

**Monitoring invasive tree species using satellite remote sensing in the temperate forests of south-central Chile**

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## Abstract

Chile's temperate forest is a global biodiversity hotspot due to its levels of endemism and rate of habitat loss. Deforestation and biological invasions are two major threats faced by this ecosystem. Both are produced by the expansion of productive forestry, which fragments the native forest and introduces alien species (from the Pinaceae family and *Eucalyptus* genus). Despite these threats, broad scale ground-based forest and alien species monitoring is not feasible due to logistical and budgetary limitations. Surveying and mapping alien species are key steps to understand biological invasions. This project uses remote sensing to understand biological invasions by alien trees from productive forestry in Chile's temperate forest at different spatial scales. Sentinel-2 is used to assess alien tree occupancy and model its relationship with fragmentation and landscape configuration at a broad scale. A temporal assessment using Landsat historical archive imagery is used to investigate the dynamics of alien tree spread over the last 40 years. A local scale assessment using very high resolution WorldView-3 imagery and object-based image analysis is used to investigate early invasion and its relationship with land cover configuration. The results demonstrate remote sensing methods are useful for alien tree monitoring at different scales and hold potential for management. The expansion of alien trees at a broad scale is dominated by socioeconomic factors such as cost effectiveness and government subsidies. Alien Pinaceae tend to invade the naturally open *Araucaria araucana* forests, threatening the long term regeneration of this endemic and culturally relevant tree. Early Pinaceae invasion is linked to fragmentation, distance to seed sources, topography and human intervention. Incidentally, this study found that native forest degradation is occurring, increasing its vulnerability to invasion in the future. Conservation and forest management need to adopt sustainable, evidence-based practices to achieve compatibility between productive forestry and native forest conservation.

Key words: biological invasions, remote sensing, temperate forests, Chile, Pinaceae, *Eucalyptus*, land cover, Sentinel-2, Landsat, WorldView-3

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## List of abbreviations

%IncMSE	Percentage increase in Mean Squared Error
AI	Aggregation Index
ANF	Native <i>Araucaria araucana</i> forest
AP	Alien plantation
AREA_MN	Mean Patch Area
ASTER	Advanced Spaceborne Thermal Emission and Reflection Radiometer
BP	Broadleaved Plantations and invasion
CLUMPY	Clumpiness Index
CONAF	Chilean National Forest Corporation
CP	Coniferous Plantations and invasion
DEM	Digital Elevation Model
DGA	Chilean General Directorate of Waters
EROS	Earth Resources Observation and Science
ESA	European Space Agency
FAO	Food and Agriculture Organization
FSC	Forest Stewardship Council
GEE	Google Earth Engine
IUCN	International Union for Conservation of Nature
LPI	Largest Patch Index
MOP	Chilean Ministry of Public Works
MSE	Mean Squared Error
NASA	National Aeronautics and Space Administration
NDVI	Normalized Difference Vegetation Index
NIR	Near Infrared
NNF	Native <i>Nothofagus</i> spp. forests
NWS	Native woody species
OBIA	Object-Based Image Analysis
OLI	Operational Land Imager
OOB	Out of Bag

PA	Producer's accuracy
PAFRAC	Perimeter Area Fractal Dimension
PD	Patch Density
PLAND	Percentage of Landscape
RPC	Rational Polynomial Coefficient
SHDI	Shannon's Diversity Index
SNIC	Simple Non-Iterative Clustering
Spp.	Species
SRTM	Shuttle Radar Topography Mission
SWIR	Short Wave Infrared
TIRS	Thermal Infrared Sensor
TM	Thematic Mapper
UA	User's accuracy
UAV	Unmanned Aerial Vehicle
UNESCO	United Nations Educational, Scientific and Cultural Organization
USGS	United States Geological Survey
VHR	Very High Resolution

## List of species

<i>Aextoxicon punctatum</i> Ruiz & Pav.	<i>Olivillo</i> (Chile)
<i>Araucaria araucana</i> (Molina) K.Koch	Monkey puzzle tree
<i>Azara alpina</i> Poepp. & Endl.	<i>Lilén de la cordillera</i> or <i>aromo</i> (Chile)
<i>Azara lanceolata</i> Hook.f.	<i>Perhue</i> or <i>aromo</i> (Chile)
<i>Aristotelia chilensis</i> (Molina) Stuntz.	Chilean wineberry
<i>Baccharis patagonica</i> Hook. & Arn.	<i>Chilca</i> or <i>vaultro</i> (Chile)
<i>Berberis trigona</i> Kunze ex Poepp. & Endl.	Barberry
<i>Caldcluvia paniculata</i> D.Don	<i>Tiaca</i> (Chile)
<i>Chusquea culeou</i> E.Desv.	Chilean bamboo
<i>Chusquea quila</i> Kunth	<i>Quila</i> (Chile)
<i>Desfontainia spinosa</i> Ruiz & Pav.	<i>Taique</i> (Chile)
<i>Drimys winteri</i> J.R.Forst. & G.Forst.	Winter's bark
<i>Eucalyptus globulus</i> Labill.	Blue gum
<i>Eucalyptus nitens</i> (H.Deane & Maiden) Maiden	Shining gum
<i>Eucryphia cordifolia</i> Cav.	<i>Ulmo</i> (Chile)
<i>Holcus lanatus</i> L.	Velvet grass
<i>Laurelia sempervirens</i> (Ruiz & Pav.) Tul.	Peruvian nutmeg or Chilean laurel
<i>Laureliopsis philippiana</i> (Looser) Schodde	<i>Tepa</i> (Chile)
<i>Luma apiculata</i> (DC.) Burret	Chilean myrtle
<i>Maytenus disticha</i> (Hook.f.) Urb.	<i>Racoma</i> (Chile)
<i>Maytenus magellanica</i> (Lam.) Hook.f.	Hard log mayten
<i>Myrceugenia chrysoarpa</i> (O.Berg) Kausel	<i>Luma blanca</i> (Chile)
<i>Nothofagus alessandrii</i> Espinosa	Ruil
<i>Nothofagus alpina</i> (Poepp. & Endl.) Oerst.	Rauli beech
<i>Nothofagus antarctica</i> (G.Forst.) Oerst.	Antarctic beech
<i>Nothofagus betuloides</i> (Mirb.) Oerst.	Magellan's beech
<i>Nothofagus dombeyi</i> (Mirb.) Oerst.	Dombey's beech

<i>Nothofagus obliqua</i> (Mirb.) Oerst.	Patagonian oak
<i>Nothofagus pumilio</i> (Poepp & Endl) Krasser	Lenga beech
<i>Ovidia pillopollo</i> (Gay) Meisn.	<i>Pillopollo</i> (Chile)
<i>Gaultheria myrtilloides</i> Cham. & Schltld.	Wintergreen
<i>Persea lingue</i> (Ruiz & Pav.) Nees	<i>Lingue</i> (Chile)
<i>Pinus contorta</i> Douglas ex Loudon	Lodgepole pine
<i>Pinus ponderosa</i> Douglas ex C.Lawson	Ponderosa pine
<i>Pinus radiata</i> D.Don	Monterey pine
<i>Pinus sylvestris</i> L.	Scots pine
<i>Poa nemoralis</i> L.	Wood bluegrass
<i>Podocarpus nubigenus</i> Lindl.	Podocarp
<i>Pseudotsuga menziesii</i> (Mirb.) Franco	Douglas fir
<i>Raukua laetevirens</i> (Gay) Frodin.	Devil's elder
<i>Rosa moschata</i> Herrm.	Musk rose
<i>Rubus constrictus</i> Lefèvre & P.J.Müll.	Blackberry
<i>Rubus ulmifolius</i> Schott	Elmleaf blackberry
<i>Sphagnum</i> L.	Peat mosses
<i>Ulex europaeus</i> L.	Gorse
<i>Weinmannia trichosperma</i> Cav.	<i>Tineo</i> (Chile)

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# Chapter 1 Introduction

## 1.1. Identifying the problem

The Chilean Valdivian temperate forest is a global biodiversity hotspot, containing a specific species assemblage that makes it highly ecologically valuable. It is, however, a fragile and vulnerable ecosystem, and has recently experienced a rapid rate of deforestation and habitat loss (Myers et al., 2000, Salas et al., 2016, Miranda et al., 2017).

A principal cause of this habitat loss in Chile has been the boom of the forestry industry which has resulted in the large-scale expansion of non-native (alien) plantations (Miranda et al., 2015), with many of these alien trees being invasive species (Pauchard and Alaback, 2004). Compounding this, continuous human disturbance has further increased the vulnerability of the native forest to biological invasions (Giorgis et al., 2016, León Cordero et al., 2016, Pauchard et al., 2016). Despite this situation, invasive species have been traditionally understudied in South America (Speziale et al., 2012) resulting in a significant gap in current knowledge as to how invasive alien tree species have impacted, or will in the future impact, the fragile native forests.

To mitigate against biological invasions of alien forestry species, appropriate monitoring informing prompt management is required (Pauchard et al., 2016, Nuñez et al., 2017). However, such tasks are difficult in Chile using conventional ground-based surveys due to the country's large geographical extent, abrupt and challenging topography, and the land cover change dynamics occurring in some areas. This research uses satellite remote sensing to overcome these limitations, mapping alien tree species at multiple spatial scales and examining their relationship with the spatial configuration of the landscape. The research outputs generated are extremely valuable for current, historical and future modelling to inform forest management and conservation planning.

