



Detecting and modelling alien tree presence using Sentinel-2 satellite imagery in Chile's temperate forests

Pilar Martín-Gallego^{a,*}, Paul Aplin^a, Christopher Marston^{a,b}, Adison Altamirano^{c,d}, Aníbal Pauchard^e

^a Department of Geography and Geology, Edge Hill University, St Helens Road, Ormskirk, Lancashire L394QP, United Kingdom

^b UK Centre for Ecology & Hydrology, Lancaster, United Kingdom

^c Landscape Ecology and Conservation Lab, Facultad de Ciencias Agropecuarias y Forestales, Universidad de La Frontera, Temuco, Chile

^d Butamallin Research Center for Global Change, Facultad de Ciencias Agropecuarias y Forestales, Universidad de La Frontera, Temuco, Chile

^e Laboratorio de Invasiones Biológicas, Facultad de Ciencias Forestales, Universidad de Concepción, Concepción, Chile



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ABSTRACT

Chile's temperate forest is a global biodiversity hotspot. An upsurge in alien forest plantations has disturbed and fragmented the landscape, promoting biological invasions. The invasion process is not fully understood since monitoring large and inaccessible areas can be prohibitively expensive and logistically challenging using field-based methods alone. Here, a remote sensing approach using Sentinel-2 satellite imagery, fragmentation analysis, and random forest modelling is applied to detect alien tree stands and describe their extent in relation to fragmentation and landscape structure in study areas around Malalcahuello National Reserve and Villarrica National Park. Detailed vegetation maps are produced, with classification accuracies > 81% and including four forest classes, two native and two alien. An altitudinal pattern was observed in both sites. At lower altitudes, there was greater total area covered by alien trees and more fragmented native forests than at higher altitudes. However, Villarrica had less alien tree cover than Malalcahuello, but was a more fragmented landscape. Random forest modelling identified that alien pine tree mean patch area was positively correlated with both land cover diversity and *Araucaria araucana* forest mean patch area in both sites. Given their conservation and cultural relevance, the locations of protected areas need reconsidering to strengthen the protection of *A. araucana*, which could be outcompeted by alien trees in a context of increasing productive forestry. This is especially urgent in Villarrica, where protected areas already have a substantial presence of alien trees, with most *A. araucana* found outside protected areas.

1. Introduction

Temperate forests are the least abundant of the world's woodland biomes; however they are affected by high rates of annual forest loss (Echeverría et al., 2006; Hansen et al., 2010), land use change (Ellis, 2011) and are increasingly threatened by the invasion of alien trees (Essl et al., 2011; Richardson and Rejmánek, 2011) and herbaceous plants (Wavrek et al., 2017). Over half of the temperate forests in the southern hemisphere occur in Chile (Donoso, cited in Miranda et al., 2015, p. 21). Chile itself is considered a global biodiversity hotspot (Myers et al., 2000). The annual rate of native forest loss in Chile was 0.3% between 1986 and 2011 (Heilmayr et al., 2016), and the Chilean biodiversity hotspot only retains 30% of its original native vegetation cover (Myers et al., 2000).

The Chilean Valdivian temperate forest ecoregion is located between the biogeographical barriers of the Andes and the Pacific Ocean (Altamirano and Lara, 2010), from 35° to 48° south latitude (Smith, 2002), at altitudes ranging between 1000 and 3000 m (Smith, 2002). This wide latitudinal range results in large variations in temperature and rainfall within the ecoregion (Smith, 2002). Having evolved in isolation, this region has very specific flora and it is sensitive to species introduction (Alpert et al., 2000). Particularly, the endemic tree *Araucaria araucana* (monkey puzzle) is classified as endangered in the IUCN Red List of Threatened Species due to its decreasing population (Premoli et al., 2013). Additionally, *A. araucana* is considered a Natural Monument by Chilean law, recognising its cultural value and forbidding felling unless in exceptional cases (Chilean Ministry of Agriculture, 1990).

* Corresponding author.

E-mail address: 23448776@edgehill.ac.uk (P. Martín-Gallego).

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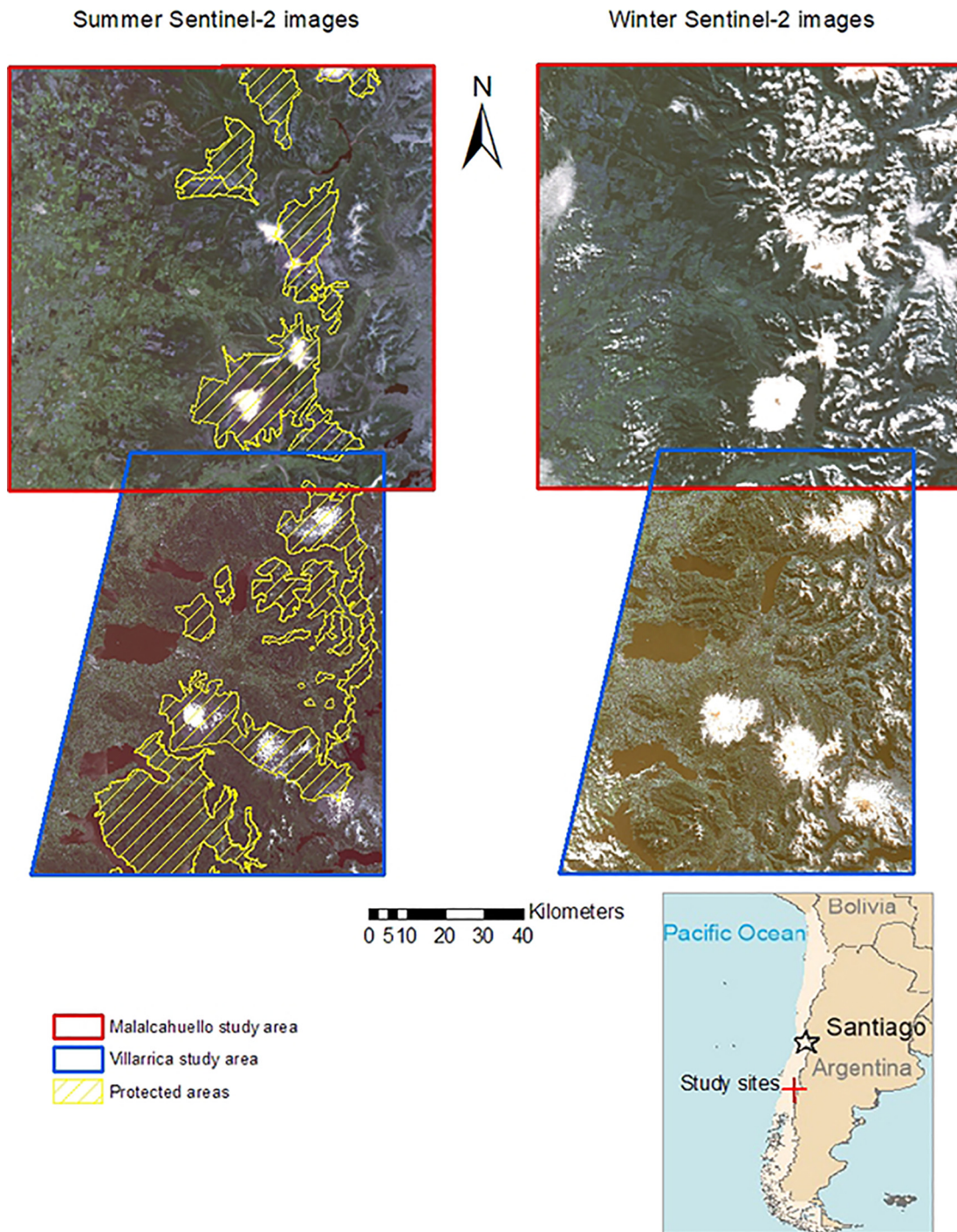


Fig. 1. Location of the Malalcahuello and Villarrica study sites and protected areas.

Significant land use change has occurred in the Valdivian temperate forests since the 1970 s, with large-scale expansion of alien plantations in recent years (Miranda et al., 2015). Deforestation and forest degradation, and subsequent fragmentation, have been ongoing processes (Echeverría et al., 2012; Miranda et al., 2015), with forest loss rates reaching 4.5% per year in some areas (Echeverría et al., 2006). This continuous human disturbance makes the landscape more vulnerable to biological invasions (Giorgis et al., 2016; León Cordero et al., 2016; Pauchard et al., 2016). As Chilean temperate forests are disturbed and considered vulnerable to invasion (Arroyo et al., cited in Pauchard et al., 2004: 255, Early et al., 2016), research on this process has been growing in recent years (Quiroz et al., 2009), after historically being an

understudied topic in South America (Speziale et al., 2012).

Forestry is the second main reason for introduction of woody plants, after horticulture (Richardson and Rejmánek, 2011). Among 443 forest plants that can become invasive worldwide, 292 are typical of commercial forest plantations, mostly belonging to the *Leguminosae*, *Pinaceae*, *Myrtaceae*, *Rosaceae*, and *Salicaceae* families (Haysom and Murphy, cited in Dodet and Collet, 2012: 1766). Several tree species planted commercially are considered invasive or potentially invasive in Chile (Bustamante and Castor, 1998; Echeverría et al., 2007; Teillier et al., 2003) and around the world (Dzikiti et al., 2016; Pauchard and Alaback, 2004; Peña et al., 2008). The spread of propagules from alien forest plantations to surrounding native forests has been documented

(Peña et al., 2008), with competition from alien saplings threatening the regeneration of endangered native species such as *A. araucana* (Pauchard et al., 2014). Prompt, targeted management is fundamental to controlling these tree invasions (Pauchard et al., 2016). The scale and topography of Chile's forested areas, however, pose challenges for ground-based monitoring, as it is costly and resource intensive.

Although alien tree spread and invasions are a cause of concern across biomes of the world, detecting alien trees in temperate forests is particularly difficult, especially when compared to other open treeless ecosystems. In many regions, similar large-scale afforestation poses an invasion risk, but lack of awareness and resources has made the assessment of the extent of such invasion processes very difficult. Under such a scenario, remote sensing is an obvious alternative, holding significant potential for large scale alien tree detection and mapping. It offers an affordable method of forest monitoring over large areas, with frequent temporal data acquisition at a variety of spatial and spectral resolutions including cost-free medium spatial resolution multispectral satellite imagery. This imagery can help producing quantitative information on alien tree spread at large scale in a simple and cost-effective way, being an invaluable tool for designing new conservation strategies for temperate forests not only in Chile, but also globally.

Specifically, the European Space Agency (ESA) Sentinel-2 satellites offer regular (every five days at the equator) medium resolution, spectrally rich data acquisition over broad areas (290 km swath width). Sentinel-2 has been used previously to detect alien shrubs in South Africa, Spain and Chile (Calleja et al., 2019; Ndlovu et al., 2018). Additionally, Sentinel-2 has been shown to have the potential to discriminate alien plants from native plants with a similar morphological structure (Tesfamichael et al., 2018).

