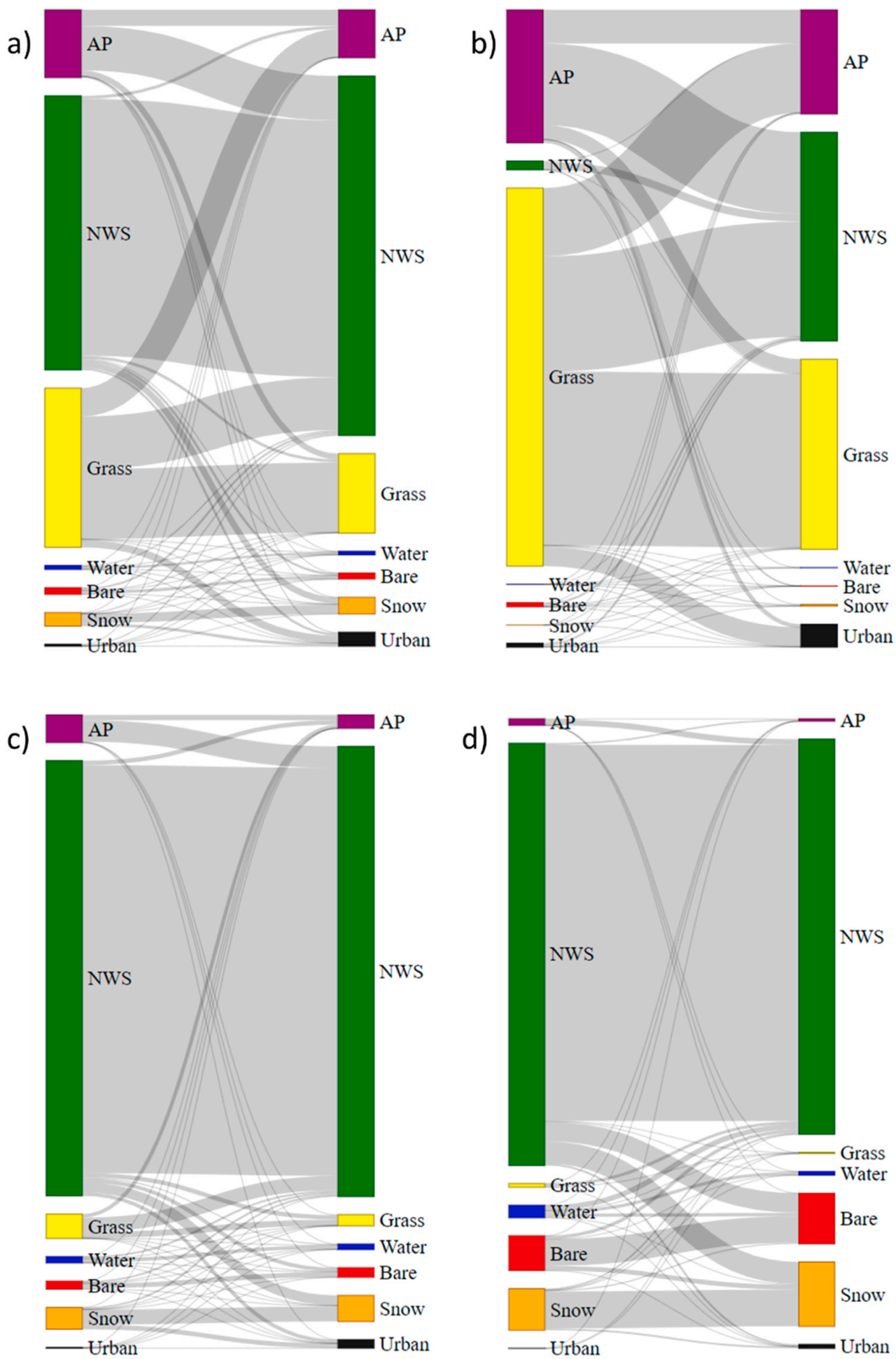

Fig. 4. Sankey plots demonstrating land cover change vectors between the 1980 s (left) and 2010 s (right) land cover classifications for (a) the full study area; (b) the valley sub-area; (c) the Andes sub-area, and; (d) the protected areas. The following abbreviations are used: AP = alien plantations, NWS = native woody species. The calculations for the protected areas were made using the protected area extents from the 1980 s decade.