## 1.2. Biological invasions

Biological invasions are among the main drivers of global change, although the type and magnitude of their impacts on native biological communities and environments remain highly uncertain (Tella et al., 2016). They are one of the most important

threats to biodiversity, together with habitat loss and degradation, overexploitation and climate change (Pettorelli et al., 2014). Specifically, plant invasions can have a wide range of environmental impacts, including: impacts on native vegetation caused by allelopathy, impacts on fauna due to toxicity or modifications in food availability, impacts through competition, disease and parasite transmission to native plants, hybridization with local vegetation, and general changes in ecosystem fluxes, nutrient pools, disturbance regimes, or physical structure (Nentwig et al., 2016).

Invasion ecology studies revolve around three major issues: invasiveness of species, invasibility of habitats, and impacts produced by invasive species (Alpert, Bone and Holzapfel, 2000). The terminology used in these studies is often confusing, obscuring the conclusions and hampering the applications of many investigations. For this reason, a set of crucial invasion ecology terms are introduced in the following paragraphs.

Regarding the level of invasion by a species, this study follows the terminology suggested by Richardson et al. (2000) in their review: an alien plant is a species whose presence in a certain area is a result of human introduction. Some of them are introduced intentionally and others by accident (Mack et al., 2000). Alien species are also called exotic, introduced, non-indigenous or non-native species. If an alien plant is capable of reproducing in its new settlement, but it is not able to establish self-sustaining populations, it becomes a casual alien plant. Casual alien plant populations depend on periodic introductions. A small portion of these casual alien plants survive in the hazards of the new habitat and become naturalised plants (Mack et al., 2000), being able to sustain populations over many generations without any help. Invasive plants are naturalised plants that propagate abundantly and reach long distances from the point of introduction. They could potentially spread over an extensive territory and persist in the landscape altering its ecosystem properties (Gordon, 1998, Bolch et al., 2020).

Invasion is thus a process, and the boundaries between naturalised and invasive plants are not fixed. They often depend on the biology of the alien species and the environmental characteristics of the place where they are introduced. Plants reproducing by seeds or propagules spreading over more than 100 metres in less than 50 years, and taxa with vegetative propagation expanding more than 6 metres every



3 years would generally be considered invasive (Richardson et al., 2000). In addition, a non-native plant could be non-invasive at the moment of introduction, but become invasive as a result of a period of unconventional precipitation or temperature, as a result of a new mutualist arriving, or through evolutionary processes (Alpert, Bone and Holzapfel, 2000).

The degree of invasiveness of species correlates with certain physiological traits and growth rates, as shown in the meta-analysis by Van Kleunen et al. (2010). However, meta-analyses have some constraints such as publishing biases favouring studies with significant results over those that found no difference between the traits of invasive and non-invasive species (Gundale et al., 2014). Single traits are not very useful to explain plant invasions (Küster et al., 2008). In consequence, the invasion process is largely dependent on the interaction between the introduced species and the new ecosystem (Alpert, Bone and Holzapfel, 2000), such as ecological interactions of facilitation and competition (Becerra and Bustamante, 2011, Cavieres, 2021), or on particular combinations of traits (Küster et al., 2008). The invasiveness of a species in an ecosystem tends to be very case-dependent due to the complexity of the ecological interactions involved.

Even though research about species invasiveness is still inconclusive (Nunez-Mir et al., 2019), a few key traits have been identified consistently in the literature or in large scale studies. A study in the United States containing all woody plants classified as invasive (63 species) and almost all the non-invasive (794 out of the 824 non-invasive alien species) aimed to produce a model of potentially highly invasive alien woody species. This large scale study identified the ability to reproduce vegetatively, long distance seed dispersal strategies (by water, mammals or birds), and growth form (lianas tend to be invasive) as key traits of alien invasive species (Nunez-Mir et al., 2019).

The traditional and widespread diversity-invasibility hypothesis states that ecosystems are less likely to be invaded if they have high community species richness (Mack et al., 2000). However, this has been proven not universal, as native species richness usually has a positive correlation with alien species richness (Fuentes et al., 2013, Smith and Côté, 2019). Biogeography plays an important role in determining invasibility. Island habitats are generally more vulnerable to invasion, as demonstrated by review studies of large alien plant datasets and surveys (Gimeno,

Vilà and Hulme, 2006, Pyšek and Richardson, 2006). Certain kinds of disturbances, such as land use change from natural to managed types (Chytrý et al., 2008, 2009, Zhou et al., 2019), extreme climatic events due to climate change (Diez et al., 2012) and wildfire upsurges (Mack et al., 2000), favour the establishment of invasive species (Early et al., 2016). In a recent study, Giorgis et al. (2016) found that habitat invasibility depends to a great extent on growth form of both the habitat and the non-native species. This agrees with the study by Nunes et al. (2020), which identified growth form as a key trait of invasive species. Landscapes such as shrublands and outcrops were less prone to woody plant invasion, and grasslands and native forests were less susceptible to invasive grasses. None of the habitat types showed any resistance to forbs invasion (Giorgis et al., 2016). Hence, resistance to invasions seems to be more associated with the structural complexity of the vegetation than with the community biodiversity (Mack et al., 2000). In their extensive review, Alpert et al. (2000) enumerate a set of factors that would likely make habitats less invasible. Those include long history of human disturbance, strong competition, high biodiversity, strong indirect species interactions, weak competition, lack of mutualists, existence of herbivores, poor dispersal agents, lack of fragmentation, absence of disturbance events such as fire, and environmental stressors. Specifically, fragmentation, which is linked to ecosystem disturbance, has been found to favour biological invasions (Marvier, Kareiva and Neubert, 2004, León Cordero et al., 2016). León Cordero et al. (2016) found that the invasive shrub *Ulex europaeus* L. tended to encroach into grasslands subject to intense cattle grazing, while undisturbed forests were free of invasion in south Brazil. According to Chytrý et al. (2009), in general, sclerophyllous vegetation, peatbogs, moors and heathlands are the less invaded landscapes in Europe. Grasslands and woodlands also have low levels of invasion. The temperate forests of Chile have a mixed set of these characteristics, some of them facilitating the invasion process and others hampering it.

Ultimately, the inherent complexity of invasion ecology limits the ability to identify a fixed set of key traits that determine habitat invasibility. A 20-year experiment with grasslands in Minnesota (United States) emphasises this complexity and the context dependency of plant invasions. This study found that the traits of the alien species, the recipient communities, and the interactions between the two heavily influenced invasion success. For example, species taller than the recipient community were the

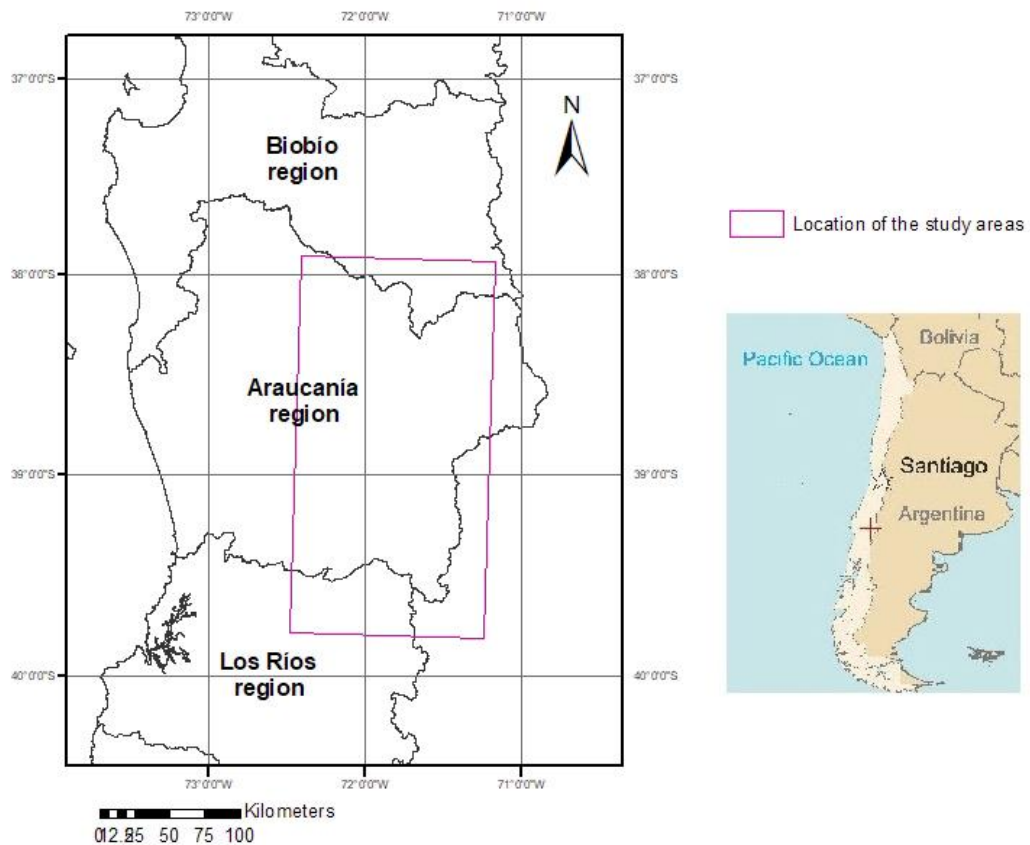
most successful invaders in the plots where alien species were planted. However, in the long run, the opposite trend was observed in the nearby unseeded plots, highlighting the importance of the environment in the success of invaders (Catford et al., 2019). A recent meta-analysis highlighted the importance of facilitative interactions between native and alien species, especially in stressful environments such as the Chilean Andes or the Patagonian steppe. This facilitation can be direct through soil stabilisation or temperature buffering, or indirect due to herbivore preferences, pollination mutualisms, physical protection from herbivores, or seed dispersal chains (Cavieres, 2021). These facilitative interactions are less prominent in more benign environments. Thus, a high native community species richness could facilitate invasion in stressful environments (as there are more facilitation opportunities) and hamper it in benign environments (due to biotic resistance) (Stotz, Pec and Cahill, 2016, Cavieres, 2021). The diversity-invasibility hypothesis depends on the environmental conditions. Another meta-analysis found that this hypothesis is also dependent on spatial scale. While fine scale studies agreed with the theory, with native biodiversity and invasion being negatively correlated, broad scale studies showed the opposite trend (Smith and Côté, 2019). Given the uncertainty regarding habitat invasibility, more research is needed to understand the interaction among native and alien species in different environmental stress conditions and at different spatial scales (Stotz, Pec and Cahill, 2016, Smith and Côté, 2019, Cavieres, 2021).