The main aim of this study is to detect alien trees using satellite remote sensing and describe their degree of occupancy in relation to fragmentation indices and landscape parameters in Chile's temperate forest ecosystem. The specific objectives are: (1) identify and map these alien tree stands and patches within the native forest matrix; (2) describe the spatial characteristics of forested areas using structural landscape metrics; and (3) model the influence of landscape variables on the presence and abundance (both as a result of plantation and invasion) of alien trees from the *Pinaceae* family. Studying these species is expected to shed light into how to detect and monitor alien tree presence inexpensively in temperate forests of the world where resources are limited, and the scale of the alien tree presence and possible invasion has not yet been fully determined.

2. Material and methods

2.1. Study areas

Two Valdivian temperate forest study areas were selected in the south-central Chilean Andes within the Araucanía region (IX administrative region), offering a mosaic of similar land cover, and both in the Andes and Andean foothills. Study area 1 (Fig. 1) comprises Malalcahuello National Reserve and surrounds in the northern Araucanía region (-38° latitude), at altitudes from 500 to 2800 m. Study area 2 (Fig. 1) covers Villarrica National Park and surrounds in the southern Araucanía (-39° latitude) at altitudes from 200 to 2800 m. Both study areas are roughly 10000 km². The Villarrica study area (42 inhabitants/km²) is more densely populated than Malalcahuello (10 inhabitants/km²) (Library of National Congress Chile, 2015). Economic activities in Villarrica are predominantly touristic, while the principal economic sectors in Malalcahuello are farming and forestry (Library of National Congress Chile, 2015). Both study sites contain a mosaic of protected areas including National Reserves, National Parks, and UNESCO's (United Nations Educational, Scientific and Cultural Organization) Biosphere Reserves. The native forest comprises various deciduous *Nothofagus* spp. mixed with the evergreen *Nothofagus dombeyi*, *Araucaria araucana* and high mountain shrubs (Smith, 2002).

More than 10 alien coniferous species were introduced inside the Malalcahuello National Reserve during 1969 and 1970, posing a threat to the endangered *A. araucana* (Peña et al., 2008). Test plantations of conifers, *Eucalyptus* spp. and other alien species started in 1950 within the Villarrica study area (Kunstmann, 1965; von Buch, 1965).

Although Malalcahuello and Villarrica have differences in land uses and dates of introduction of the first alien tree species, they are geographically close without a hard boundary between them. Their characteristics change gradually, along a gradient. Consequently, the areas have a slight overlap (Fig. 1) that accounts for this gradient.

Both sites can be divided in two subsets based on altitude. The lower altitude subsets are in valley areas (< 600 m elevation), while the higher altitude (> 600 m) subset corresponds to the Andes and Andean foothills. The valleys are typically more influenced by human activity than the Andes and contain the major urbanised and agricultural areas.

2.2. Study species

Two categories of alien trees were selected for this study. These were coniferous trees from the *Pinaceae* family (*Pinus radiata*, *Pinus contorta*, *Pinus ponderosa*, *Pinus sylvestris* and *Pseudotsuga menziesii*), and broadleaved trees from the *Eucalyptus* genus (*Eucalyptus globulus* and *Eucalyptus nitens*). Both are widely used in commercial forest plantations in Chile. These species are highly productive, establish easily, have fast growth rates and most exhibit low shadow tolerance, characteristics common in many invasive species (Dodet and Collet, 2012).

Invasion is highly dependent on the ecology and growth form of the alien plants and their positive or negative interactions with the structure and the characteristics of the ecosystem (Giorgis et al., 2016). For this reason, focusing on functional groups rather than individual species is more appropriate for this study, as the species from each group (*Pinaceae* and *Eucalyptus* spp.) share ecological characteristics and growth form.

Pinaceae are heliophytes (adapted to grow in full sun) with very high recruitment rates, tending to create closed canopies where there were previously open areas. Most *Pinaceae* species used in plantations are invasive and listed by the IUCN (International Union for Conservation of Nature) in the Global Invasive Species Database (IUCN Invasive Species Specialist Group, 2006; Nuñez et al., 2017). For example, *Pinus contorta* has started colonizing treeless steppes in Patagonia (Langdon et al., 2010; Pauchard et al., 2016), and is reported as spreading to the naturally open *A. araucana* forests in Malalcahuello (Peña et al., 2008), potentially resulting in canopy closure and preventing *A. araucana* regeneration (Pauchard et al., 2014). In addition, *Pinus radiata* has been identified as an invasion threat for the native forest in Australia (Calviño-Cancela and van Etten, 2018). Of the *Pinaceae* group studied here, the only relatively shade tolerant species is *Pseudotsuga menziesii*, which can regenerate under the forest canopy, especially if it has been thinned (Pauchard et al., 2008; Peña and Langdon, 2007). Nonetheless, it has been included in the *Pinaceae* group class due to technical limitations. The species of the *Pinaceae* family, including *Pseudotsuga menziesii*, have very similar spectral signatures and cannot be separated accurately using spectral remote sensing methodologies.

Eucalyptus spp., a group of very fast growing heliophytes, are not formally listed as invasive species. This genus is not well adapted to the cold winters of this area (Geldres and Schlatter, 2004), however they are planted widely in Chile, raising concerns about their role in landscape change and native forest loss in numerous studies (Altamirano et al., 2013; Echeverría et al., 2012; Heilmayr et al., 2016).

2.3. Data sources

Medium spatial resolution multispectral Sentinel-2 level-1C imagery (Fig. 1) was used to perform the land cover mapping in this study. Sentinel-2 (comprising two satellites: Sentinel-2a and 2b) captures

imagery at 10 to 60 m spatial resolution in 13 spectral bands every five days at the equator. Its spectral capabilities include four visible and near-infrared bands (10 m resolution), six red-edge and shortwave-infrared bands (20 m) and three atmospheric and coastal aerosol bands (60 m) (Drusch et al., 2012). The four red edge bands are especially useful for vegetation monitoring (ESA, 2016) as this part of the electromagnetic spectrum captures a very abrupt increase in reflectance of plants (Gitelson and Merzlyak, 1996). Rich data acquisition in the red edge section of the spectrum strengthens the capabilities of Sentinel-2 to separate vegetation and other land cover types, making it a well-suited sensor for this application.

Four Sentinel-2 images were analysed in this study (Fig. 1), acquired from the Copernicus Open Access Hub (ESA, 2014). For each study area, imagery was acquired in both summer and winter seasons (Malcahuello summer image: 30/11/2016, winter image: 19/05/2017; Villarrica summer image: 30/11/2016, winter image 26/05/2017). This multi-season approach was used because it is expected to improve the classification accuracy due to the phenology of the species, which changes their reflectance throughout the year. This approach was demonstrated to improve land cover classification previously (Langley et al., 2001). Additionally, the 30 m resolution ASTER (Advanced Spaceborne Thermal Emission and Reflection Radiometer) Global Digital Elevation Map Version 2 (ASTER GDEM Validation Team et al., 2011) was utilised in the classifications as some species distributions are expected to exhibit an altitudinal pattern.

Training and validation datasets for the classifications were acquired from a combination of sources. Firstly, field data were collected during two field campaigns in June-July 2017 and January-February 2018. During these campaigns, land cover surveys were conducted across each study site. The abrupt topography and forest structure made access to certain areas challenging; therefore, data collection locations were limited to areas accessible by car or on foot. Although this approach influenced the extent to which survey locations were fully random, a variety of land cover types were surveyed within the logistical constraints of the study area with 146 points surveyed in Malcahuello, and 132 in Villarrica. At each point, the data collected included: GPS location, qualitative description of the site; list of the tree and shrub species (both native and alien) present in a 5 m radius circular plot from the point; cardinal photographs (looking north, east, south, west from the survey location); and, for some locations, additional upwards photos showing forest canopy cover/stand density. Additional photos of features of interest in and around the survey points were also taken. Roughly, 10% of the field data was used for training the classifier, and 90% for accuracy assessment for each study site.

To supplement the field data, further reference information was extracted from the following sources: (1) the GPS recorded field photos,

(2) a pre-existing 30 m resolution land cover map of Chile created in 2014 with an overall accuracy of 80% (Zhao et al., 2016); and (3) from the vegetation cadastre created by the Chilean National Forest Corporation (CONAF) for management and national statistics purposes (Corporación Nacional Forestal, 2017) with 0.5 ha minimum mapping unit and dating from 2013 or 2008, depending on area. The 2014 land cover map and the CONAF's cadastre were only used to train the classifier if both data sources consistently identified the same land cover class for a particular point location. Very High Resolution (VHR) satellite imagery of the study areas (Heilmayr et al., 2016), together with WorldView 3 imagery, were used to complement the reference data.

Finally, to derive additional variables for modelling the distribution of the alien species, road locations were extracted from the Chilean Ministry of Public Works (MOP) website, rivers and lakes locations from that of the Chilean General Directorate of Waters (DGA), and number of forest fires between 2015 and 2017 from CONAF.

2.4. Image pre-processing

Layer stacking was performed separately for each summer and winter image for both study areas, creating four 10-band multispectral data sets with 10 m pixel size (the atmospheric and coastal aerosol bands were excluded). Cloud and cloud shadow masking was performed on each image using histogram thresholding. The cloud/shadow masks from both summer and winter images for the respective sites were merged, with this merged mask used to exclude all corresponding areas of the composited images from further analysis. For each study area, a 20-band multi-season composite image (comprising both summer and winter images) was then created; with a further composite incorporating the 20-band composite plus the ASTER elevation layer. Further radiometric or atmospheric correction was not necessary since this study does not compare spectral pixel values directly over time (Bakr et al., 2010; Lin et al., 2015; Song et al., 2001). Rather, analysis involves thematic land cover classifications (comprising discrete class values rather than continuous spectral values), each of which is independently accuracy assessed. Previous studies involving land cover classifications have successfully followed this approach (Kamlun et al., 2016; Pekkarinen et al., 2009).

2.5. Land cover classification

The classification system (Table 1) was designed to provide separation of the different types of alien and native forests. It comprised 10 land cover classes, including six discrete vegetated classes. The classification was designed using the FAO (Food and Agriculture Organization) land cover classification system (Di Gregorio and Jansen,

Table 1
Land cover class nomenclature, abbreviations and class descriptions.