years, often being harvested at 15 years (Riesco Muñoz, 2007, Salas et al., 2016) and Pinaceae rotations range generally between 18 and 35 years (Cantero, Espinel and Sáenz, 1995, Salas et al., 2016). These species are fast growing, and most of the plantations observed in the 1980 s image could have been established during the 1970 s, soon after the issue of the Forestry development law. Rotation periods then explain the decreases in *Eucalyptus* spp. between the 1980 s and the 1990 s and the sharpest decrease in Pinaceae land cover, which occurred between the 1990 s and the 2000 s

Native *Nothofagus* spp. forest is one of the most dominant land cover classes in all sub areas of the study area (Fig. 3c), however it shows different dynamics at higher elevations (Andes and protected areas) and lower elevations (valley area). The Andes and protected areas show relatively stable levels of *Nothofagus* spp. forest, however in the valley, *Nothofagus* spp. were a minority class in the 1980 s and 1990 s increasing to the second most dominant class in the 2000 s and 2010 s, only surpassed by grasslands, which consistently declined throughout the study period. The valley *Nothofagus* spp. forest stands are likely a result of forestry activities producing wood for local communities.

The endemic *A. arucana* forests have decreased in extent in all time periods and sub-areas (Fig. 3d), despite having been declared a Natural Monument in 1976 with laws forbidding its felling in almost all cases (Chilean Ministry of Agriculture, 1990). Accompanying the decrease in *A. araucana* forests has been an increase in shrublands, although it should be noted that high elevation *A. araucana* forests are generally open with an understorey of shrubs, potentially resulting in confusion between these two classes.

3.2. Land cover change analysis

Overall inter-class change between land cover classes for the full study area and valley, Andes and protected area sub-areas between the 1980 s and 2010 s periods are presented in Fig. 4, with equivalent figures for change between individual decadal periods (Figures S1 to S3) and full change matrices (Tables S9 to S12) available in supplementary information.

Between the 1980 s and the 2010 s (Fig. 4, Table S9), most of the alien forest cover shifted to native woody species (1479.22 km^2, 8.44% of the whole study area). This shift could be related to the end of forest subsidies, which may have encouraged other land uses, and therefore other land covers in the area. The rotation of native *Nothofagus* spp. plantations for timber ranges between 25 and 50 years depending on the site (Donoso and Soto, 2010). However, *Nothofagus* spp. is often used for sustainable forestry (FAO, 2016) or firewood and other local uses (Altamirano and Lara, 2010, Roco et al., 2023). In these cases, selective thinning, rather than clearcuttings, is the method used to extract wood, having a shorter silvicultural rotation and thus becoming a more attractive choice for smaller plots. Firewood production does not require trees as large as timber production, also shortening the silvicultural rotation periods. The fact that 1754.15 km^2 grassland (10.01% of the whole study area) was converted to native forest supports the hypothesis that native species forestry is an increasingly attractive enterprise in the area. Also, 938.83 km^2 grassland (5.36% of the whole study area) shifted to alien plantations, indicating that industrial forestry is still a relevant economic activity in Malalcahuello. Less than 5% of the study area experienced net native forest loss between the 1980 s and the 2010 s. An area of 8534.07 km^2 (48.68% of the whole study area) was native forest in the 1980 s and 2010 s. However, the land cover maps do not differentiate between undisturbed native forest and early successional or plantation native forest, so this apparent permanence could be masking other land cover changes in the intermediate decades. This shift of alien to native forest could also be an effect created by some confusion between alien plantations and *Nothofagus* spp. plantations, mainly in the 1980 s classification, which is subject to more uncertainty due to its earlier date.

Examination of alien plantation cover change between individual decades showed that between the 1980 s and 1990 s (Figure S1, Table S10) alien plantations remained (888.40 km^2, 5.28%) or were converted to native forest (686.41 km^2, 4.08%). This conversion from alien plantation to native woody species was the dominant trajectory in the Andes and within the protected areas. In the second period (1990–2000 s, Figure S2, Table S11) a large area of alien plantations remained (836.15 km^2, 4.97% of the whole study area), although an extensive area (709.70 km^2, 4.22%) was also converted to native forests. A large proportion of grasslands (1018.05 km^2, 6.05%) were converted to native forests, especially in the valley subset. This indicates that short rotation native forestry is an increasingly important activity, as the valley is dominated by anthropogenic land covers such as farming and forestry. As in the 1980–1990 s period, in the Andes and protected areas, alien plantations mainly changed to native forest cover.