### 1.3. The Valdivian temperate forests of Chile

Temperate forests are the rarest among the world's woodland biomes, representing only 16% of the total area, and being affected by very large annual forest loss rates (Echeverría et al., 2006, Hansen, Stehman and Potapov, 2010). Temperate forests have high agricultural and biomass productivity, host high biodiversity, and are densely populated by humans. Consequently, they are greatly impacted by land use change (Ellis, 2011) and threatened by biological invasions (Essl, Milasowszky and Dirnböck, 2011, Richardson and Rejmánek, 2011).

The Valdivian temperate forest ecoregion spreads along a slim strip of land between the Andean mountain range and the Pacific Ocean, from 35° to 48° south latitude (Smith, 2002). Elevations range between 1000 and 3000 m, maximum temperatures range between 13°C and 21°C, minima between 4°C and 7°C; and precipitation is

seasonal, ranging from 1000 mm annual rainfall to more than 6000 mm (Smith, 2002). These intense variations are a consequence of the large latitudinal range. The Valdivian temperate forest of southern Chile, together with the rainiest part of the Chilean matorral ecoregion (located in the central area of the country and with a Mediterranean climate), constitute a biogeographical island, isolated by deserts, mountain ranges and oceans (Altamirano and Lara, 2010). The area is marked by two parallel mountain ranges, the Coastal Range to the west and the Andes to the east, separated by the Central Valley. The research chapters of this project focus on an area in the northern part of the Valdivian temperate forest ecoregion, as can be observed in figure 1.1.



**Figure 1.1** General location of the different study areas used in this project. Map created using data from the Library of National Congress of Chile (2021).

This ecoregion can be subdivided into five types of forests from north to south (Smith, 2002): 1) deciduous forests, which are influenced by a Mediterranean climate and dominated by deciduous *Nothofagus* spp.; 2) Valdivian laurel forests, which comprise various tree species such as *Laureliopsis philippiana* (Looser) Schodde, *Aextoxicon punctatum* Ruiz & Pav., *Eucryphia cordifolia* Cav., *Caldcluvia paniculata*

D. Don, and *Weinmannia trichosperma* Cav.; 3) northern Patagonian forests, populated mainly by evergreen trees such as *Nothofagus dombeyi* (Mirb.) Oerst., *Podocarpus nubigenus* Lindl. and *Drimys winteri* J.R.Forst. & G.Forst.; 4) Patagonian Andean forests, which include deciduous *Nothofagus* spp. mixed with *Araucaria araucana* (Molina) K.Koch, which is endemic and endangered (Premoli, Quiroga and Gardner, 2013), and high mountain shrubs; and 5) evergreen *Nothofagus betuloides* (Mirb.) Oerst. forests with *Sphagnum* spp. bogs. Chile is one of the few temperate countries in the southern hemisphere along with Australia, New Zealand and South Africa (Pauchard et al., 2004); but more than half of the temperate forests in the southern hemisphere occur in Chile (Donoso, cited in Miranda et al., 2015: 21). Less than 11% of these are protected (Miranda et al., 2017). Central and southern Chile is considered a global biodiversity hotspot because it has a significant number of endemisms, although it is losing habitats and native forest at a rapid rate (Myers et al., 2000, Salas et al., 2016, Miranda et al., 2017). Only 30% of its original native vegetation cover remains (Myers et al., 2000). An annual rate of native forest loss of 0.3% occurred in south-central Chile between the years 1986 and 2011 (Heilmayr et al., 2016), but it reached 4.5% in areas of the Coastal Range between 1975 and 2000 (Echeverría et al., 2006). The majority of the remnant pristine native forest is located in remote zones of the Andean range and the southernmost parts of the country (Miranda et al., 2015).

The Chilean Valdivian temperate forest is critically threatened by deforestation and alien species introduction for forestry (Echeverría et al., 2006, Heilmayr et al., 2016, Locher-Krause et al., 2017). Deforestation has been occurring since 1973; initially woodland was replaced by agriculture, later by shrublands, and in recent years plantation forests have expanded significantly (Miranda et al., 2015). This has led to landscape homogenisation and native forest fragmentation, especially in the most accessible parts of the Coastal Range and the Central Valley, compromising species richness and ecosystem services (such as carbon sequestration) in the area (Echeverría et al., 2006, Miranda et al., 2015, Locher-Krause et al., 2017, Heilmayr, Echeverría and Lambin, 2020). In addition, some of the native woodland remnants are highly degraded due to selective logging for timber and firewood for the local populations (Echeverría et al., 2006, Salas et al., 2016). This phenomenon is reflected in the amount of early successional forest and shrubland in certain areas (Carmona and Nahuelhual, 2012, Miranda et al., 2017). Selective logging changes the structure

and composition of the remnant native forest patches, hampering their capacity to sustain their original biological community and ecological processes (Echeverría et al., 2007). Globally, temperate forests are more affected by invasion than other forest types (Pyšek and Richardson, 2006). In addition, as a result of its isolated evolution, the Valdivian temperate forest is highly vulnerable to alien species introduction and biological invasions (Alpert, Bone and Holzapfel, 2000, Gimeno, Vilà and Hulme, 2006, Pyšek and Richardson, 2006). Also due to its isolation, the Valdivian temperate forest (like the rest of the South American temperate forests) has a series of vacant ecological niches (Donoso, González and Lara, 2014: 674). Species that are native and/or abundant in many ecosystems of the world but not to South American forests include: coniferous trees from the Pinaceae family, species that regrow quickly after a disturbance such as a wildfire, and nitrogen fixers (Donoso, González and Lara, 2014: 674).

This research project focuses on two study areas in the Araucanía region of Chile. The areas are within the Andes and include a mix of *Nothofagus* spp. forests and high mountain Patagonian Andean forests. Within these broad categories of forest, different plant community compositions can be found depending on the mesoclimatic conditions and the degree of human disturbance. In the following paragraphs, these plant community compositions are briefly summarised using the categories described by Luebert and Plissock (2004) in their review for the Valdivian ecoregion. These descriptions concentrate on woody species (trees and shrubs), as these are the focus of this research project.

At lower elevations, where the Andes meet the Central Valley, human influence is intense. Patches of highly degraded deciduous *Nothofagus obliqua* (Mirb.) Oerst. forest can be found within the matrix of farms and urban settlements. In these forests, *N. obliqua* is mixed with *Persea lingue* (Ruiz & Pav.) Nees in drier areas, and with *Laurelia sempervirens* (Ruiz & Pav.) Tul. and sometimes also *N. dombeyi* in more humid areas. Selective thinning has degraded *N. obliqua* forests promoting shrub encroachment. These shrubs vary depending on the degree of disturbance. Less disturbed areas are colonised by *Chusquea quila* Kunth, while the more disturbed areas (for example by excessive grazing) are colonised by *Aristotelia chilensis* (Molina) Stuntz., *Rubus constrictus* Lefèvre & P.J.Müll or the alien *Ulex europaeus* (Luebert and Plissock, 2004).