Land cover class	Class description	
Alien tree stands	Coniferous plantations and invasion (CP) Broadleaved plantations and invasion (BP)	<i>Pinaceae</i> family (<i>Pinus radiata</i> , <i>Pinus contorta</i> , <i>Pinus ponderosa</i> , <i>Pinus sylvestris</i> , <i>Pseudotsuga menziesii</i>) <i>Eucalyptus</i> genus (<i>E. globulus</i> and <i>E. nitens</i>)
Native forests	Native <i>Nothofagus</i> spp. (broadleaved) forests (NNF) Native <i>Araucaria araucana</i> (mixed coniferous-broadleaved) forest (ANF)	<i>Nothofagus</i> spp. or laurel forests (<i>Laureliopsis philippiana</i> , <i>Aextoxicon punctatum</i> , <i>Eucryphia cordifolia</i> , <i>Caldcluvia paniculata</i> , <i>Weinmannia trichosperma</i> , etc.) Patagonian forests (<i>Nothofagus</i> spp. mixed with <i>A. araucana</i> and high mountain shrubs).
Shrubs		<i>Chusquea culeou</i> mixed with <i>Holcus lanatus</i> , <i>Rosa moschata</i> , <i>Rubus ulmifolius</i> or other, less common shrubs. Also, large shrubs (or very small trees) such as <i>Aristotelia chilensis</i> , <i>Ovidia pillopollo</i> . Sometimes including smaller size, stunted <i>Nothofagus</i> spp. individuals.
Grassland		Agricultural grasslands or livestock grazing plots dominated by <i>Holcus lanatus</i> , <i>Agrostis alba</i> , <i>Nothofagus obliqua</i> and <i>Luma apiculata</i> . 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.

2005), and the classification scheme used in the 2014 Chile land cover map (Zhao et al., 2016) as basic guidelines, with adaptations for this study.

For each study site, land cover classification was conducted separately on the four different image data sets: the 10-band summer image, the 10-band winter image, the 20-band multi-season composite image and the multi-season + a digital elevation model (DEM) composite image. Classification was conducted using a random forest classifier (Breiman, 2001) in R (R Development Core Team, 2011) using the randomForest package (Liaw and Wiener, 2002). Random forests are a machine learning algorithm that work as an ensemble model. This means that it uses the results from numerous different models to calculate a result that will be more accurate than the result of any of the separate models (Horning, 2010; Rodríguez-Galiano et al., 2012). This ensemble model classifies land covers accurately (Marston and Giraudoux, 2019; Müllerová et al., 2017; Rodríguez-Galiano et al., 2012), and is well established for classifying multispectral imagery (Rodríguez-Galiano et al., 2012). Random forests have been used previously to detect invasive plant species by Naidoo et al. (2012) with 87.68% accuracy, Peerbhaya et al. (2016) with 91.33%, 85.08%, and 67.90% accuracy, and Müllerová et al. (2017) with 60% to 100% accuracy depending on the case. Immitzer, Atzberger and Koukal (2012) used random forests to measure tree species diversity, identifying ten different woody species in a temperate forest of the northern hemisphere.

The random forest classifier was trained using the reference data outlined in Section 2.3. The training dataset included 491 training polygons for Malalcahuello and 439 for Villarrica. This algorithm can generate an out-of-bag (OOB) internal accuracy assessment using part of the input reference data (Horning, 2010), but it tends to overestimate the accuracy when working with geographical data (Cánovas-García et al., 2017). For this reason, an independent accuracy assessment was performed by comparing a sample of classified points against reference data (outlined in Section 2.3) to construct a confusion matrix. The validation dataset had 542 points for Malalcahuello and 469 for Villarrica.

2.6. Landscape metrics

To quantitatively assess whether the occurrence and prominence of alien invasive species are linked to specific landscape structural characteristics, in particular fragmentation, landscape metrics were calculated from the land cover classifications using Fragstats 4.2 (McGarigal et al., 2002). These metrics measure the geometric spatial configuration and aggregation of the patches in a landscape (McGarigal, 2014). It is possible to quantify numerous landscape metrics for a given area, but many metrics are co-correlated and thus redundant. Therefore, certain targeted landscape metrics were selected, based on applications in previous literature to best describe land cover spatial configuration patterns relevant to vegetation studies, and on their robustness to assess fragmentation. Ideally, a robust fragmentation index needs to be correlated with aggregation and as independent as possible of class area (Neel et al., 2004). Although a larger class area would normally mean a better-connected class, if the metric was too dependent on class area, the connectivity of the less widespread habitats would be underestimated, even though they were not markedly fragmented.

The class level landscape metrics (table 2) calculated were the patch density, the mean patch area, the largest patch index, the perimeter area fractal dimension and the clumpiness index. They were all calculated for each vegetated land cover: *Nothofagus* spp. native forest, *Araucaria* spp. native forest, shrubs and grasslands. Two landscape level metrics (Table 2) were also included: the aggregation index and the Shannon's diversity index.

Given the considerable variability in biogeographical and topographical characteristics across the study areas, it is inappropriate to generate global landscape metric values as this 'global study area' approach incorporating both highly managed and more natural

landscapes, may conceal important localised patterns (Miranda et al., 2015). Hence, a series of subset locations within the overall study areas were selected, with landscape metric values calculated individually for each subset. This enables both local scale relationships to be assessed, as well as broader patterns for each study site by pooling sub-site results. There is a natural divide between lower altitude areas that are more actively managed including alien species plantations and agriculture, and less impacted higher altitude areas including protected areas, where invasion has also been recorded (Pauchard et al., 2014; Peña et al., 2008). The boundary dividing the lower and higher altitude subsets for each study area is 600 m elevation, with this being the minimum altitude at which *A. araucana* can thrive (Premoli et al., 2013). To ensure that both subsets are accurately represented, a systematic sampling approach was taken using a regularly spaced buffered point grid. The sampling buffers were 19 km² circular plots. This size is a compromise between achieving a meaningful sample size (> 100 in each site) and a large enough buffer to capture the landscape heterogeneity. This type of sampling strategy has been previously used in large scale landscape studies (Gonzalez-Abraham et al., 2007; McGarigal et al., 2009). Each point and sampling buffer were identified as either in the low (valley subset) or high (Andes subset) altitude areas using a 30 m ASTER digital elevation model. Additionally, a targeted set of sampling buffers were created inside the protected areas to ensure that these areas, which constitute a small proportion of the overall study areas, were represented. For Malalcahuello, 104 sampling buffers were located in the low altitude subset, 64 in the high altitude subset and 18 inside protected areas. For Villarrica, 75 were at low altitude, 126 at high altitude and 48 inside protected areas. The minimum patch size considered was 100 m², with the intention of capturing the earliest stages of invasion possible at this resolution. However, it is important to mention that, at this medium resolution, the predominant alien tree stand detection will be plantation, especially in the case of larger patches and in easily accessible areas.

2.7. Modelling

To better understand the influence of fragmentation (represented by the landscape metrics) and other landscape variables (topographical, hydrological and related to human influence) on the presence and abundance of alien trees, random forests were used in a regression capacity to identify the respective importance of the variables driving the distribution of alien tree cover. This is key for forestry and conservation management in areas susceptible to invasion, as the presence of alien tree patches is a source of propagule pressure, increasing the risk of invasion (Gundale et al., 2014). The random forest algorithm is especially useful for large and complex datasets (Cutler et al., 2007), and is robust to outliers, noise and over-fitting (Breiman, 2001). Random forest analysis was performed for each study area, with mean patch area (AREA_MN) of coniferous alien tree stands (CP) as the response variable. The 27 explanatory variables were derived from the landscape metrics, the land cover map, topography, hydrology and human activity.

Broadleaved alien trees (BP) were neither modelled nor included as explanatory variables for the following reasons: (1) there is not enough area of this land cover class to guarantee a reliable model; (2) they are not good competitors in this climate (Geldres and Schlatter, 2004); and (3) they are not listed as invasive by the International Union for Conservation of Nature (IUCN Invasive Species Specialist Group, 2006).

The explanatory variables based on the landscape metrics and the land cover map relate to fragmentation of native vegetated covers, which has been reported as correlated with biological invasions (Arellano-Cataldo and Smith-Ramírez, 2016; León Cordero et al., 2016; Tella et al., 2016). The explanatory variables based on the land cover map were the proportion of *Nothofagus* spp. native forest area, the proportion of native *A. araucana* forest area, the proportion of shrubs area, the proportion of grasslands area and distance to other coniferous

Table 2
Landscape metrics calculated in this study and metric descriptions.

Landscape metrics	Abbreviation	Description
Class level (calculated for a specific land cover class)	Patch density	PD PD is the number of patches of a class divided by the whole area of the landscape (McGarigal, 2014). This has been previously used in this environment by several authors (Altamirano et al., 2013; Echeverría et al., 2012, 2008; Molina et al., 2016).
	Mean patch area	AREA_MN AREA_MN calculates the sum of the total area of all the patches of a class, divided by the number of patches (McGarigal, 2014). This has been used before for fragmentation studies in Chile by Hernández et al. (2016).
	Largest patch index	LPI LPI is a metric that calculates dominance as the percentage of the total area of the landscape that is comprised by the largest patch of the class (McGarigal, 2014). This has been selected based on previous uses in studies in the south-central Chilean temperate forests (Altamirano et al., 2013; Echeverría et al., 2012; Molina et al., 2016).
	Perimeter area fractal dimension	PAFRAC PAFRAC informs about the shape complexity of the patches, whatever their extent (McGarigal, 2014). It is calculated as 2 divided by the slope of the regression line of the logarithm of patch area (m ²) against the logarithm of patch perimeter (m) (McGarigal, 2014). A Euclidean geometry has a value of 1, increasing gradually towards 2 as the perimeter gets more convoluted (McGarigal, 2014). It is correlated with aggregation and not very correlated with habitat abundance (Neel et al., 2004; Wang et al., 2014). For class areas between 5% and 80%, PAFRAC is independent to changes in area (Neel et al., 2004).
	Clumpiness index	CLUMPY CLUMPY is a measure of aggregation. It is the proportional deviation of the proportion of like class adjacencies involving from what would be expected under a spatially random distribution (McGarigal, 2014). It ranges between -1 for maximum disaggregation, and 1 for maximum aggregation. A value of 0 means a random distribution (McGarigal, 2014).
Landscape level (calculated for the total patches in the landscape)	Aggregation index	AI AI shows the percent of like cell adjacencies (McGarigal, 2014). This metric has a value of 0 for a maximally disaggregated landscape (McGarigal, 2014). This has been used in previous deforestation studies in Chile (Echeverría et al., 2006).
	Shannon's diversity index	SHDI SHDI is calculated using the following formula, where P _i is the proportion of each class: $SHDI = -\sum_{i=1}^m (P_i * \ln P_i)$ (McGarigal, 2014). SHDI is used here as a measure of land cover diversity. It has a value of 0 for landscapes made of only one class and increases as class richness increases (McGarigal, 2014). A closely related diversity index has been previously used in Chile by Molina et al. (2016).

alien patches. The proportions were calculated for each sampling buffer using the land cover map previously created and the Geospatial Modelling Environment software package (Beyer, 2012). The distance to other coniferous alien patches was calculated as the Euclidean distance from the centre of the sampling buffer to the nearest coniferous alien patch.

The topography (elevation, slope and aspect at buffer centre-point) and hydrology (Euclidean distance to rivers and lakes from buffer centre-point) determine microclimatic conditions potentially limiting the presence of alien species. The location of these water features was provided by the Chilean General Directorate of Waters. Finally, explanatory variables related to human activity were distance to roads, distance to urban areas, and number of forest fires between 2015 and 2017. Road locations were provided by the Chilean Ministry of Public Works, urban area locations derived from the land cover map, and forest fire frequency was provided by the Chilean National Forest Corporation. Distances to roads and urban areas were calculated as the Euclidean distance from the centre of the sampling buffer to the nearest road or urban patch. Although forest fires can occur naturally, the majority start as a result of human negligence near roads, paths or agricultural areas. The frequency of forest fires was calculated as the number of fires that occurred inside a sampling buffer between 2015 and 2017.