The last period (2000–2010 s, Figure S3, Table S12) showed that a large portion of alien plantations remained (871.23 km^2, 4.97% of the whole study area). Part of them were converted to native forests, although this conversion was not as marked as in earlier decades. A larger proportion of native forests than in earlier periods were converted to grasslands, especially in the valley, where 6.62% of the area (442.54 km^2) experienced this change. This indicates that many native plantations have been clear-cut in recent decades, again suggesting native species forestry activities. Also, the 2000–2010 s decade showed more conversion from native forests to alien plantations in all areas than in earlier decades. This was expected in the valley area as a result of conversions between different forest plantations, but is a concern for the Andes and protected areas, as it could mean that either new alien plantations are being established in the more pristine part of the study area, or that invasion is occurring.

The spatial distribution of the land cover change patterns observed between the 1980 s and 2010 s is displayed in Fig. 5. Although Fig. 5 shows the different protected area limits for each of the decades, the numerical calculations were made using only the protected area extent from the earliest decade of the pair being compared to allow consistent comparisons. The altitudinal zones are clearly distinct in this figure, with most of the land cover dynamism concentrated in the valley area, to the west of the study area (Fig. 5, panels a and b). As mentioned earlier, most conversion to alien forest took place in the valley, especially in earlier decades, when the largest alien forest patches were established (Fig. 5, panels c, e and g). The valley also shows native forest establishment throughout the decades (Fig. 5, panels d, f and h), but it is restricted to smaller plots, perhaps a result of subsistence forest planting for firewood or natural regeneration. Finally, as stated earlier, the maps also show that native forest loss in the Andes is slowly, but steadily occurring, even within protected areas (Fig. 5, panels d, f and h).

4. Discussion

This study investigated the dynamics of alien tree spread and native forest loss over 31 years in the study area of the Malalcahuello National Reserve and surrounds, a region of the Chilean Andes and Andean foothills. This area of Chile is still relatively pristine, experiencing only the first stages of deforestation, whereas other areas of Chile (the Coastal Range) are already severely impacted by deforestation, fragmentation and species introduction for industrial forestry purposes (Echeverría et al., 2006, Miranda et al., 2017).

Assessment of the distribution of native and alien plantation forested areas across four time intervals from the 1980–2010 s has demonstrated distinct land cover patterns for the subset areas based on their elevation, as previously found in this environment (Hora et al., 2022, Martin-Gallego et al., 2020). Most human activity and related land covers are located in the lower elevation valley sub-area, which is dominated by grasslands and alien forest classes, while the higher elevation Andes sub-area is predominantly covered by native woody species, with some alien Pinaceae presence. Although this general landscape structure was noticeable throughout the full study period,