At medium elevations (740-1220 m), the forest is dominated by deciduous *Nothofagus alpina* (Poepp. & Endl.) Oerst. and evergreen *N. dombeyi*. The trees are of different ages, creating a highly dynamic and multi-layered tree canopy. The fact that *N. alpina* is deciduous and the uneven tree ages allow light to penetrate through the canopy, enabling a wide range of native shrubs to thrive (*Raukaua laetevirens* (Gay) Frodin., *Azara lanceolata* Hook.f., *Chusquea culeou* E.Desv., *Maytenus magellanica* (Lam.) Hook.f., *Myrceugenia chrysocarpa* (O.Berg) Kausel, *Berberis trigona* Kunze ex Poepp. & Endl. and *L. philippiana*). Large disturbances like forest fires or volcanic activity lead to an even aged regenerated forest that, over time and due to small scale disturbances, increases its complexity. Intense human intervention in these forests lead to shrub encroachment and presence of alien species like *Rubus ulmifolius* Schott (Luebert and Plissock, 2004). Slightly higher elevations (800-1300 m) are dominated by evergreen *N. dombeyi* and *A. araucana*. The shrub layer includes *C. culeou*, *R. laetevirens*, *Desfontainia spinosa* Ruiz & Pav., and *D. winteri*. When fire or volcanic disturbances occur, *N. dombeyi* regenerates faster than *A. araucana*. Over time, small disturbances create gaps in the *N. dombeyi* canopy, allowing *A. araucana* trees to access the top layer of the forest (Luebert and Plissock, 2004).

The highest elevations with tree presence (1300-1600 m) host a mixed deciduous *Nothofagus pumilio* (Poepp & Endl) Krasser and evergreen *A. araucana* forest with a shrub layer comprising *C. culeou*, *Maytenus disticha* (Hook.f.) Urb., *M. chrysocarpa*, *Azara alpina* Poepp. & Endl., and *Gaultheria myrtilloides* Cham. & Schlttdl. At lower elevations within this level, some *N. dombeyi* can coexist with *N. pumilio* and *A. araucana*. Pure *A. araucana* forests also grow at 1300-1600 m elevation. The trees are scattered forming an open canopy with an understory with native shrubs such as *Baccharis patagonica* Hook. & Arn. and native grasses typical from the Andean steppe. *N. pumilio* and *Nothofagus antarctica* (G.Forst.) Oerst. can sometimes be found, but in extremely low densities. When volcanic events or extreme fires occur, the *Nothofagus* trees, shrubs and grasses die, while *A. araucana* typically survive due to its thick, protective bark. After fire events, *Nothofagus* trees resprout in even aged cohorts (Luebert and Plissock, 2004, González, Veblen and Sibold, 2005).

Apart from anthropogenic land use change (Ellis, 2011, Miranda et al., 2015), fire is the most important disturbance regime influencing the *A. araucana* and *Nothofagus* spp. forests. A fire regime comprising frequent low intensity surface and infrequent,

severe crown fires has determined the structure of these forests. Previous research about forest fire history in one of the study areas used in this project found that extensive and severe forest fires are relatively infrequent, and mainly circumscribed to the *A. araucana* with *Nothofagus* spp. mixed forests. These large fires were linked to drought years, which are increasingly common since the 1970s (González, Veblen and Sibold, 2005). In a climatic context of increasing temperatures and frequency of extended drought periods, the fire regime in the Valdivian temperate forest region is expected to change dramatically. A recent study in central and south-central Chile found that fire season was getting longer, fires were more numerous, burned area was rising, and fire duration was increasing (González et al., 2018). A change in fire regime is very concerning, as the native *Nothofagus* spp. species are not well adapted to recurrent fires (Kitzberger et al., 2016). A study in the Araucanía region, where the study areas of this project are located, found that, in the short term, an intense fire event reduced plant diversity by half. The same study found that low intensity fire benefitted alien species (Urrutia-Estrada, Fuentes-Ramírez and Hauenstein, 2018). The projected increase in both severe and low intensity fire frequency as a result of climate change could trigger a long-term shift in plant community composition favouring alien species.

Alien productive plantations generally increase fire hazard due to their homogeneity, high density, and fuel load (Veblen et al., 2008). This, coupled with climate change, could contribute to a change in fire regime that affected the nearby native forests. A recent study on the effect of drought on the fire regime found that, even though burned area increased for all forest types, alien plantations were the most affected. In addition, the Araucanía region has a great extent of alien plantations and was among the most affected by fire (González et al., 2018). Also, fire and invasions could act synergistically. Fire could further benefit alien species by creating suitable environmental conditions for their regeneration, spread, and invasion (Zouhar et al., 2008, Taylor et al., 2017) or by opening up suitable areas for livestock grazing, which is a vector of alien species dispersal (Urrutia-Estrada, Fuentes-Ramírez and Hauenstein, 2018).

#### 1.4. The Chilean forestry industry and its impact

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). In 2020, forestry exports were US\$ 4.95 million, 6.9% of the total exports (Álvarez, Soto and Bañados, 2021, Central Bank of Chile, 2021). The sector is predominantly based on intensive silviculture of alien Pinaceae (mainly *Pinus radiata* D.Don) and *Eucalyptus* spp. (mainly *Eucalyptus globulus* Labill. and *Eucalyptus nitens* (H.Deane & Maiden) Maiden) (Salas et al., 2016). Between the years 1976 and 2012, the forestry development law partially subsidized new productive plantations (Chilean Ministry of Agriculture, 1998). The forest subsidies drove alien plantation expansion at the expense of native forest between 1986 and 2011 (Heilmayr, Echeverría and Lambin, 2020).

Although native forest species could also be used for productive forestry and provide high quality wood, they are not widely used for these purposes (Salas et al., 2016). This is mainly a result of continuous illegal selective logging of native forests, which has resulted in forest degradation. Consequently, these native forests require further economic investment to become productive which is unfeasible for smallholders, which form the majority of the owners of native forest plots (Salas et al., 2016). Although forestry law offering small subsidies for native forestry exist, its effect has been negligible (Salas et al., 2016).

Despite the success of the current forestry business model in Chile, there is controversy around it due to the contrasting perceptions of different stakeholders (Salas et al., 2016). Environmental and social concerns are the major challenges of the forestry sector nowadays. Such concerns include the increasing presence of large alien monoculture plantations, clearcuttings, and degradation of the native forest through selective logging (Salas et al., 2016). The conversion of native forest into alien plantations has had a detrimental impact on biodiversity and above ground carbon storage. Models estimate that, between 1986 and 2011, approximately 12 ha of native forests were lost to subsidised productive forestry in Chile, resulting in a net decrease in above ground carbon storage of  $46\pm 87$  ktC and a decrease in area standardised species richness of  $0.0016\pm 0.00024$  s.d. for the whole country (Heilmayr, Echeverría and Lambin, 2020). Alien plantations store  $37$  t ha<sup>-1</sup> of carbon, while the Valdivian temperate forests store  $146$  t ha<sup>-1</sup> (Heilmayr, Echeverría and Lambin, 2020). This large difference means that for each hectare of Valdivian temperate forest lost, almost 4 hectares of alien plantations need to be established to compensate for the carbon released. Conversion of native forest into alien

plantations has promoted biological invasions through species introduction (Peña et al., 2008). In productive plantations, harvesting is performed through extensive clearcuttings. Chilean law has no limitations on maximum areas or slopes for clearcuttings, leaving extensive areas bare, with the subsequent detrimental effect on biodiversity, water and soil processes, and landscape aesthetics (Salas et al., 2016).

The FSC (Forest Stewardship Council) certifies that a forest is being managed sustainably by verifying that it adheres to a series of principles relating to tenure rights and responsibilities, indigenous people's rights, workers' rights, efficient use of the forest, environmental impact, adequate planning, monitoring and assessment, and maintaining high conservation value forests (FSC, 1996). While Chilean forestry companies have become more environmentally conscious, with 52% of the plantations being certified by the FSC, extensive clearcuttings and monocultures are still common practice (Salas et al., 2016). A change in the forestry sector is required, with more strict conservation policies to reduce environmental impacts and adequately protect the Valdivian temperate forest from further degradation and fragmentation. Conservation should be a priority, and forest plantations should be managed at the landscape level to reduce negative environmental impact (Salas et al., 2016).

### 1.5. Alien trees in south-central Chile

Invasion by alien tree species has grown in relevance in the last decades - there are more invasive trees, larger invaded areas, greater variety of impacts and many complex management challenges being recognised (Richardson et al., 2014). However, trees are still relatively understudied despite the often substantial impacts of the invasion process (Richardson et al., 2014).

Fuentes et al., (2008) analysed herbarium records and identified an expansion of alien flora in Chile between 1980 and 2000, coinciding with a period of increasing forestry production. A comprehensive database of alien species created by Fuentes et al. (2013) includes 743 alien plant species, divided in 361 genera and 74 families, representing about 15% of Chile's total flora. They are present in all territories, but mostly clustered in the central part of the country, and spreading northwards and southwards (Fuentes et al., 2008). The most invaded regions are those with humid Mediterranean or temperate climates (Casado et al., 2015). It is important to recognise that most forest invasive plants were first voluntarily introduced, and

quite often through productive plantations such as those in south-central Chile (Haysom and Murphy, cited in Dodet and Collet, 2012: 1766). According to the same authors, among 443 forest plants categorised as invasive, 292 are typical of productive plantations, most of them belonging to the Leguminosae, Pinaceae, Myrtaceae, Rosaceae, and Salicaceae.

Plant invasions in Chile are closely related to forest fragmentation, silvicultural systems and land use change. In native forests, the highest density of invasive species is found in the centre of canopy gaps and in forest edges (Arellano-Cataldo and Smith-Ramírez, 2016). Forest plantations and fragmented forests have the highest amount of alien species (Becerra and Simonetti, 2013). In forest plantations, alien species can quickly invade after management operations, inhibiting the establishment of native species (Heinrichs, Stiehl and Müller-Using, 2016). Remaining native forest fragments are considered of high importance for native biodiversity conservation. However, the fact that they also host large numbers of alien and invasive species may limit their conservation value. Consequently, it is of utmost importance to control invasion within them (Becerra and Simonetti, 2013). Invasive plants are present within protected areas, but they tend to be limited to road edges and open areas with high human influence, being uncommon in the shaded understory (Pauchard and Alaback, 2004).