To achieve a parsimonious model, a nested iterative method known as recursive feature elimination was applied. At the end of each random forest run, a fixed proportion (in this case, 10%) of the variables of lowest importance are removed before running the model again. The lowest importance variables are those that have the smallest impact on the mean square error (MSE) when removed (those with the smallest OOB percentage of increment in MSE). This iterative process continues for as long as the OOB errors keep decreasing. Once the iterations are complete, the set of variables with the smallest OOB error is selected (Díaz-Uriarte and Alvarez de Andrés, 2006; Genuer et al., 2010). The final model comprises the set of explanatory variables with the smallest OOB error, with these explanatory variables ranked in order of

influence on the response variable. Finally, partial dependence plots were generated for each variable, to illustrate the nature of the relationships present (Marston and Giraudoux, 2019).

3. Results

3.1. Land cover classification

Four land cover classifications were produced for each study area, corresponding to the winter image, the summer image, the multi-season composite, and the multi-season composite plus DEM. The overall accuracies for the Malalcahuello classifications were 0.69, 0.77, 0.78 and 0.81 respectively. For Villarrica, classification accuracies were 0.75, 0.82, 0.85 and 0.86 respectively. As expected, the highest accuracies were achieved for the multi-season plus DEM composite, as it contains additional information to train the classifier. The summer classification accuracies were higher than those of the winter classification, likely due to the spectral variability between land cover types being less acute in winter as a result of senescing vegetation. Table 3 shows the accuracies for all land cover classes for each composite. The full confusion matrices can be found in the Appendix A. Both study areas had extremely high accuracies for the water, urban, bare and snow classes, as those are very spectrally distinct from the remaining classes. The alien tree stands were generally accurately mapped at both sites, except for broadleaved plantations in Villarrica. This low accuracy was expected, as broadleaved plantations are scarcer in Villarrica than Malalcahuello, limiting the quality of the training data for this class. The native woody classes (*Nothofagus* spp. native forests, araucaria native forests and shrubs), although essentially different types of vegetated areas, have a certain degree of similarity because all three include *Nothofagus* spp. in their biological assemblages. This resulted in a degree of confusion among them. However, this confusion is not a principal concern for this study, as the three classes constitute variations of the Chilean Valdivian temperate forest, and they are well separated from the alien tree stands in the land cover maps.

Table 3

Land cover classification accuracies calculated using confusion matrices for the different image data composites. The following abbreviations are used: CP = coniferous plantations and invasion, BP = broadleaved plantations and invasion, NNF = *Nothofagus* spp. native forests, ANF = *Araucaria araucana* native forests, winter = winter composite, summer = summer composite, multi. = multi-season composite, multi. + DEM = multi-season with DEM composite, P = producer's accuracy and U = user's accuracy.

Land cover classes	Malalcahuello								Villarrica							
	Winter		Summer		Multi.		Multi. + DEM		Winter		Summer		Multi.		Multi. + DEM	
	P	U	P	U	P	U	P	U	P	U	P	U	P	U	P	U
CP	0.78	0.96	0.73	0.98	0.72	0.86	0.78	0.89	0.81	0.80	0.79	0.94	0.84	0.89	0.76	0.92
BP	0.81	0.88	0.77	0.90	0.74	0.85	0.77	0.9	0.60	0.56	0.73	0.55	0.67	0.63	0.73	0.46
NNF	0.70	0.40	0.84	0.46	0.92	0.51	0.86	0.58	0.79	0.69	0.88	0.82	0.89	0.82	0.85	0.83
ANF	0.35	0.60	0.45	0.73	0.42	0.75	0.60	0.77	0.37	0.57	0.51	0.72	0.54	0.70	0.66	0.82
Shrubs	0.43	0.33	0.81	0.56	0.77	0.61	0.68	0.53	0.44	0.30	0.94	0.55	0.79	0.57	0.81	0.58
Grass	0.57	0.86	0.76	1	0.83	0.98	0.81	1	0.67	0.98	0.75	1	0.80	0.99	0.88	1
Water	0.77	0.95	0.83	0.98	0.85	0.98	0.85	0.98	0.96	1	0.93	1	1	1	0.96	1
Bare	1	0.74	1	0.78	1	0.7	1	0.8	0.74	0.80	0.63	0.80	0.66	0.89	0.79	0.83
Snow	1	0.87	1	1	1	1	1	1	0.98	0.95	0.90	0.95	0.98	1	1	1
Urban	1	0.89	1	1	1	1	1	0.98	0.95	0.89	0.98	0.83	1	0.84	1	0.95
Overall accuracy	0.69		0.77		0.78		0.81		0.75		0.82		0.85		0.86	

Figs. 2 and 3 show the land cover maps generated using multi-season + DEM composite classification, the most accurate approach (highest overall accuracy), for Malalcahuello and Villarrica respectively. These land cover classifications were used in the next steps of this study for the landscape analysis and the modelling of *Pinaceae* area.

Table 4 shows the vegetated land cover class areas for the multi-season with DEM classifications for the two study areas and their different subsets: the lower altitude subset (valley, below 600 m) and the high altitude subset (Andes, above 600 m). Protected areas are all within the Andes subset, but are presented separately as well due to their conservation status.

In Malalcahuello, 343 km² are covered by coniferous alien trees, while 556 km² are broadleaved alien trees (Table 4). These alien tree stands are located mostly in the north-west of the study area, at lower altitudes and outside national park boundaries. However, some medium sized, irregular coniferous alien patches are observed inside the Malalcahuello National Park, covering 4 km² in the east of the study area. These patches are embedded in a mix of shrubs and *A. araucana* forest patches. These land cover classes are naturally open, being an ideal environment for alien pine trees to spread (Ledgard, 2001; Taylor et al., 2016). Native *Nothofagus* spp. forests were distributed throughout the study area, although there is a clearer dominance of these land covers to the centre and east of this area, where altitude increases. At high altitudes (> 600 m), the dominant native forests are *Nothofagus* spp. (NNF) and *A. araucana* (ANF). In fact, in the high altitude area the presence of alien trees is restricted to 75 km² for coniferous alien species, and 81 km² for broadleaved alien species. *A. araucana* forests were restricted to the eastern part of the study area, at the highest altitudes.

Villarrica has a smaller overall area covered by alien trees, with 236 km² covered by coniferous alien species, and 143 km² covered by broadleaved alien species (Table 4). Of this, 147 km² of coniferous and 112 km² of broadleaved are located in lower altitude areas, to the western part of the study site. A considerable amount of coniferous alien patches is, however, located inside the boundaries of the Villarrica National Reserve and a UNESCO Biosphere Reserve to the south of the study area, diminishing their conservation value. This agrees with the findings of Altamirano et al. (2010) about other protected areas elsewhere in Chile. In fact, 82 km² of coniferous alien tree cover and 23 km² of broadleaved alien tree cover are located within the boundaries of protected areas in this study. Broadleaved alien trees are primarily located in the north-west of the study area. The *A. araucana* native forests are primarily located to the east and at the highest altitudes, with *Nothofagus* spp. forests occurring mostly in the centre of the site and covering a large area.

3.2. Landscape metrics

Table 5 summarizes the results obtained in the landscape metric assessment of the thematic land cover maps. Full detail of the landscape metrics results can be found in the Appendix B.

Both study sites exhibit similar altitudinal patterns, with higher patch density, clumpiness index and dominance (LPI) of alien species in the valley area than in the Andes. This was expected, as accessibility plays a major role when establishing new productive plantations. In addition, eucalyptus trees cannot tolerate the harsh mountain winters of the Andes, being restricted to lower altitudes in this environment. Alien patch density is the highest within protected areas in the Villarrica site. This is concerning, as it could diminish the conservation value of these protected areas. This is not the case in Malalcahuello's protected areas, which show the lowest patch densities for alien species overall.

Like alien tree stands, native forests also show an altitudinal pattern. Both sites have slightly higher patch density of native *Nothofagus* spp. forests (NNF) in the valleys. *Nothofagus* spp. forests include a range of species of this genus, some of them able to reach medium-high mountain conditions, but also are a very common class in lowland areas. In these lowland areas, they are found as hedgerows in agricultural areas, as plantation for firewood for the local communities or as relict patches of native forests among other land uses. *A. araucana* native forests (ANF), on the other hand, are restricted to higher altitudes, and the PD results are consistent, being much higher in the Andes areas of both study sites. Malalcahuello's protected areas have the greatest PD for ANF overall in that study area. However, Villarrica has the highest density of the endangered *A. araucana* forest in the Andes subset, not specifically inside the protected areas. This, together with the fact that there are coniferous alien tree patches inside, suggests that the protected areas in Villarrica may be ineffective for *A. araucana* native forest conservation.

Alien tree classes, both coniferous (CP) and broadleaved (BP) plantations, have higher patch densities (PD) in Villarrica. However, their mean patch area (AREA_MN) and largest patch index (LPI) are higher in Malalcahuello. This means that Villarrica's alien populations are more fragmented and scattered than those of Malalcahuello. This is further confirmed by the clumpiness index (CLUMPY), a measure of aggregation, which is lower in Villarrica. The distribution of patch density data (Appendix B, Fig. B1) shows that, even though the mean patch density (Table 5) of Villarrica is higher for most land cover classes, its values vary considerably, indicating that while some parts had very few patches, others had a complex mix of land covers.