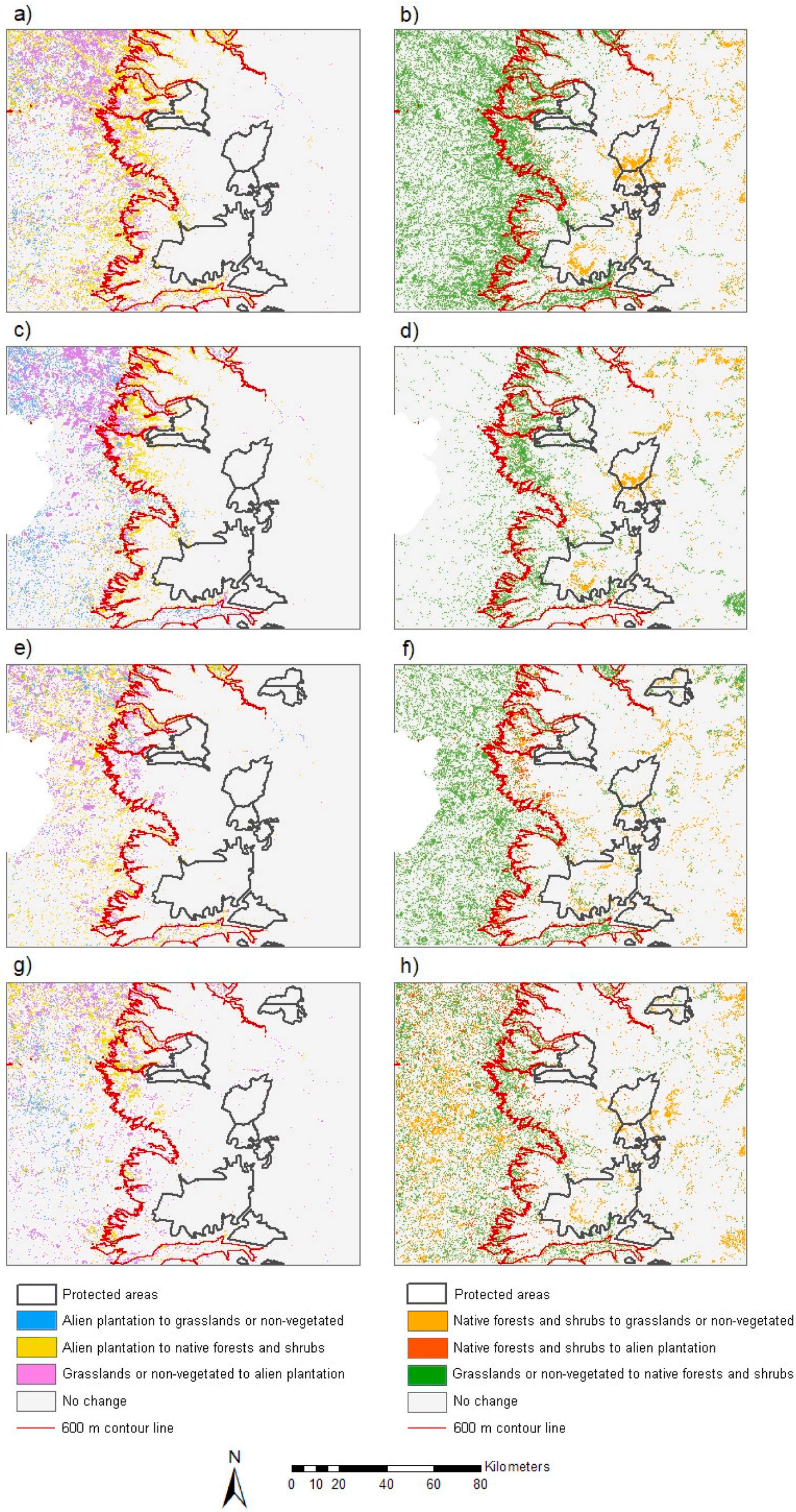

Fig. 5. Forest change maps for dynamics of a) alien plantation loss and expansion 1980–2010 s, b) native forest loss and expansion 1980–2010 s, c) alien plantation loss and expansion 1980–1990 s, d) native forest loss and expansion 1980–1990 s, e) alien plantation loss and expansion 1990–2000 s, f) native forest loss and expansion 1990–2000 s, g) alien plantation loss and expansion 2000–2010 s, and h) native forest loss and expansion 2000–2010 s. The white area to the west of panes c-f corresponds to an area of no data due to persistent cloud cover.

results indicate that alien Pinaceae and *Eucalpytus* spp. and native *Nothofagus* spp. forest classes are highly dynamic, converting from one to another throughout the study period. This dynamism is a result of anthropogenic intervention and, as such, is mainly observed in the valley. The three groups of species (alien Pinaceae and *Eucalpytus* spp. and native *Nothofagus* spp.) are commonly planted in the area. Pinaceae is used for the wood industry, *Eucalyptus* spp. is mainly for pulp and the native *Nothofagus* spp. is sometimes planted for sustainable forestry or for wood and firewood for local communities. Rotation periods range between 18 and 35 years for Pinaceae, 10–20 for *Eucalyptus* spp., and 25–50 for *Nothofagus* spp. depending on the site (Cantero, Espinel and Sáenz, 1995, Riesco Muñoz, 2007, Donoso and Soto, 2010, Salas et al., 2016). Between the first period (1980 s) and the last (2010 s), roughly 15% of the valley subset area changed from alien forest classes (coniferous Pinaceae and broadleaved *Eucalyptus* spp.) to native forest. Since *A. araucana* and native shrubs are virtually absent from the valley, this change was mainly to *Nothofagus* spp.