The high productivity of the trees used in productive forestry plantations relies on their easy establishment, fast growth and low shadow tolerance. These characteristics are also very common in invasive species (Dodet and Collet, 2012). Several conifers often used in plantation forests are considered invasive in some parts of the world (Pauchard and Alaback, 2004). In Chile, there are *Nothofagus* sp. forest fragments of the Coastal Range that are becoming invaded by the *Pinus radiata* that grows in the surrounding plantations (Bustamante and Castor, 1998, Echeverría et al., 2007). Peña et al. (2008) found alien *Pinus contorta* Douglas ex Loudon regeneration within a protected area in south-central Chilean Andes, compromising the conservation of the endemic *Araucaria araucana*. Landscape level management, rather than species-by-species approach could be the most cost-effective way of controlling invasive trees (Marvier, Kareiva and Neubert, 2004).

This research project focuses on a group of alien trees from the Pinaceae family and two species from the *Eucalyptus* genus (Myrtaceae) that are widely employed in

productive forest plantations in Chile. These species are *Pinus contorta*, *Pinus radiata*, *Pinus ponderosa* Douglas ex C.Lawson, *Pinus sylvestris* L., *Pseudotsuga menziesii* (Mirb.) Franco, *Eucalyptus globulus*, and *Eucalyptus nitens*. A brief description of the ecology of each of these is provided in the following paragraphs.

*P. contorta* is a North American pine tree that covers 33° of latitude and 3900 m of elevation within its native range in western North America, being able to thrive in a wide variety of environmental conditions (Rehfeldt et al., 1999). It can withstand minimum temperatures of -57°C and maximum temperatures of over 38°C (Lotan and Critchfield, 1990). Consequently, *P. contorta* can establish successfully in a wide range of environmental conditions, including poor sites, having great potential to become invasive (Despain, 2001). Seedlings are especially successful in low competition sites with mineral soils with granites, shales or lavas as parent materials (Lotan and Critchfield, 1990, Engelmark et al., 2001). *P. contorta* seed dispersal is largely determined by wind direction and force, although occasionally they are also dispersed by vertebrates. Exposed sites and storm events favour long-distance dispersal of propagules (Engelmark et al., 2001). *P. contorta* is adapted to reproduce after fire (Despain, 2001). A study in parts of Patagonia and New Zealand found that *P. contorta* invasions increase fuel load in the forest and soil heating during fire events (Taylor et al., 2017). In *A. araucana* forests, the presence of *P. contorta* increases vertical fuel continuity (Cóbar-Carranza et al., 2014). This increases the probability of severe crown fire events in *A. araucana*. When *P. contorta* densities are high (above invasion threshold), soil heating during forest fires may have a positive impact on post-fire *P. contorta* regeneration, accelerating invasion (Taylor et al., 2017). Many trees within this species exhibit serotiny, especially when they are older than 20-30 years (Despain, 2001, Engelmark et al., 2001). Serotinous pine cones are sealed by resin and only open when exposed to high heat (Muir and Lotan, 1985, Engelmark et al., 2001). Forest fires provide the required high heat for serotinous cones to open and spread abundant *P. contorta* propagules.

*P. radiata* is native from California and parts of Mexico, with the natural populations occurring mainly in Mediterranean or maritime regions. It withstands temperatures between -5°C and 41°C. It is a very adaptable tree that has been introduced in many regions of the world for productive forestry due to its quick growth and quality wood (Richardson and Brown, 1986, McDonald and Laacke, 1990, Mead, 2013). Hail and snow are limiting factors for *P. radiata* establishment and development (Mead, 2013).

It can grow in many soil conditions, but the best are deep sandy loams, generally derived from marine sediments (McDonald and Laacke, 1990, Mead, 2013). Like other *Pinus* spp., the cones of *P. radiata* open when environmental conditions are warm and dry. Consequently, forest fires cause large releases of seeds. In addition, the open, low competition, full sunlight conditions created by large forest fires or clearcuttings lead to quick regeneration and seedling growth (McDonald and Laacke, 1990). *P. radiata* has serotinous cones, and dense sprouting occurs after fire events (Richardson and Brown, 1986). A previous study about *P. radiata* invasion in *Nothofagus* spp. forests in Chile found that canopy openness, and hence light availability, was the most relevant variable determining *P. radiata* regeneration (Gómez et al., 2019). The same study found that the native community did not pose any biotic resistance to *P. contorta* spread.

*P. ponderosa* is native from Western North America, with an extensive distribution stretching from Canada to Mexico. It can grow at elevations up to 3050 m and withstand extreme temperatures from -40 to 43°C. It grows on soils derived from igneous, metamorphic and sedimentary materials. Dry soil conditions during low rainfall periods limit *P. ponderosa* growth (Oliver and Ryker, 1990). In their native range, *P. ponderosa* forests have a scant herbaceous stratum due to allelopathic compounds produced by the trees (Lodhi and Killingbeck, 1982). *P. ponderosa* starts producing cones at 7 years old. Seeds do not normally travel further than 30 m from the parent tree, but long distance dispersal can occasionally happen (Oliver and Ryker, 1990, Lesser and Jackson, 2013). *P. ponderosa* is adapted to resist wildfire. Adult trees can resist fire events due to their thick bark (Oliver and Ryker, 1990). Saplings typically die during high intensity fires but can withstand low intensity fires if they are well watered before the fire event (Partelli-Feltrin et al., 2020).

*P. sylvestris* is widely distributed in its native range across Eurasia, growing in many different environmental conditions and even in poor soils. It is present from Spain in the west to far east Russia. Its northern limit is in Scandinavia, and its southern limit is in the south of Spain. It grows from sea level to 2600 m elevation, depending on the site (Houston Durrant, de Rigo and Caudullo, 2016). However, climate change is causing a shift in *P. sylvestris* distribution, as the southernmost populations are being replaced by other species due to increasing temperatures and aridity (Matías and Jump, 2012). *P. sylvestris* is mainly dispersed by wind and, consequently, most seeds germinate at short distances from the parent tree, with occasional longer

distance dispersal (Calama et al., 2017). *P. sylvestris* possesses various fire adaptations such as thick bark, deep roots, and rapid regeneration in mineral soils and open conditions. Low intensity fire that does not completely burn the organic horizon of the soil tends to be survived by adult *P. sylvestris*. High intensity fire events may kill large *P. sylvestris* trees, but this species quickly and abundantly regenerates after fire, as low competition and light availability benefits this pioneer species (Kuuluvainen and Rouvinen, 2000, Adámek, Hadincová and Wild, 2016). Like other Pinaceae species, *P. sylvestris* benefits from the changes in light conditions after fire. In addition, frequent burnings favour the dominance of *P. sylvestris* in temperate natural and semi-natural forests in Europe (Adámek, Hadincová and Wild, 2016).

*P. menziesii* is a highly adaptable North American conifer growing from British Columbia to Mexico along the Pacific Coast and the Rocky Mountains. Within its native range, it can be found from sea level to 3200 m elevation (Da Ronch, Caudullo and de Rigo, 2016). *P. menziesii* thrives in a wide range of soil conditions and is drought resistant, but its growth is limited by the increasing temperatures occurring as a result of climate change (Eckhart et al., 2019). Seeds from *P. menziesii* can sometimes reach long distances, but most often fall within 100 m of the parent tree. Saplings benefit from shade at early stages of development. However, this species can also regenerate completely exposed to sunlight (Hermann and Lavender, 1990, Da Ronch, Caudullo and de Rigo, 2016). Its quick growth, longevity, thick corky bark, and ability to grow adventitious roots make *P. menziesii* a strong competitor during and after forest fires. Repeated wildfires create large monospecific *P. menziesii* stands within its native range (Hermann and Lavender, 1990).

*E. globulus* is native from south-eastern Australia and is widely used in plantations for industrial purposes due to its fast growth and its ability to resprout from dormant buds in the stem. Lack of water or nutrient availability and low temperatures limit its growth (Cesaroli et al., 2016). *E. globulus* does not produce abundant seeds, and most of them are dispersed within 15 m of the parent tree (Silva and Marchante, 2012, Calviño-Cancela and Rubido-Bará, 2013). However, many Myrtaceae species, among which the *Eucalyptus* genus is included, have epicormic (under bark) buds that enable the tree to resprout after a fire, being a strong competitors in this scenario (Crisp et al., 2011, Catry et al., 2013). As an alternative to *E. globulus*, *E. nitens* is

another Australian tree that is widely used in productive plantations and it is better adapted to low temperatures than *E. globulus* (Cesaroli et al., 2016).