The native forest classes have higher patch densities but less

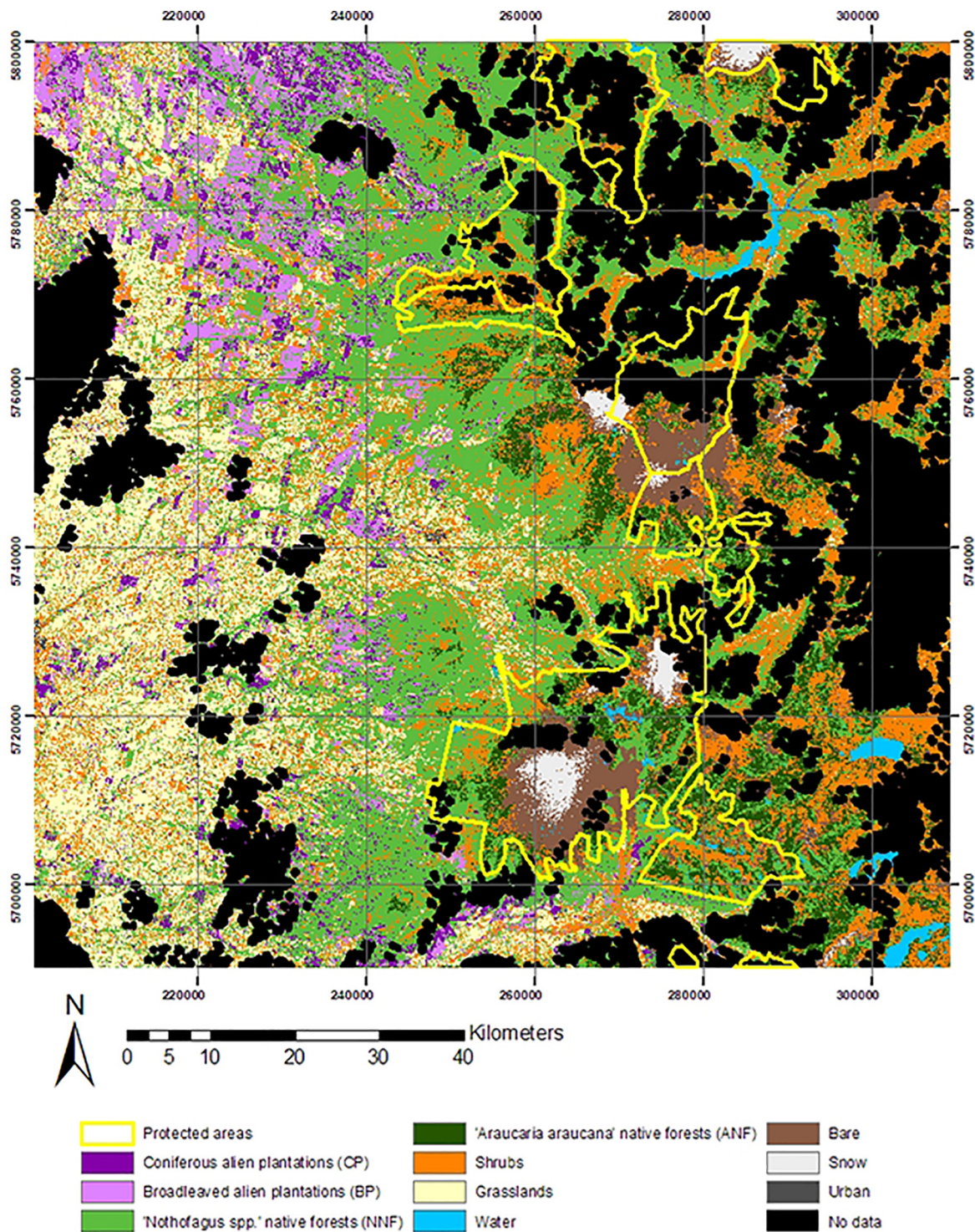


Fig. 2. Land cover classification of Malalcahuello using the multi-season with DEM composite.

aggregation (CLUMPY) in Villarrica, meaning that the native forest cover is more fragmented. Malalcahuello had a larger mean patch area of *A. araucana* forests and a smaller mean patch area of *Nothofagus* spp. However, Villarrica's larger area of NNF results from a series of large outliers (Appendix B, Fig. B2), while Malalcahuello has a more consistent normal distribution. The dominance metric (LPI) indicates that the native *Nothofagus* spp. forest dominates in all subsets of the data. *A. araucana* has high patch densities and low mean patch areas inside protected areas in each study site. This is especially concerning, as it means that *A. araucana* forest is fragmented inside protected areas.

Shrubs are a mixed class that comprises numerous native shrub species, including some of the *Nothofagus* genus. This wide range of species makes the class ubiquitous, having high patch densities in all cases. Shrubs are associated to various land uses and form very different spatial patterns depending on location. Both the Andes and the valley subsets at both sites show a high PD of shrubs, but there is an obvious structure where more patches are present in the valley. However, the Andes have larger mean patch area of shrubs, more dominance (LPI) and are more aggregated (CLUMPY). This is because high mountain *Nothofagus* spp. have a shrub lifeform and, together with *A. araucana*,

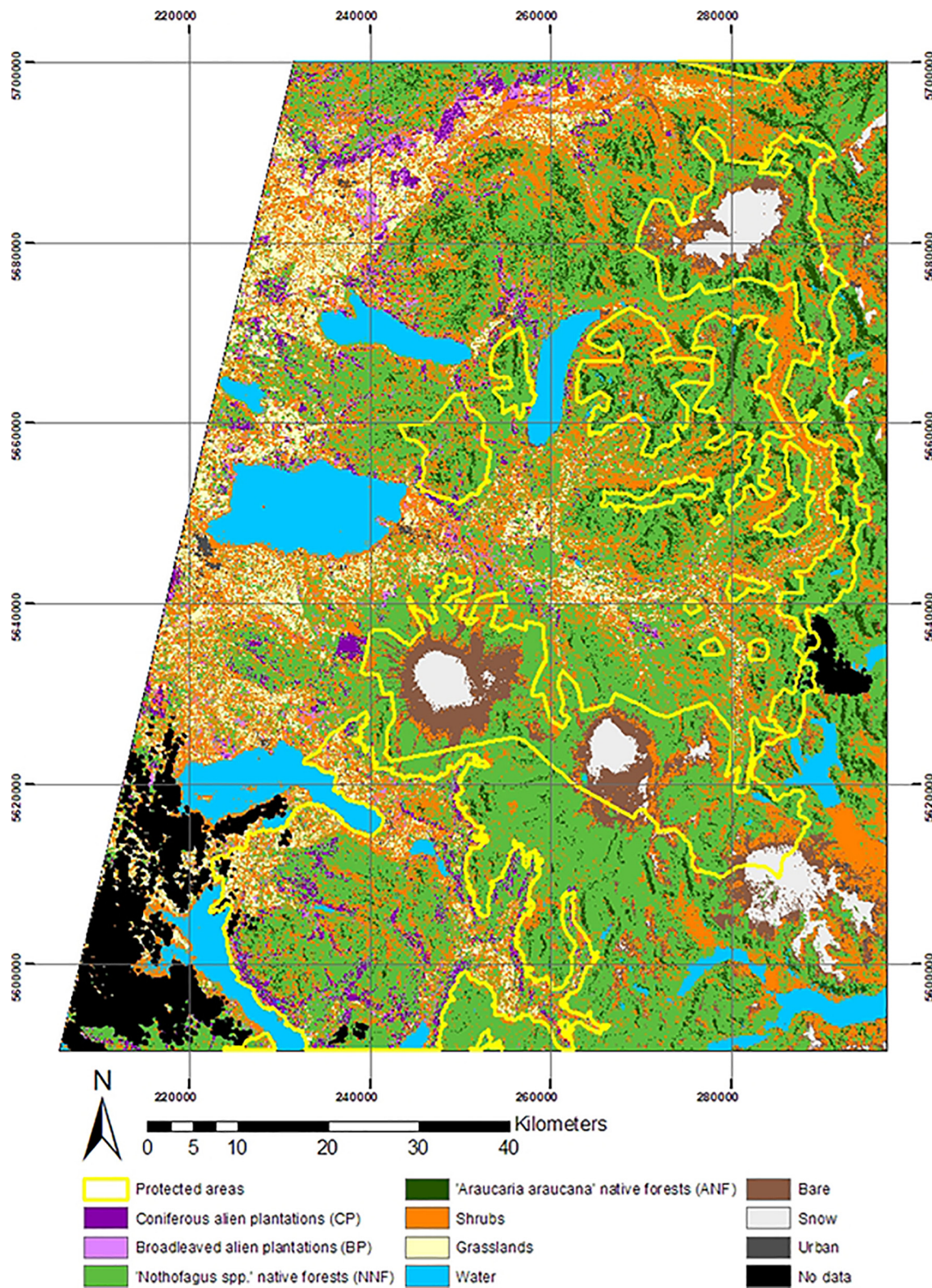


Fig. 3. Land cover classification of the Villarrica using the multi-season with DEM composite.

are among the few woody species that can tolerate high mountain conditions.

The perimeter-area fractal dimension (PAFRAC) does not show any strong patterns, possibly influenced by the too coarse 10 m resolution of the Sentinel-2 imagery and the subsampling approach.

A lower value of the landscape level metric aggregation index (AI) is indicative of a more fragmented landscape mosaic. The results show

that Malcalahuello is a more aggregated (less fragmented) landscape than Villarrica. For both sites, the maximum aggregation is achieved inside protected areas, with the Andes subsets being more aggregated than the corresponding Valley subsets. This is positive from a conservation perspective, as a more aggregated landscape means larger and better connected habitats.

The landscape level metric Shannon's diversity index (SHDI)

Table 4 Vegetated land cover class areas for the multi-season with DEM classifications. The land cover classes use the following abbreviations: CP = coniferous plantations and invasion, BP = broadleaved plantations and invasion, NNF = *Nothofagus* spp. native forests, ANF = native *A. araucana* forest. *Whole* identifies the complete study area, *valley* corresponds to the low areas (below 600 m), *Andes* refers to the higher altitudes (above 600 m) and *protected* are the areas inside protected areas.

Land cover classes	Malacahuello						Villarrica									
	Whole		Valley		Andes		Protected		Whole		Valley		Andes		Protected	
	Area (km ²)	Area (%)	Area (km ²)	Area (%)	Area (km ²)	Area (%)	Area (km ²)	Area (%)	Area (km ²)	Area (%)	Area (km ²)	Area (%)	Area (km ²)	Area (%)	Area (km ²)	Area (%)
CP	342.62	2.84	267.82	5.74	75.43	1.02	4.02	0.26	236.32	2.74	146.90	4.88	89.51	1.60	82.30	3.61
BP	556.15	4.61	475.27	10.18	81.06	1.10	0.40	0.03	142.52	1.65	111.77	3.71	30.76	0.55	21.71	0.95
NNF	2997.45	24.86	1082.52	23.19	1940.52	26.26	302.01	19.53	3513.61	40.76	667.08	22.15	2934.99	52.34	1068.10	46.78
ANF	710.22	5.89	15.21	0.33	708.70	9.59	208.7	13.50	519.43	6.03	32.35	1.07	518.64	9.25	166.55	7.30
Shrubs	1782.49	14.79	670.53	14.36	1135.62	15.37	192.39	12.44	2104.60	24.41	947.04	31.44	1198.24	21.37	473.40	20.74
Whole	12,055	100	4668	100	7390	100	1546	100	8621	100	3012	100	5608	100	2283	100

indicates that lower altitude areas have a greater mix of land cover classes in both sites. It is counterintuitive, however, that protected areas have slightly more diversity than the Andes subsets for both sites, as protected areas should have an abundance of native vegetated areas but a small amount of alien classes, agricultural lands (grasslands) and built up areas, consequently being less varied. This means that other land cover types aside from native forests are in fact present in protected areas, potentially being a source of disruption or an entryway for alien species in protected areas. Overall, Malacahuello shows a slightly more diverse landscape than Villarrica.

3.3. Random forest modelling of Pinaceae

Random forest analysis established the relative importance of the explanatory variables in relation to the mean patch area of coniferous alien trees (*Pinus* spp. and *Pseudotsuga menziesii*). The stepwise removal of explanatory variables (Section 2.7. modelling) resulted in a model for Malacahuello retaining 13 explanatory variables, and a model for Villarrica comprising 27 variables (table 6).