Government subsidies for productive forestry began in the 1970 s and ended in 2012. This has likely driven smallholders to move from planting Pinaceae to alternative species with shorter rotation periods such as *Eucalyptus* spp., and from industrial forestry (Pinaceae and *Eucalyptus* spp.) to sustainable or short rotation forestry destined for more local markets (*Nothofagus* spp.). For small farm owners, native forest such as *Nothofagus* spp. acts as an emergency resource for low-income years, being used for firewood extraction (Roco et al., 2023). This is supported by the observed changes in 2000–2010 s, where less than 5% of native forest converted to alien forest cover in the valley area. However, it is important to remark that native deforestation in favour of productive forestry is still occurring in the area. Although the rate of conversion in 2000–2010 s is lower than observed in the 1980–2010 s, it had increased compared to the 1990–2000 s. This is especially concerning in the Andes, where change from native to alien species has also increased in the last period.

The consistent grassland decrease and forest increase in the valley sub-area indicates that anthropogenic activities here had previously focused on forestry, with agriculture being displaced to other areas. However, with the end of forest subsidies, smallholders whose income depends on small land areas may need to transition from long rotation forestry to other activities more consistently profitable in the short-term. These could include short rotation forestry (such as *Eucalyptus* spp. or *Nothofagus* spp.) or other, alternative activities such as sustainable forestry or tourism, with the less profitable estates (due to size or inaccessibility) naturally shifting from industrial monoculture forestry to other uses. This, however, may not affect extensive plantations owned by larger, highly profitable forestry companies, which could potentially keep deforesting native forest areas (there is a rise in native to alien conversion within the Andes in the last period). In the absence of subsidies, the landscape could diversify as smallholders change their activities, and Pinaceae productive plantations may generally become larger.

Assessment of the spread or invasion process of Pinaceae and *Eucalyptus* spp. and deforestation of the native forests showed that, overall, between the 1980 s and the 2010 s, there has not been a net spread of alien forest cover at the expense of native forest in the area. However, conversion from native to alien forest has been accelerating throughout the time period of this study, suggesting that forest is a highly dynamic land cover class as a result of intense silvicultural activity, especially in the valley area. Comparing the first decade (1980 s) with the last (2010 s) conceals this dynamism as this method does not differentiate between undisturbed native forests and native plantations; loss of undisturbed native forest could be concealed by native plantations or natural regeneration established later. Similar results have been reported previously in other parts of Chile by Altamirano et al. (2013). However repeat monitoring on a decadal scale, as performed here, can identify these shorter-term dynamics. For small farm owners, native forest such as *Nothofagus* spp. acts as an emergency resource for low-income years, being used for firewood extraction resulting in forest degradation (Roco et al., 2023). Separating different types of native forest (i.e. undisturbed, disturbed, planted, natural regeneration) could also provide a more accurate measure of forest loss, and should be a focus of future work.

When the processes of alien spread and forest loss are compared, no clear link between them is found. At Landsat's resolution, roughly 110 km^2 of the alien forest stands that were present in the Andes in the 1980 s still remain today. These are likely to be abandoned plantations, as some test plantations were introduced in the early 1970 s (Peña et al., 2008). These trees represent sources of propagules that may be triggering an invasion process. In fact, natural *Pinus contorta* (Pinaceae) regeneration was found at 1200 m from the parent stand inside the Malalcahuello National Reserve (Peña et al., 2008). The earliest stages of invasion, where only scattered young trees are present, are not detected using Landsat due to the 30 m pixel size. These abandoned plantations should be closely monitored in the close future, as biological invasions can only be controlled at the early stages (Pauchard et al., 2016).

A concerning observation was the rise in shrub cover in the Andes and protected sub-areas, although the shrubs class is sometimes confused with *Nothofagus* spp., especially with young early successional forests as these include a mix of native shrubs and arborescent shrubs, some of which belong to the *Nothofagus* genus. *A. araucana* forests are generally open and with an understory of grasses and small shrubs and thus potentially confused with the shrubland class. Despite these potential sources of classification confusion, the widespread rise in shrub is still concerning. Alongside the decrease of *A. araucana* forest in all decades, this sign of forest degradation requires further investigation, as conversion from native forest to shrub has also been documented in nearby areas (Zamorano-Elgueta et al., 2015, Miranda et al., 2017). Shrub encroachment in the naturally open *A. araucana* forest could also prevent *A. araucana* regeneration, jeopardizing the conservation policies in place for this endemic, culturally relevant tree.