Fire has been a determining factor in the evolution of both Pinaceae and the *Eucalyptus* genus (Crisp et al., 2011, Pausas, 2015). As mentioned in the species descriptions above, both groups have advantageous characteristics for reproduction under a recurrent fire regime. Many Pinaceae exhibit serotiny, which enables them to regenerate abundantly, even after fire. *Eucalyptus* spp., due to their epicormic buds, resprouts after fire events. These two strategies, abundant seedling recruitment and resprouting from surviving tissues, allow Pinaceae and *Eucalyptus* spp. populations to persist even after severe fires (Crisp et al., 2011, Pausas and Keeley, 2014, Pausas, 2015). As mentioned in section 1.3., species that grow quickly after fire like Pinaceae and *Eucalyptus* spp. are uncommon in South American forests (Donoso, González and Lara, 2014: 674). In a context of increasing temperatures and frequency of severe fires (González, Veblen and Sibold, 2005, González et al., 2018), alien Pinaceae and *Eucalyptus* spp. could displace the native *A. araucana* and *Nothofagus* spp., as these native trees do not resprout or regenerate as quickly and abundantly after fire (Drake, Molina and Herrera, 2012, Kitzberger et al., 2016).

Different terms are used throughout this project to refer to the various types of alien and native forested land covers. The term “alien plantation” is used for actively managed productive monocultures using alien tree species. “Alien forest stand” or “alien forest cover” is used for any type of area covered mostly by alien trees. This can be a plantation, an abandoned plantation, or a product of natural invasion. “Native forest” or “undisturbed native forest” is used for mature native forests that show no apparent signs of recent disturbance. Finally, “early successional native forest” is used for naturally regenerating native forest in an area that shows signs of recent disturbance.

## 1.6. Satellite remote sensing for alien species monitoring

Alien plants, such as the trees used in productive forestry, are challenging to completely eradicate once they have become invasive, with this only realistically possible in early stages of invasion (Schmiedel et al., 2016). Consequently, invasion monitoring and rapid, targeted management are the most effective tools for invasive species control (Pauchard et al., 2016, Nuñez et al., 2017). Adequate forest

management at both the landscape and local scale is essential to control invasive species (Salas et al., 2016, Sitzia et al., 2016).

In this context, remote sensing is a highly valuable tool, as it enables continuous vegetation monitoring over large areas including abrupt topography – as is often the case in Chilean forested areas – which inhibits ground-based survey. Remote sensing obtains information about an object (the Earth surface) without establishing physical contact with it. Sensors can be located on the ground, on an airborne platform, or on a satellite (as is used in this study). Sensors can also be passive or active, depending on their energy source (Huang and Asner, 2009). Passive sensors, such as those used in this study, detect the solar radiation reflected by features on the Earth's surface. Remote sensing, especially satellite based, allows questions to be addressed on very broad scales – impossible using ground-based methods – enabling integrated management (Pettorelli et al., 2014), early invasion detection and invasion modelling (Bradley, 2014). In addition, satellite remote sensing can assess the consequences of particular management decisions thanks to continuous monitoring (Pettorelli et al., 2014). Despite this potential, remote sensing is an underutilized tool in invasion ecology (Bradley, 2014).

There are many types of remote sensing imagery available from public and private satellites. Typically, there is a trade-off between extent of the image, spatial resolution (pixel size), range of spectral bands and frequency of acquisition (Bradley, 2014). Among the most widely accessible multispectral satellite imagery sources are Landsat 8 (and the archive of imagery from earlier Landsat missions) and Sentinel-2, both of which are freely available. Although these products have a large spatial coverage, they have moderate spatial resolution (10-30 m pixel sizes), which would only allow the mapping of widespread invasive species that form large homogeneous patches (Huang and Asner, 2009). Despite its limitations, moderate resolution multispectral imagery such as that from the Landsat or Sentinel-2 missions has been used quite widely for alien species mapping (Matongera et al., 2017, Ndlovu et al., 2018, Calleja et al., 2019). However, it is not possible to use them to map early stages of invasion, as their spatial resolution only allows detection of large patches (i.e. advanced stages of invasion). The small patches formed by early invasions could potentially be detected using very high resolution (VHR) multispectral satellite imagery such as WorldView or IKONOS, which have pixel sizes as small as 2 m. This type of imagery is the most intuitive data source to map plant invasions (Huang and



Asner, 2009). VHR imagery has previously been successfully used for alien species monitoring in forests (Alvarez-Taboada et al., 2017; Malahlela et al., 2015; Peerbhay et al., 2016). Hyperspectral imagery can identify plant pigments over hundreds of spectral bands, as hyperspectral sensors collect a continuous region of the electromagnetic spectrum. This has been widely used for alien plant detection (Huang and Asner, 2009). However, both VHR and hyperspectral imagery are typically limited in spatial coverage and revisit frequency, and can often be costly (Bradley, 2014).

Remote sensing image analysis for alien species mapping generally uses pixel-based spectral methods (Bradley, 2014). The land cover types (or species) being mapped need to be sufficiently spectrally distinct to be mapped accurately. Consequently, the alien species of interest must have a special spectral or structural characteristic that separates it from the native vegetation (Ustin and Gamon, 2010). Pixel-based spectral differentiation requires the invasive species to have a pattern of light absorption and reflection (spectral signature) different to the native vegetation. Spectral differences are more easily captured if the invasive plant is a different functional type from the surrounding, native vegetation (Bradley, 2014). Unlike pixel-based approaches, object-based image analysis (OBIA) recognises patterns within a neighbourhood of adjacent pixels using contextual information (Blaschke, 2010, Bradley, 2014). Object-based methods for image classification have two steps. Initially, the image is segmented into sets of pixels known as objects, and then each object is classified. This approach requires the pixel size to be smaller than the objects that are being segmented (individual trees, patches) to be able to identify meaningful objects (Benz et al., 2004, Bradley, 2014). OBIA can potentially reduce the noise, often termed 'salt and pepper' effect, that pixel-based spectral approaches create when using VHR imagery (Blaschke, 2010).

Using time series and vegetation indices such as NDVI (normalized difference vegetation index), it is possible to detect key phenological stages (Huang and Asner, 2009, Bolch et al., 2020) and separate vegetated land covers that have a distinct spectral signature at a particular time. To use phenology for invasive plant detection, the invasive plant needs to have a different seasonal or inter-annual growth pattern than the native vegetation. This approach requires the creation of time series of vegetation indices, requiring large amounts of imagery. Consequently, freely available moderate resolution sensors such as Landsat 8 and Sentinel-2 are the most

widely used for this approach (Bradley, 2014). This project uses multi-seasonal imagery to separate different forested land covers accurately. Different tree species exhibit different phenology throughout the year. As a result of this different phenological stages, two tree species that are spectrally similar at a particular time of the year can be spectrally dissimilar at other times. Using multi-seasonal imagery exploits this difference by using data from different, key phenological moments during the year, improving tree species discrimination.

### 1.7. Research aim and objectives

This is a landscape ecology project that uses remote sensing to assess alien tree presence, expansion, and invasion. The questions that inspired it were the following: 1) which remote sensing data sources are most suitable to assess alien tree occupancy, expansion, and invasion?; 2) where are the alien trees located within the Valdivian temperate forest?; and 3) is there a relationship between alien tree occupancy and landscape configuration?.

The aim of this research project is to understand alien tree expansion, invasion, and occupancy in the Valdivian temperate forests at different spatial scales to inform forest and conservation management and improve the sustainability of the forestry sector in Chile.

To achieve this aim, the following objectives are defined: 1) evaluate the capabilities of different satellite remote sensing data sources to detect alien trees accurately; 2) assess the current extent and invasion of alien trees in the Chilean Valdivian temperate forest at different spatial scales; 3) assess the expansion and invasion of alien trees over time; 4) relate the presence of alien trees and degree of invasion to the landscape spatial configuration and fragmentation at different scales; and 5) provide evidence-based management recommendations to improve forest and conservation management.

### 1.8. Thesis outline

This thesis includes three research chapters, each of which addresses one or more of the research objectives mentioned in section 1.7. The first objective, evaluating the capabilities of different satellite remote sensing data to detect alien trees, is fulfilled throughout the three research chapters and the conclusion, as each chapter uses a

different remote sensing data source, and the conclusion chapter discusses the pros and cons of each.

The second objective, assessing the current extent and invasion of alien trees, is tackled in chapters two and four. Each of these chapters assesses the current extent and invasion of alien trees at a different spatial resolution and scale. Chapter two assesses the extent of alien trees over an extensive territory using moderate resolution satellite imagery. Consequently, most of the alien tree patches mapped are plantations. Chapter four focuses on isolating natural invasion in localised study areas by using VHR satellite imagery.

The third objective, assessing the expansion of alien trees over time, is covered by chapter three, which is a temporal study that assesses the expansion of alien tree cover over last decades. Again, the use of moderate resolution imagery limits the detection of early invasion, but overall patterns of forest conversion and alien plantation expansion are assessed.

The fourth objective, relating the presence of alien trees to the landscape spatial configuration, is tackled in chapters two and four at different spatial scales. Chapter two relates the presence of alien plantations to the landscape spatial configuration at a broad scale, and chapter four relates alien tree invasion patches to the landscape spatial configuration at a local scale.

The last objective, providing evidence-based management recommendations, is discussed in the conclusion chapter, drawing on the results of the three research chapters.