Both models share the Shannon's diversity index in the top quartile. This is consistent with previous studies showing that landscape heterogeneity (Altamirano et al., 2016) and fragmentation (Arellano-Cataldo and Smith-Ramírez, 2016; León Cordero et al., 2016; Tella et al., 2016) play major roles in the invasion process. In both models, elevation was among the top half of ranked variables. Elevation has been previously reported as having a relation with invasive plant species richness (Gassó et al., 2009; Pauchard and Alaback, 2004). Some of the grassland metrics (largest patch index, mean patch area and proportion) were also in the top half. More than half of the variables of the simpler model (Malacahuello) are among the top half most important variables in Villarrica. These are the diversity index and the grassland related metrics already mentioned, together with patch density of shrubs and distance to nearest lake.

Notably, the variable ranked as being of highest importance in Villarrica, distance to other pine patches, was eliminated during the stepwise removal process when developing the Malacahuello parsimonious model. Distance to seed sources (Richardson et al., 2000) and propagule pressure (Gundale et al., 2014) are generally considered relevant to, and major drivers of, invasion in Chile (Altamirano et al., 2016; Arellano-Cataldo and Smith-Ramírez, 2016; Pauchard et al., 2016) and Argentina (Giorgis et al., 2016, 2011). At this resolution, it is highly likely that a large proportion of the alien patches detected are a result of plantation, rather than natural invasion, as at the 10 m spatial resolution of Sentinel-2 it would not be possible to detect the small patches at the initial stages of invasion. This way, the patterns of invasion would be masked by the patterns of plantation, especially in the case of Malacahuello, where distance to seed sources is not present in the model. Even though Malacahuello and Villarrica are geographically very close, they exhibit different land uses. The presence of distance to other *Pinaceae* stands (seed sources) as the most important explanatory variable in the Villarrica model, could indicate that this study site is being naturally invaded by these trees, even at this scale, while the establishment of new pine tree stands in Malacahuello is mostly a result of human plantation.

Coniferous alien tree stands are of anthropogenic origin, being planted for wood and fibre. Consequently, directly human related variables like distance to roads and urbanised areas were expected to be of high importance. However, none of these variables were present in the Malacahuello model, and in Villarrica, with only the distance to roads among the top half. This could, again, be a scale related issue. The roads dataset might not contain very narrow forest roads or seasonal paths within forested areas. However, there could also be other more important socio-economic drivers determining plantation locations of new alien pines, such as land ownership, the possibility of other more profitable activities in the area such as farming, or the availability of suitable machinery and land access to establish plantations further

Table 5

Overall class level and landscape level metrics for each study area. The values are calculated as the mean of the subsamples (5 km diameter circular plots) within each altitude subset. In the table, *whole* identifies the complete study area, *valley* corresponds to the low areas (below 600 m), *Andes* refers to the higher altitudes (above 600 m) and *protected* are the areas inside protected areas. The metrics are abbreviated as follows: PD = patch density, AREA_MN = mean patch area, LPI = largest patch index, PAFRAC = perimeter area fractal dimension, CLUMPY = clumpy index, AI = aggregation index, SHDI = Shannon's diversity index. The land cover classes are abbreviated as follows: CP = coniferous plantations and invasion, BP = broadleaved plantations and invasion, NNF = *Nothofagus* spp. native forests, ANF = native *A. araucana* forest.

			Class level metrics					Landscape level metrics	
			PD (patches / 100 ha)	AREA_MN (ha)	LPI (%)	PAFRAC	CLUMPY	AI	SHDI
Malalcahuello	Whole	Landscape	x	x	x	x	x	87.40	1.21
		CP	11.72	0.41	1.52	1.35	0.75	x	x
		BP	20.70	0.44	3.68	1.38	0.73	x	x
		NNF	30.15	1.89	26.29	1.38	0.80	x	x
		ANF	14.98	0.61	7.99	1.38	0.75	x	x
		Shrubs	58.98	0.38	4.88	1.43	0.71	x	x
	Valley	Landscape	x	x	x	x	x	85.68	1.31
		CP	14.05	0.58	2.24	1.35	0.82	x	x
		BP	27.73	0.58	5.47	1.38	0.77	x	x
		NNF	33.52	1.20	15.66	1.39	0.79	x	x
		ANF	0.70	0.31	0.08	1.41	0.74	x	x
		Shrubs	70.93	0.24	2.75	1.45	0.67	x	x
	Andes	Landscape	x	x	x	x	x	89.69	1.07
		CP	8.48	0.18	0.52	1.35	0.66	x	x
		BP	9.77	0.23	0.90	1.38	0.66	x	x
		NNF	25.61	2.83	40.64	1.38	0.82	x	x
		ANF	16.66	0.64	8.92	1.38	0.75	x	x
		Shrubs	43.05	0.57	7.72	1.41	0.76	x	x
	Protected	Landscape	x	x	x	x	x	90.47	1.11
		CP	2.26	0.11	0.11	1.32	0.63	x	x
		BP	1.44	0.06	0.01	1.44	0.39	x	x
		NNF	30.46	0.90	14.89	1.39	0.77	x	x
		ANF	23.67	0.73	12.18	1.40	0.73	x	x
		Shrubs	28.52	0.47	7.75	1.38	0.75	x	x
Villarrica	Whole	Landscape	x	x	x	x	x	86.81	1.15
		CP	16.84	0.21	1.35	1.41	0.65	x	x
		BP	35.10	0.05	0.38	1.45	0.38	x	x
		NNF	34.49	1.95	33.30	1.42	0.76	x	x
		ANF	38.45	0.16	2.60	1.43	0.60	x	x
		Shrubs	48.61	0.66	13.16	1.40	0.76	x	x
	Valley	Landscape	x	x	x	x	x	85.56	1.26
		CP	19.70	0.24	1.61	1.39	0.67	x	x
		BP	47.04	0.08	0.72	1.47	0.51	x	x
		NNF	40.40	0.70	13.10	1.44	0.73	x	x
		ANF	11.52	0.05	0.32	1.42	0.37	x	x
		Shrubs	43.32	0.94	21.91	1.42	0.75	x	x
	Andes	Landscape	x	x	x	x	x	87.34	1.10
		CP	14.73	0.18	1.16	1.42	0.63	x	x
		BP	26.85	0.03	0.14	1.44	0.28	x	x
		NNF	31.96	2.48	41.92	1.41	0.78	x	x
		ANF	47.11	0.20	3.33	1.44	0.67	x	x
		Shrubs	50.87	0.55	9.42	1.40	0.76	x	x
	Protected	Landscape	x	x	x	x	x	88.18	1.12
		CP	30.50	0.22	2.47	1.43	0.62	x	x
		BP	50.61	0.03	0.09	1.48	0.35	x	x
		NNF	27.35	2.32	34.58	1.40	0.80	x	x
		ANF	29.01	0.17	2.06	1.42	0.65	x	x
		Shrubs	48.87	0.48	7.41	1.40	0.73	x	x

away from accessible areas.

To illustrate the nature of the relationships between the explanatory and response variable, a series of partial dependence plots were generated. Only plots demonstrating a high relative importance in the random forest models and showing a meaningful relationship with the mean area of *Pinaceae* are shown here, with the remaining partial dependence plots presented in the Appendix C. The plot for the Shannon's diversity index (Fig. 4), which is a highly relevant variable in both models (table 6) shows that as the mean patch area of alien pine trees in both study areas increases, patch diversity also increases. Previous models of invasion within Chile have shown comparable results (Altamirano et al., 2016). Although both models show an increase in area of coniferous alien trees with increasing landscape diversity, this effect is much more marked in Malalcahuello. This may be a

consequence of the different predominant land uses in the two areas (agriculture/forestry, and tourism), which have a direct impact on the patchiness of the landscape. An increase in SHDI is a sign of a more diverse and potentially fragmented landscape. These results were supported by the results of previous studies showing correlations between invasive species and fragmentation (Arellano-Cataldo and Smith-Ramírez, 2016; León Cordero et al., 2016; Tella et al., 2016).

Altitude was also expected to influence the distribution of alien species, as it has been previously reported as having a negative correlation with invasive plant species richness (Gassó et al., 2009; Pauchard and Alaback, 2004). In addition, productive plantations are theoretically preferentially located in more easily accessible low and flat areas, decreasing with altitude. This hypothesis was confirmed for Malalcahuello (Fig. 4), where the main economic activities are related to

Table 6

Relative importance of the explanatory variables ranked from highest (top) to lowest (bottom) importance. The abbreviation %IncMSE is the increase in mean squared error.

Malalcahuello		Villarrica	
Explanatory variables	%IncMSE	Explanatory variables	%IncMSE
Shannon's diversity index	28.54	Distance to nearest patch of coniferous alien tree stand	27.89
Largest patch index of grasslands	24.01	Shannon's diversity index	18.62
Elevation	23.24	Patch density of <i>Nothofagus</i> spp. native forest	16.33
Mean patch area of grasslands	21.61	Mean patch area of <i>Nothofagus</i> spp. native forest	15.86
Proportion of grasslands	20.51	Patch density of grasslands	13.89
Largest patch index of <i>A. araucana</i> native forest	18.55	Distance to lakes	13.74
Mean patch area of <i>A. araucana</i> native forest	18.17	Mean patch area of grasslands	12.93
Proportion of <i>A. araucana</i> native forest	17.53	Largest patch index of grasslands	12.64
Largest patch index of <i>Nothofagus</i> spp. native forest	16.72	Proportion of grasslands	12.58
Patch density of shrubs	16.07	Elevation	10.26
Patch density of <i>A. araucana</i> native forest	15.80	Aggregation index	9.50
Proportion of <i>Nothofagus</i> spp. native forest	14.37	Distance to roads	9.48
Distance to lakes	12.93	Patch density of shrubs	9.24
		Mean patch area of shrubs	9.08
		Mean patch area of <i>A. araucana</i> native forest	8.47
		Distance to nearest urban area	7.76
		Proportion of <i>Nothofagus</i> spp. native forest	7.66
		Patch density of <i>A. araucana</i> native forest	7.56
		Largest patch index of <i>A. araucana</i> native forest	7.01
		Largest patch index of <i>Nothofagus</i> spp. native forest	6.98
		Proportion of shrubs	6.54
		Proportion of <i>A. araucana</i> native forest	6.30
		Largest patch index of shrubs	5.89
		Distance to rivers	1.72
		Slope	1.18
		Forest fires 2015–2017	0.74
		Aspect	−0.09

forestry, agriculture and farming. However, Villarrica shows an inverse trend (Fig. 4), with slightly more coniferous alien tree cover at higher altitudes. Here, lower, easily accessible areas in Villarrica are more restricted to recreational and touristic uses, relegating forestry to more remote or higher locations.

Several metrics related to grassland cover were also among the top half ranked importance variables in both models. However, the partial dependence plots did not exhibit a particular trend. In almost all of them, the area of alien pine remained constant with an increasing grassland cover, as shown in Malalcahuello in Fig. 5.