Climate change also has the potential to influence land cover change in the area, with increasing temperatures and decreasing rainfall occurring and projected to occur in the future (Kitzberger et al., 2016, González et al., 2018, Urrutia-Jalabert et al., 2018). This has negatively impacted productivity of rainfed crops and grasslands (Garreaud et al., 2017), and is corroborated by the results of this analysis, which show a steady decrease in grassland cover over time. The relative increase in *Eucalyptus* spp. compared to Pinaceae could also be a result of the warmer and drier conditions; *Eucalyptus* spp. are highly productive and profitable trees that have been traditionally limited by low temperatures in this environment (Cesaroli et al., 2016). Degradation of *A. araucana* forest could also be influenced by climate change, further reducing its already limited high mountain habitat. Additionally, increasing temperatures will allow alien trees to colonise higher elevations, resulting in direct competition with *A. araucana*. A warmer and drier climate will increase forest fire frequency (González, Veblen and Sibold, 2005, González et al., 2018), and a mixed stand of *A. araucana* and Pinaceae alters fuel structure increasing flammability and crown fire hazard (Cingolani et al., 2023). In this context, Pinaceae and *Eucalyptus* spp. could outcompete the native *Nothofagus* spp. and *A. araucana*¸ as they regenerate and resprout very efficiently after wildfires (Drake, Molina and Herrera, 2012, Kitzberger et al., 2016).

The high dynamism observed in the forested classes has direct ecological implications due to the increasing relative abundance of plantations and successional forests. Alien species forest plantations and early successional native forests have a lower potential to support ecosystem functions and services compared to native, undisturbed vegetated covers (Brockerhoff et al., 2013, Balthazar et al., 2015, Zeng et al., 2019). However, when compared to degraded or agricultural lands, their potential is higher (Chazdon, 2008, Balthazar et al., 2015). These constant forest land cover changes promote disturbance and fragmentation of the native forest ecosystem, decreasing biodiversity (Echeverría et al., 2007). Forest plantations (both alien and native) and early successional native forests tend to have a simple canopy structure and species composition. This lack of landscape heterogeneity and structural complexity negatively impacts biodiversity (Hartley, 2002, Díaz et al., 2005, Lindenmayer, Hobbs and Salt, 2013). Although not ideal for biodiversity conservation, forest plantations and early successional forest could act as low contrast land covers to enhance native forest connectivity and contribute to the preservation of biodiversity (Brockerhoff et al., 2008). Endemic fauna such as the arboreal marsupial *Dromiciops gliroides* are dependent on the native forest structural complexity, but could move through low contrast land covers (i.e. not treeless) to access other suitable habitat patches (Fernández et al., 2022). However, certain specialist species will still be restricted to the remnant undisturbed native forest patches (Barlow et al., 2007, Dent and Joseph Wright, 2009). Successional native forests gradually increase their species richness over time. However, for other forested systems, it has been reported that it is only after centuries that a successional forest restores the species richness of an undisturbed forest (Rozendaal et al., 2019). In addition, the increase in species richness in successional forests is higher when there are undisturbed native forest remnants nearby (Middendorp et al., 2016).

From a management perspective, a new approach to forestry is required to achieve sustainability and increase its social an economic benefit (Donoso and Romero, 2020). It is important to preserve native forest patches with a high structural complexity and species richness, as they support specific biodiversity and act as seed sources for regenerating native forests nearby. Promoting alternative forestry practices such as mixed species, mixed age plantations using native species, and avoiding clearcuttings would enhance ecosystem service provision in the area. The land cover mapping approach applied here could be used to enhance and update traditional forest surveys to identify high invasion risk areas and improve protected area management. Compared to the CONAF's vegetation cadastre, the method used in this study provides a less resource intensive option for targeting specific locations or spatially explicit ecological problems like invasion. Consequently, continuous, frequent monitoring becomes feasible. For management planning, accurate mapping of the distribution of alien species is essential. Whereas historically this has been performed using herbarium records, EO analysis offers a superior approach, avoiding common bias in ground-based methods such as those related with choosing collection locations (Bradley, 2014). Using EO imagery such as Landsat or Sentinel-2 is a cost-effective way of monitoring forests over large areas, with the Landsat imagery archive being especially useful for retrospective monitoring, and to assess the impacts of landscape scale management actions. The information provided using these methods can be a cornerstone for the design of successful new policies, land management and invasion control strategies that minimise the risk of invasion stemming from productive plantations in Chile and elsewhere. This way, the highly profitable Chilean forestry industry could become more sustainable.