### 1.8.1. Introduction

Chapter one identifies and introduces the overarching theme investigated throughout this thesis, the problem of alien tree invasions stemming from productive forestry plantations into the environmentally sensitive Chilean native forest. It also explains why remote sensing is a valuable tool to address this environmental problem. Background contextual information in relation to biological invasions and remote sensing are provided. The Chilean Valdivian temperate forests are characterised, and the nature and impacts of the forestry industry are explained. Finally, the aims and objectives of the research are stated, and an outline of the thesis is provided.

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

Chapter two compares the degree of alien tree occupancy at the landscape scale between two study areas in south-central Chile and models its relationship with the landscape configuration of the sites. Land cover maps of the two study areas are initially generated using Sentinel-2 data and a pixel-based method. These maps include six different vegetated classes, two alien tree groups, two types of native forest, shrublands and grasslands. Despite being able to separate very spectrally similar classes such as the different alien and native forested classes, the moderate spatial resolution of Sentinel-2 (10 m pixel size) does not allow for the detection of early stages of invasion, being mostly limited to fairly large stands or plantations. The boom of the forestry industry in Chile started in the 1970s. In some areas of the country, severe deforestation has occurred as a result of alien plantations over the last decades. Examining the expansion of alien plantations, and the land cover change during this time period requires historic satellite imagery, and Sentinel-2 has only been operating for the past 5 years.

### 1.8.3. Mapping the dynamics of alien plantation spread and native forest loss in Chile using Landsat time series analysis and Google Earth Engine

Chapter three investigates alien tree expansion in a study area in south-central Chile at the landscape scale over the last four decades using archive Landsat imagery. Land cover maps for each decade are generated using a pixel-based method, with a land cover change analysis focusing on alien species spread and native deforestation performed. Alien tree species distribution patterns are identified over the 1980s, 1990s, 2000s and 2010s decades, and the dynamics of land cover change are analysed to identify the relationship between alien species spread and native species deforestation. As in the second chapter, early stages of invasion cannot be mapped using Landsat imagery due to the spatial resolution (30 m pixel size), requiring a higher spatial resolution imagery.

#### 1.8.4. Monitoring alien invasive Pinaceae in Chilean temperate forests using object-based image analysis of WorldView-3 imagery

Chapter four investigates alien presence at a different scale of observation, the local scale, using VHR satellite imagery to detect the early stage of alien tree invasion, and relating this to landscape configuration. In this chapter, VHR WorldView-3 imagery and OBIA is used to map early invasion in areas surrounding plantations. Then, the patterns of occupancy of different alien and native tree species are analysed in relation to distance to alien plantations and fragmentation. Alien plantations would theoretically act as seed sources, being one of the major drivers of invasion (Richardson et al., 2000). Landscape fragmentation, on the other hand, has been identified as being closely linked with invasion (Marvier, Kareiva and Neubert, 2004, León Cordero et al., 2016). Using VHR imagery can detect the early stage of invasion, enabling these hypotheses to be tested in this environment.

#### 1.8.5. Conclusions

Chapter five summarises the relevance and main findings of the thesis and draws upon the previous chapters to provide general conclusions regarding alien tree invasions within the Chilean Valdivian temperate forest. This chapter overviews how each research objective of the thesis has been fulfilled, and provides general concluding remarks regarding each of them. In addition, a series of evidence-based management strategies to ensure sustainable forest management that minimises the risk of biological invasions are provided. Finally, the limitations of this research project are discussed, together with suggestions of future research directions that could contribute further to biological invasion science and forest management.

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

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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 elevations, there was greater total area covered by alien trees and more fragmented native forests than at higher elevations. 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.

Key words: Alien species, fragmentation, temperate forests, Sentinel-2, random forests, land cover.

### 2.1. 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, Stehman and Potapov, 2010), land use change (Ellis, 2011) and are increasingly

threatened by the invasion of alien trees (Essl, Milasowszky and Dirnböck, 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: 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 elevations 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, Bone and Holzapfel, 2000). Particularly, the endemic tree *Araucaria araucana* (monkey puzzle) is classified as endangered in the IUCN (International Union for Conservation of Nature) Red List of Threatened Species due to its decreasing population (Premoli, Quiroga and Gardner, 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).

Significant land use change has occurred in the Valdivian temperate forests since the 1970s, 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, Teillier, Rodríguez and Serra, 2003, Echeverría et al., 2007) and around the world (Pauchard and Alaback, 2004, Peña et al., 2008, Dziki et al., 2016). 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 broad 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 broad 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) moderate 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 (Ndlovu et al., 2018, Calleja et al., 2019). Additionally, Sentinel-2 has been shown to have the potential to discriminate alien plants from native plants with a similar morphological structure (Teschfamiel 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.

## 2.2. Materials and methods

### 2.2.1. Study areas

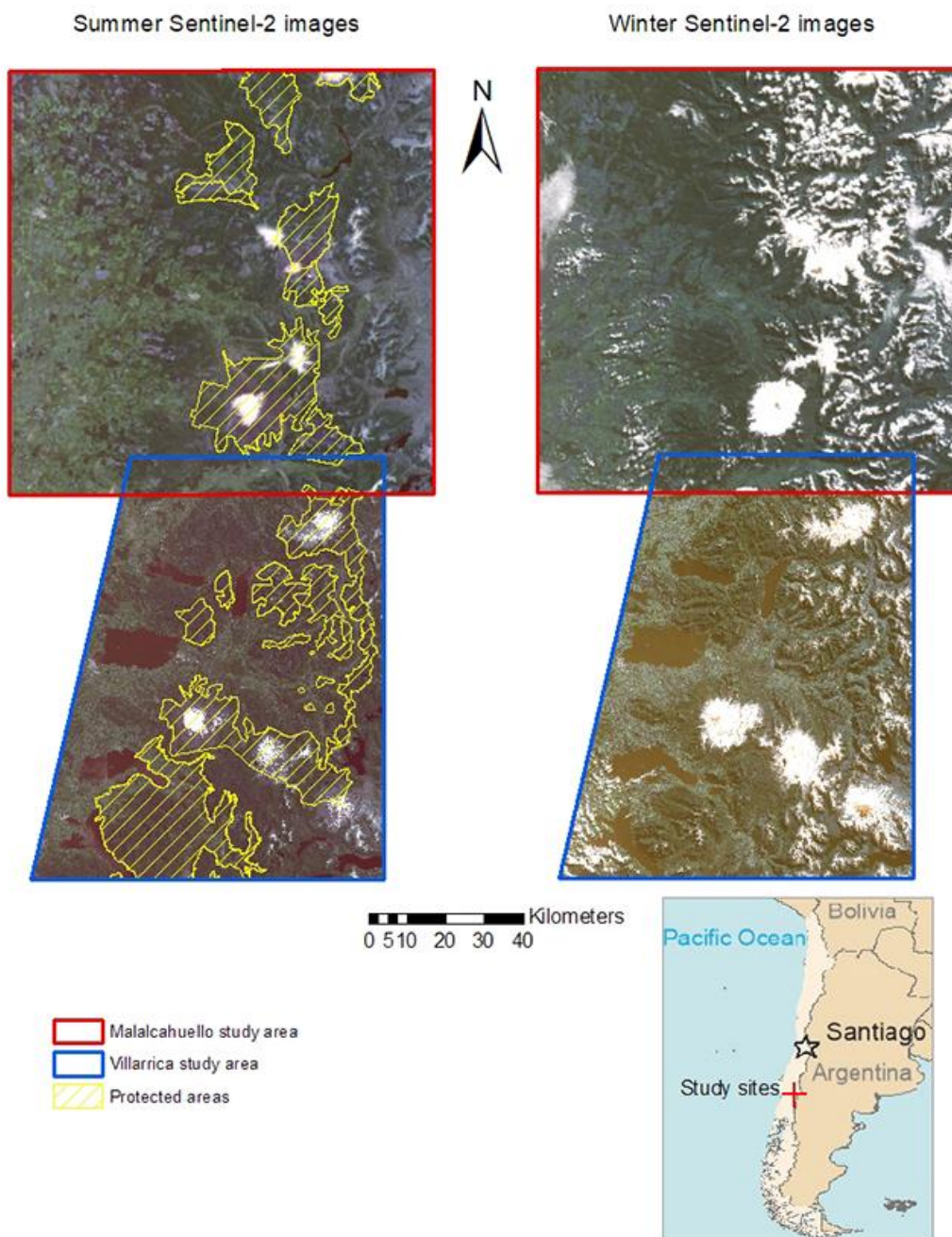
Two Valdivian temperate forest study areas were selected in the south-central Chilean Andes within the Araucanía region (IX administrative region), offering a mosaic of similar land cover, and both in the Andes and Andean foothills. Study area 1 (figure 2.1) comprises Malalcahuello National Reserve and surrounds in the northern Araucanía region ( $-38^{\circ}$  latitude), at elevations from 500 to 2800 m. Study area 2 (figure 2.1) covers Villarrica National Park and surrounds in the southern Araucanía region ( $-39^{\circ}$  latitude) at elevations from 200 to 2800 m. Both study areas are roughly 10000 km<sup>2</sup>. The Villarrica study area (42 inhabitants/km<sup>2</sup>) is more densely populated than Malalcahuello (10 inhabitants/km<sup>2</sup>) (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 areas 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 (figure 2.1) that accounts for this gradient.

Both sites can be divided in two subsets based on elevation. The lower elevation subsets are in valley areas (<600 m elevation), while the higher elevation (>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.