The nature of the relationship between coniferous alien tree cover and *A. araucana* native forest is especially relevant given their endemic, endangered and culturally relevant status. *Pinus* spp. plantations, especially those that are abandoned or have long rotations, are a threat to this native ecosystem (Pauchard et al., 2014; Peña et al., 2008; Tella et al., 2016). Of the three metrics related to mixed native forest, only mean patch area is shown here, as all showed similar results (Fig. 5). There is an initial decrease in coniferous alien tree cover with an increase in mixed native forest, which is especially pronounced in Malalcahuello. This is followed by an abrupt increase of alien tree cover with a further increase in the native forest. This is not surprising, as patches of pine trees were observed within the native *Araucaria araucaria* forest, which is naturally open. In addition, the openness of the native forest constitutes an ideal environment for heliophytes like *Pinus* spp. to colonize, provided that there is enough propagule pressure (Ledgard, 2001; Taylor et al., 2016).

Lastly, the partial dependence plots for two variables that were highly relevant in the Villarrica model but not present in the Malalcahuello model are shown in Fig. 6. Here, the traditional and generally accepted hypothesis that distance to seed sources is of great importance for invasion (Altamirano et al., 2016; Arellano-Cataldo and Smith-Ramírez, 2016; Giorgis et al., 2016, 2011; Pauchard et al., 2016; Richardson et al., 2000) is confirmed. As mentioned earlier, the fact that distance to seed sources is not present in the Malalcahuello model indicates that, especially in the Malalcahuello site, the dominant

pattern is driven by plantations, masking the effect of invasions. Patch density of *Nothofagus* spp. native forest (Fig. 6) shows a very sharp increase in coniferous alien stand mean patch area at the higher patch density of *Nothofagus* spp. native forest. This abrupt increase did not occur in the plots for largest patch index and mean patch area for the same land cover class, suggesting that fragmentation, rather than habitat loss, could play a role in the presence of alien species. Higher patch densities involve higher fragmentation compared to an untouched forest, which agrees with the widespread idea that fragmentation and alien species are closely related (Arellano-Cataldo and Smith-Ramírez, 2016; León Cordero et al., 2016; Tella et al., 2016).

4. Discussion

The selected alien trees (*Eucalyptus* genus and *Pinaceae* family) were successfully mapped using Sentinel-2 medium resolution imagery within the native forest matrix, as required in the first objective of this study. The land cover map demonstrates that alien trees are located within the protected areas, especially in Villarrica. This agrees with existing concerns about alien species (particularly *Pinaceae*) reproducing naturally and spreading to *A. araucana* forests, which are located chiefly inside protected areas, potentially impeding the regeneration of *A. araucana* (Pauchard et al., 2014; Peña et al., 2008).

The methodological approach applied, based on the random forest classification of multi-seasonal and DEM composite images, was able to detect alien tree patches as small as 100 m² within a native vegetation matrix, addressing the first objective of mapping the locations of the alien patches. This demonstrates strong potential for large-scale forestry monitoring, particularly for management and monitoring of alien species, helping prevention of biological invasions. Large-scale management of biological invasions is a practical way of reducing them (Marvier et al., 2004). Conventional forest inventories are expensive, time-consuming activities typically only performed every few years at best and are always subject to budgetary and resource constraints. In contrast, satellite-derived land cover maps such as those presented in

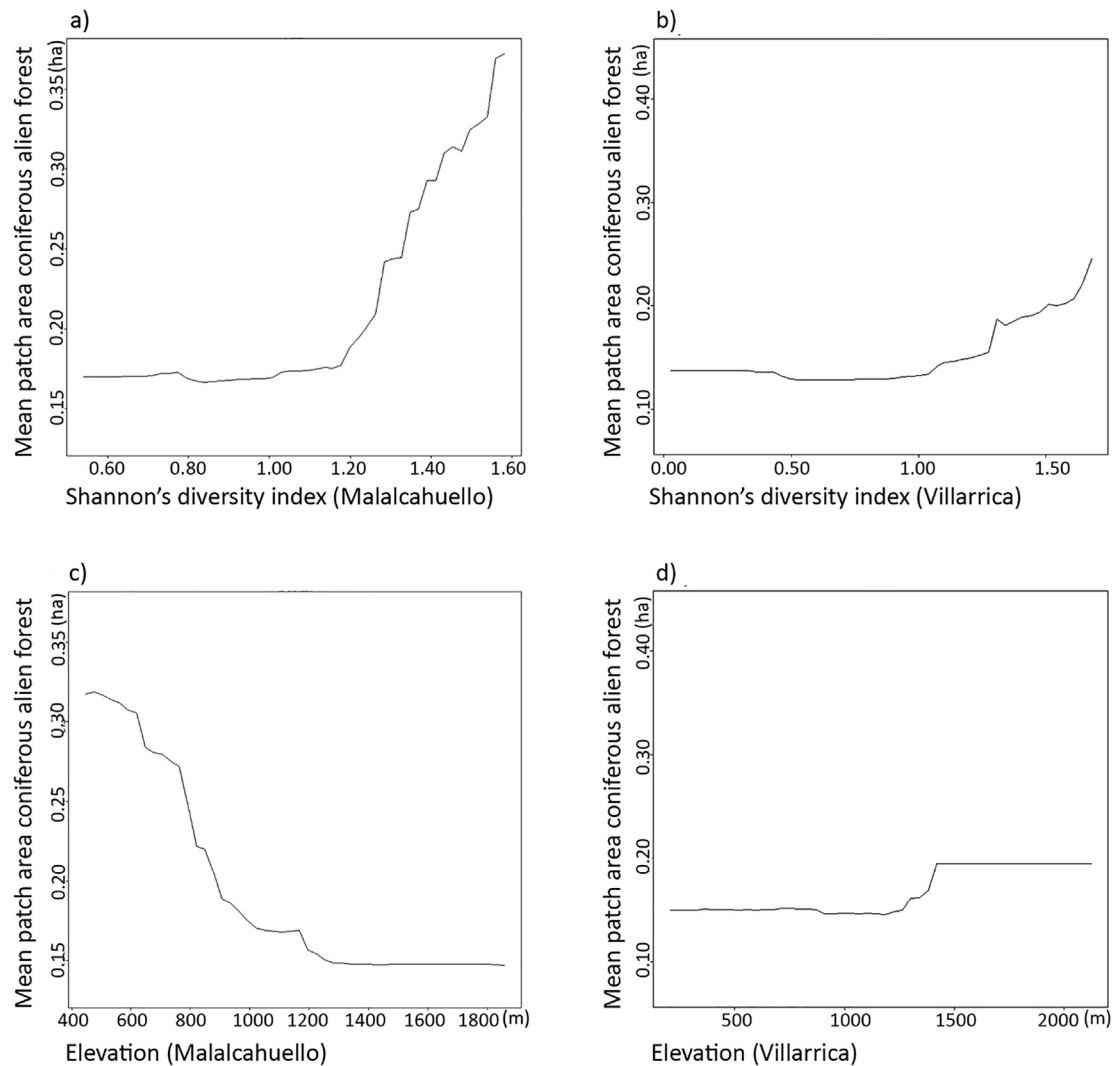


Fig. 4. Partial dependence plots of variables in the Malalcahuello and Villarrica models.

this study can be produced several times annually. They are far less resource intensive and Sentinel-2 data is freely available with regular temporal coverage, although availability can depend on cloud cover conditions. In extensive and topographically challenging areas such as the Andes and the Andean foothills, the capability to produce periodically land cover maps such as those shown here using cost-free resources such as Sentinel-2 imagery and the R software could better inform management operations and improve and optimise conservation activities by enabling specific targeting of high interest areas.

Addressing the second objective, which aimed to describe the spatial characteristics of forested areas, analysis using landscape metrics revealed an altitudinal pattern shared by Malalcahuello and Villarrica, as well as some contrasts between the two study areas. Valleys are shown to have higher patch densities with higher mean area and largest patch index of alien species, and lower mean area and largest patch index of native forests. The higher elevation areas exhibit lower patch densities with lower mean area of alien tree stands, and higher mean area of native forests. Consequently, in valleys, native forests are markedly more fragmented than in the Andes and are accompanied by an increased proportion of alien tree cover. This conforms with the generally accepted theory that alien species and landscape fragmentation are closely linked (Arellano-Cataldo and Smith-Ramírez, 2016; León Cordero et al., 2016; Tella et al., 2016). However, it is important to highlight that fragmentation inside protected areas is high for *A. araucana*, which suggests that a reevaluation of the conservation

strategy for this tree is needed.

Malalcahuello and Villarrica have contrasting land uses and differing histories of alien tree introduction. Villarrica has a higher population density (Library of National Congress Chile, 2015), intense tourism (Library of National Congress Chile, 2015), and a longer history of alien introduction than Malalcahuello. Introductions started in 1950 in Villarrica and 1969 in Malalcahuello (Kunstmann, 1965; Peña et al., 2008; von Buch, 1965). These differences could have led to the contrasts in their landscape structure. In general, Villarrica has a less clear landscape pattern when compared to Malalcahuello. Malalcahuello has higher mean patch area of alien trees, and they are predominantly clustered in accessible areas. Extreme values of most of the metrics occur in Villarrica, suggesting a more complicated mix of land covers. Even though Villarrica has comparatively lower mean patch area of alien tree stands, its native forests (when both *Nothofagus* spp. and *A. araucana* are considered together) are more fragmented, with higher patch density and less mean patch area.

Aside from Villarrica's native forest populations being more fragmented, further results indicate that its conservation status and prospects are worse than that of Malalcahuello. *A. araucana* forests reach their highest patch densities and mean areas inside protected areas in Malalcahuello, but this is not the case in Villarrica. Additionally, Villarrica has a high mean patch area and largest patch index of coniferous alien trees inside protected areas. This is concerning for the long-time conservation of the endemic *A. araucana* as Villarrica is a