Despite the advantages of remote sensing methods for alien species mapping, there are also limitations. The earliest stages of invasion, where only scattered young trees are present, are not detected using Landsat due to the 30 m pixel size, and the availability of reliable ground reference data is another limitation for remote sensing, especially for historical analysis. Future work could benefit from utilizing alternative higher resolution satellite sensors such as Sentinel-2 to provide more accurate separation between vegetated land covers due to its spectral capabilities (Martin-Gallego et al., 2020), or very-high resolution imagery from satellites such as Planet (O'Shea, 2020) or from drones (Barbizan Sühs, Ziller and Dechoum et al., 2023) to detect early invasion patches.

5. Conclusion

This study has demonstrated that continuous monitoring of forests is crucial for sustainable land planning, as both natural and introduced plantation forests can be highly dynamic. This high dynamism, supported by similar results from other areas of Chile (Altamirano et al., 2013), together with the long silvicultural rotation periods, consequently require long time series of data to model alien species spread and forest loss processes accurately. Continuous future monitoring of forest changes is also essential for a sustainable land planning strategy that allows forestry, while conserving ecologically valuable native forests. Earth observation methods, such as those presented here, enable this continuous monitoring of forest dynamics over extensive and inaccessible areas in a timely and cost-effective manner. This has particular value in countries such as Chile, which hosts a wide but threatened biodiversity, and suffers a lack of funding for conservation (Waldron et al., 2013) and monitoring activities.

There is a clear separation in land cover and change patterns by elevation in the landscape of the Malalcahuello study area, as supported by earlier studies (Hora et al., 2022, Martin-Gallego et al., 2020). The lower elevation valley area is highly disturbed and dynamic, hosting most productive forestry and anthropogenic land uses. The higher elevation Andes area remains relatively untouched, with native forests being the dominant land cover. Although alien forest stands are located mostly in the valley area, the presence of abandoned alien forest plantations embedded in the native forest in the Andes poses a threat for future invasions. In addition, shrub encroachment and *A. araucana* loss are occurring in the native forests, suggesting that there are other disturbances causing forest degradation. The reasons for this encroachment fall outside of the scope of this study, but they should be further investigated, as the conservation value of the area will diminish in the future if this pattern continues.

Finally, the results of this study suggest that socioeconomic factors have a great impact in the landscape composition. Particularly, socioeconomic variables have already been found to impact the forest area change in the Chilean Valdivian ecoregion (Roco et al., 2023). The end of the subsidies for productive forestry in 2012 marked an inflection point for the landscape in Malalcahuello, as smallholders have shifted from long rotation plantations to shorter rotation forestry or other land uses. Consequently, the land uses in the area are expected to diversify in future decades. Despite the end of forest subsidies, large-scale industrial forest plantations will persist, as they will still be profitable. If no strict boundaries are set for the location of new productive plantations, a process of land consolidation could start and eventually lead to deforestation of higher elevations over time, as the rise in native to alien conversion in the last period suggests. From a management perspective, it is key to manage forest at the landscape scale (Donoso and Romero, 2020), monitor alien plantation establishment and invasion patterns and control the invasive species appropriately to preserve undisturbed native forest remnants. It is also important to promote species and structural complexity to safeguard this highly ecologically valuable and fragile ecosystem.

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CRediT authorship contribution statement

Aníbal Pauchard: Writing – review & editing, Methodology, Conceptualization. **Paul Aplin:** Writing – review & editing, Supervision, Methodology, Conceptualization. **Pilar Martin-Gallego:** Writing – original draft, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Christopher Marston:** Writing – review & editing, Supervision, Methodology, Conceptualization. **Adison**

Altamirano: Writing – review & editing, Conceptualization.

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.

Data Availability

Data will be made available on request.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.foreco.2024.121847.

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