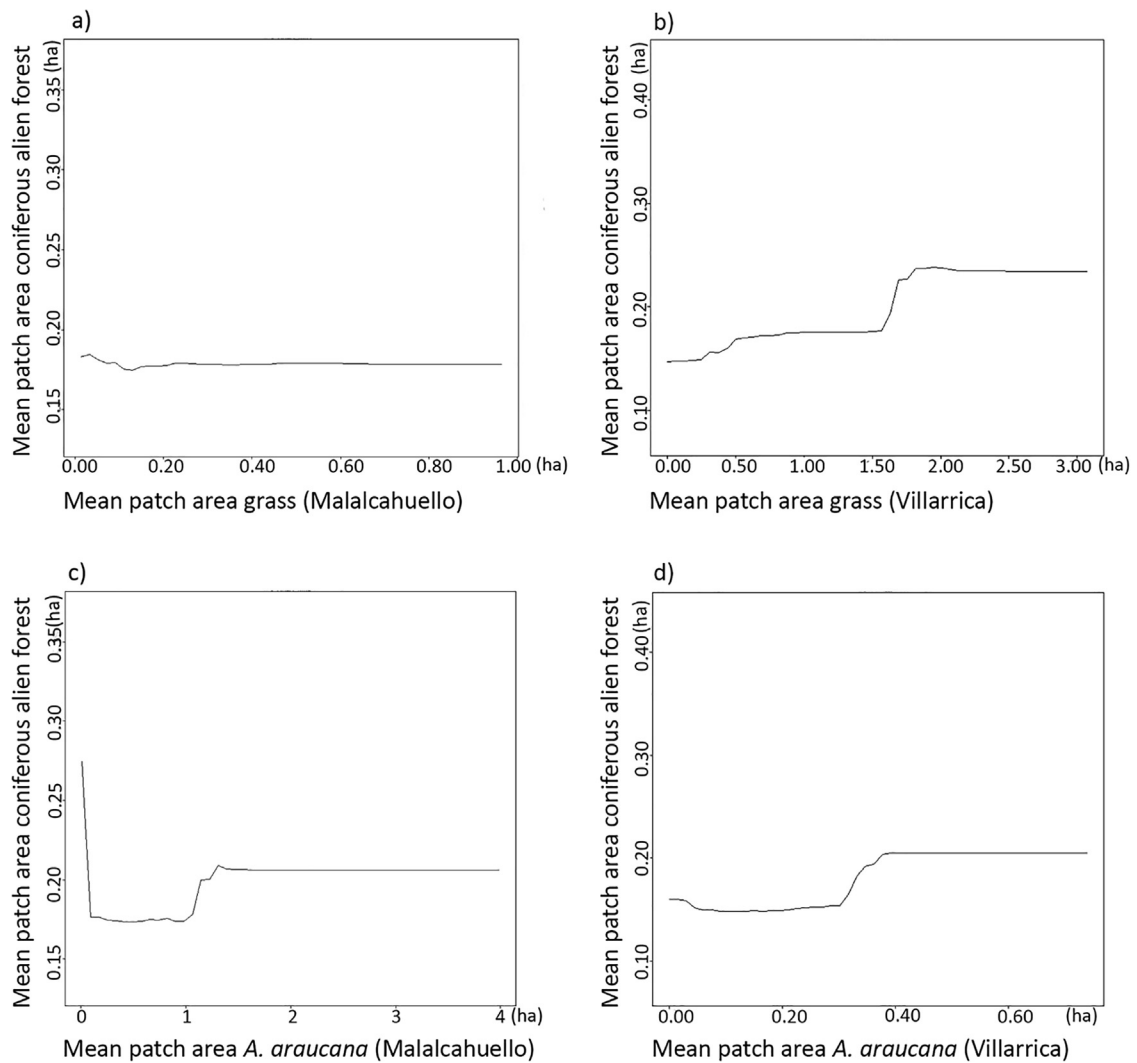


Fig. 5. Partial dependence plots of variables in the Malalcahuello and Villarrica models.

touristic area undergoing intensive construction, with certain areas interesting from a conservation perspective remaining unprotected. The location of protected areas in Villarrica needs to be revised to ensure that they achieve their conservation objectives. Regarding alien tree

cover, even though Malalcahuello has a larger area, it also has a lower patch density and a larger average patch size than Villarrica. This suggests that there is a larger number of small alien tree patches scattered in Villarrica's landscape, which could pose a risk of invasion

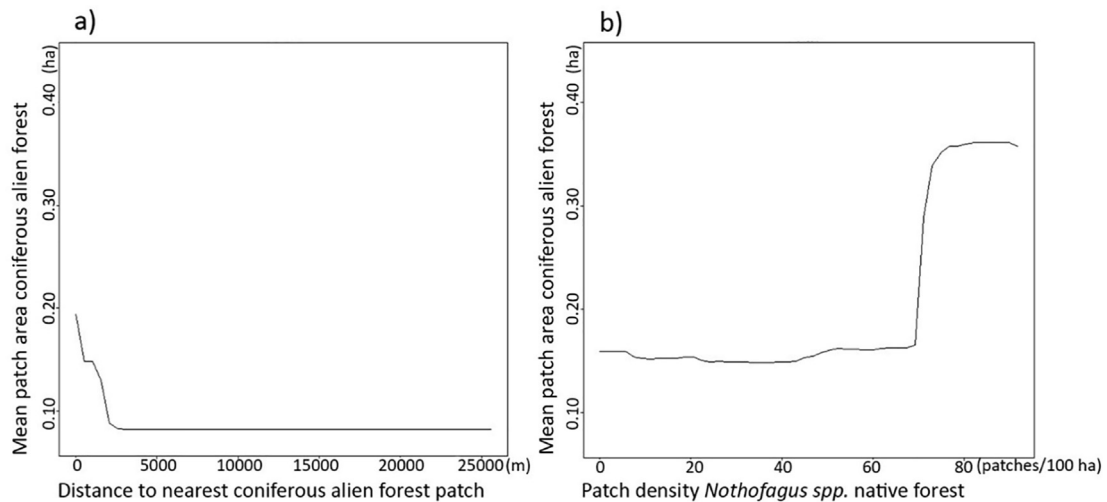


Fig. 6. Partial dependence plots of highly relevant variables in the Villarrica model.

regardless of the origin of these patches (plantation or invasion) due to the number of propagule foci.

The third objective was to model the influence of landscape structure on the *Pinaceae* extent (mean patch area of coniferous alien tree stands was the response variable). The random forest models for the two study sites consistently identified some of the same variables (Shannon's diversity index, elevation, grassland related metrics) as being of high importance, however they had different levels of complexity. The random forest model was more complex for Villarrica (retaining 27 explanatory variables as opposed to 13 for Malalcahuello). The land cover diversity quantified using Shannon's diversity index was identified as a relevant predictive variable for mean patch area of coniferous alien stands in both sites. An increase in land cover diversity was associated to an increase in alien land cover, especially in Malalcahuello, where this trend was more marked as a result of its more structured landscape. This is concerning, as a slightly higher Shannon's diversity index was found inside protected areas in both sites. It is unclear, however, if the increase in diversity is merely a result of the introduction of new species for forestry, or if a diversification of the land covers in fact triggered the expansion of alien trees. Future work should address this temporal aspect to enable the use of diversity metrics as predictors for future expansion of alien species.

Distance to seed sources (other alien tree patches) was only a relevant predictor for Villarrica, where alien tree cover decreases with distance from seed source, consistent with accepted invasion theories (Giorgis et al., 2016; Pauchard et al., 2016; Richardson et al., 2000). This, together with the spatial pattern information extracted, suggests that invasions might be occurring in Villarrica, while there are other drivers determining the location of the patches in Malalcahuello. These drivers could be socio-economic in nature, with forestry being one of the main economic activities in Malalcahuello. However, invasion at smaller scales could be occurring in this area, as there were some coniferous alien tree patches located in highly sensitive areas (*A. araucana* forests) inside the National Reserve of Malalcahuello. Also, it has been reported that alien species are entering protected areas using roads as corridors in locations very close to these study sites (Pauchard and Alaback, 2004). However, it is important to highlight that this methodology does not differentiate between alien plantation and alien invasion patches. The fact that distance to seed sources has not been identified as a relevant explanatory variable in the Malalcahuello model suggests that the plantation pattern is masking the invasion pattern. On the other hand, the higher complexity model for Villarrica suggests that the invasion pattern has not been completely concealed by the plantation pattern. In addition, Villarrica has distance to seed sources as the most important explanatory variable, and this agrees with invasion theory. That is why it is suggested that Villarrica might be in greater danger of invasion than Malalcahuello. Future work could address alien trees in targeted areas at a finer scale, potentially using higher spatial resolution satellite imagery, to try to distinguish alien tree invasion from plantation.

The contrasting trends in the relationship between altitude and coniferous alien tree cover show that topography plays a secondary role behind other variables, likely related to land use. In Malalcahuello, a rural area with abundant productive alien plantations, these tend to be located in lower, easily accessible areas. In Villarrica however, where other industries like tourism are more relevant, alien tree patches are pushed to higher altitudes, where most of the native forests are located, being a potential source for invasion.

Finally, it is important to highlight that mean patch area of coniferous alien tree stands increased with mean patch area of *A. araucana* forests in both models. This means that even though *A. araucana* is a Natural Monument in Chile and is protected from felling, there is presence of *Pinaceae* in its surrounding environment. This *Pinaceae* presence constitutes an invasion risk, as *A. araucana* forests are especially vulnerable to *Pinaceae* invasions due to their open forest structure. This, together with the fact that there are abandoned pine plantations inside

the National Reserve of Malalcahuello, and that *Pinaceae* are located in higher areas with part of the *A. araucana* trees not inside protected areas in Villarrica, calls for a reevaluation of the conservation strategies that are being followed for this endangered (Premoli et al., 2013) and endemic tree. In its current situation, *A. araucana* could be out-competed by trees from the *Pinaceae* family, which are mostly heliophytes and strong competitors in open environments (Ledgard, 2001; Taylor et al., 2016), reducing it to a relict state in due course.

5. Conclusion

This study has presented a cost-effective and relatively quick way of surveying alien tree stands within a matrix of native forests. It has been demonstrated that the methodology applied using freely available medium resolution Sentinel-2 optical satellite data is useful for the monitoring of alien trees within the Chilean Valdivian temperate forest, whose extent and topography limit traditional, ground based methods. Very high levels of mapping detail and accuracy have been achieved, detecting small patches (100 m²) of alien *Pinaceae* and *Eucalyptus* spp. trees within a matrix of *Nothofagus* spp. and *A. araucana* native forests. This enables the monitoring and management of alien trees at the landscape scale in an effective and inexpensive manner (Marvier et al., 2004).

Within each study site, native forest fragmentation is accompanied by an increase in alien tree cover following an altitudinal gradient. The presence of alien trees acting as a source of propagules and the fragmentation of the landscape could lead to biological invasions (Marvier et al., 2004) in the area. However, when comparing the two study sites, Malalcahuello, which has a comparatively higher area of alien trees, has less fragmented native forests than Villarrica. Very large management scales, i.e. between sites, imply a difference in socioeconomic and even environmental variables. These differences have a synergistic impact in the specific landscape configuration of each site. For this reason, the management of biological invasions needs to be conducted at the appropriate scale and using meaningful metrics for that scale, staying away from 'one size fits all' conservation policies.

The results of this study agree on the concern about the long-term continuity of the *A. araucana* forests, as the modelled results show increasing *Pinaceae* cover with increasing *A. araucana* native forest cover, and high land cover diversity and fragmentation of *A. araucana* within protected areas. Consequently, even though *A. araucana* is protected from felling by law, it could be outcompeted by *Pinaceae*. The protection status of *A. araucana* is particularly alarming especially in Villarrica, as a large part of the population is outside protected areas and there is a large alien tree cover within protected areas and at higher altitudes, the natural habitat of *A. araucana*. A reconsideration of the location of protected areas and the restriction of productive forest plantations to lower altitudes, far from the *A. araucana* native forests, are recommended conservation measures to ensure the long-term continuity of the *A. araucana* populations.

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CRedit authorship contribution statement

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

Declaration of Competing Interest

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

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Appendices. Supplementary material

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.foreco.2020.118353>.

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