**Figure 2.1** Location of the Malalcahuello and Villarrica study areas and protected areas.

### 2.2.2. 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 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, Pauchard and Aguayo, 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 (Peña and Langdon, 2007, Pauchard, Langdon and Peña, 2008). 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 methods.

*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, 2009), however they are planted widely in Chile, raising concerns about their role in landscape change and native forest loss in numerous studies (Echeverría et al., 2012, Altamirano et al., 2013, Heilmayr et al., 2016).

### 2.2.3. Data sources

Medium spatial resolution multispectral Sentinel-2 level-1C imagery (figure 2.1) 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 (figure 2.1), 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, Cheshire and Humes, 2001). Additionally, the 30 m resolution ASTER (Advanced Spaceborne Thermal Emission and Reflection Radiometer) Global Digital Elevation Map Version 2 (Tachikawa 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 area. 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 Malalcahuello, 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 area.

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 (CONAF, 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), including WorldView 3 imagery, were used to complement the reference data. More detail about how these field and data sources were used to derive qualitative land cover labels is available in appendix A.

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.

#### 2.2.4. 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 (Song et al., 2001, Bakr et al., 2010, Lin et al., 2015). 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 (Pekkarinen, Reithmaier and Strobl, 2009, Kamlun, Bürger Arndt and Phua, 2016).

#### 2.2.5. Land cover classification

The classification system (table 2.1) 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, 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.

**Table 2.1** Land cover class nomenclature, abbreviations and class descriptions.

Land cover class		Class description
Alien tree stands	Coniferous plantations and invasion (CP)	Pinaceae family ( <i>Pinus radiata</i> , <i>Pinus contorta</i> , <i>Pinus ponderosa</i> , <i>Pinus sylvestris</i> , <i>Pseudotsuga menziesii</i> )
	Broadleaved plantations and invasion (BP)	<i>Eucalyptus</i> genus ( <i>E. globulus</i> and <i>E. nitens</i> )
Native forests	Native <i>Nothofagus</i> spp. (broadleaved) forests (NNF)	<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.)
	Native <i>Araucaria araucana</i> (mixed coniferous-broadleaved) forest (ANF)	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> L., <i>Rosa moschata</i> Herrm., <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> (Gay) Meisn.. Sometimes including smaller size, stunted <i>Nothofagus</i> spp. individuals.
Grassland	Agricultural grasslands or livestock grazing plots dominated by <i>Holcus lanatus</i> , <i>Poa nemoralis</i> L., <i>Nothofagus obliqua</i> and <i>Luma apiculata</i> (DC.) Burret. 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.

For each study area, 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, Rodriguez-Galiano et al., 2012). This ensemble model classifies land covers accurately (Rodriguez-Galiano et al., 2012, Müllerová et al., 2017, Marston and Giraudoux, 2019), and is well established for classifying multispectral imagery (Rodriguez-Galiano et al., 2012). Random forests have been used previously to detect invasive plant species by Naidoo et al. (2012) with 87.68% accuracy, Peerbhay 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.

#### 2.2.6. 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, McGarigal and Cushman, 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 2.2) 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: coniferous alien stands, broadleaved alien stands, *Nothofagus* spp. native forest, *Araucaria* spp. native forest, shrubs and grasslands. Two landscape level metrics (table 2.2) were also included: the aggregation index and the Shannon's diversity index.

**Table 2.2** 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 (Echeverría et al., 2008, 2012, Altamirano et al., 2013, Molina et al., 2015).
	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 (Echeverría et al., 2012, Altamirano et al., 2013, Molina et al., 2015).

	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 <sup>2</sup> ) 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, McGarigal and Cushman, 2004, Wang, Blanchet and Koper, 2014). For class areas between 5% and 80%, PAFRAC is independent to changes in area (Neel, McGarigal and Cushman, 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).
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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 area by pooling sub-site results. There is a natural divide between lower elevation areas that are more actively managed including alien species plantations and agriculture, and less impacted higher elevation areas including protected areas, where invasion has also been recorded (Peña et al., 2008, Pauchard et al., 2014). The boundary dividing the lower and higher elevation subsets for each study area is 600 m elevation, with this being the minimum elevation at which *A. araucana* can thrive (Premoli, Quiroga and Gardner, 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 km<sup>2</sup> 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 broad scale landscape studies (Gonzalez-Abraham et al., 2007, McGarigal, Tagil and Cushman, 2009). Each point and sampling buffer were identified as either in the low (valley subset) or high (Andes subset) elevation 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 elevation subset, 64 in the high elevation subset and 18 inside protected areas. For Villarrica, 75 were at low elevation, 126 at high elevation and 48 inside protected areas. The minimum patch size considered was 100 m<sup>2</sup>, with the intention

of capturing the earliest stages of invasion possible at this resolution. However, it is important to mention that, at this moderate resolution, the predominant alien tree stand detection will be plantation, especially in the case of larger patches and in easily accessible areas.

#### 2.2.7. 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, 2009); 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 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 squared 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, Poggi and Tuleau-Malot, 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).

## 2.3. Results

### 2.3.1. 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 2.3 shows the accuracies for all land cover classes for each composite. The full confusion matrices can be found in the appendix B. 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.

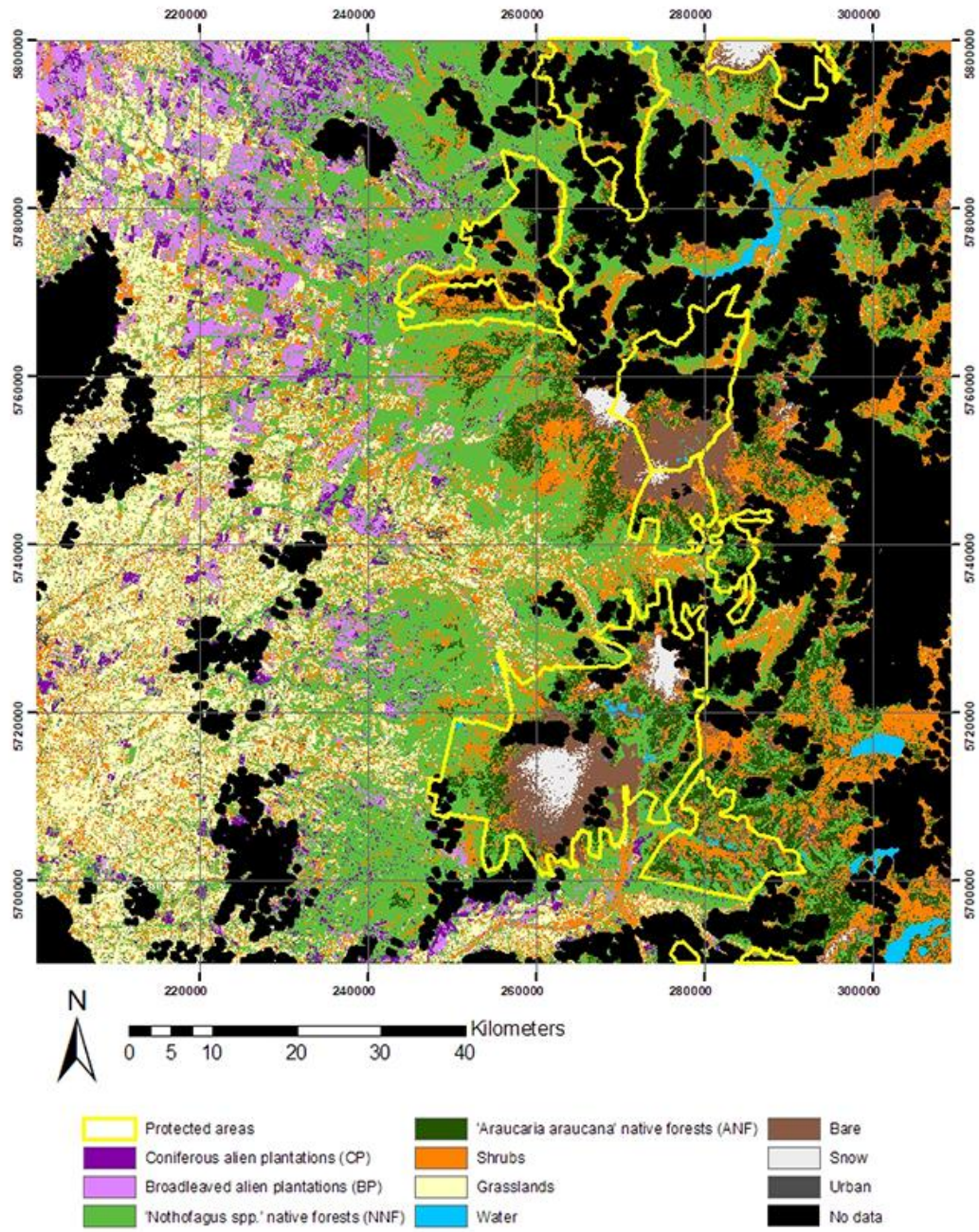
Figures 2.2 and 2.3 show the land cover maps generated using multi-season + DEM composite classification, the most accurate approach (highest overall accuracy), for Malalcahuello and Villarrica respectively. These land cover classifications were used in the next steps of this study for the landscape analysis and the modelling of Pinaceae area.



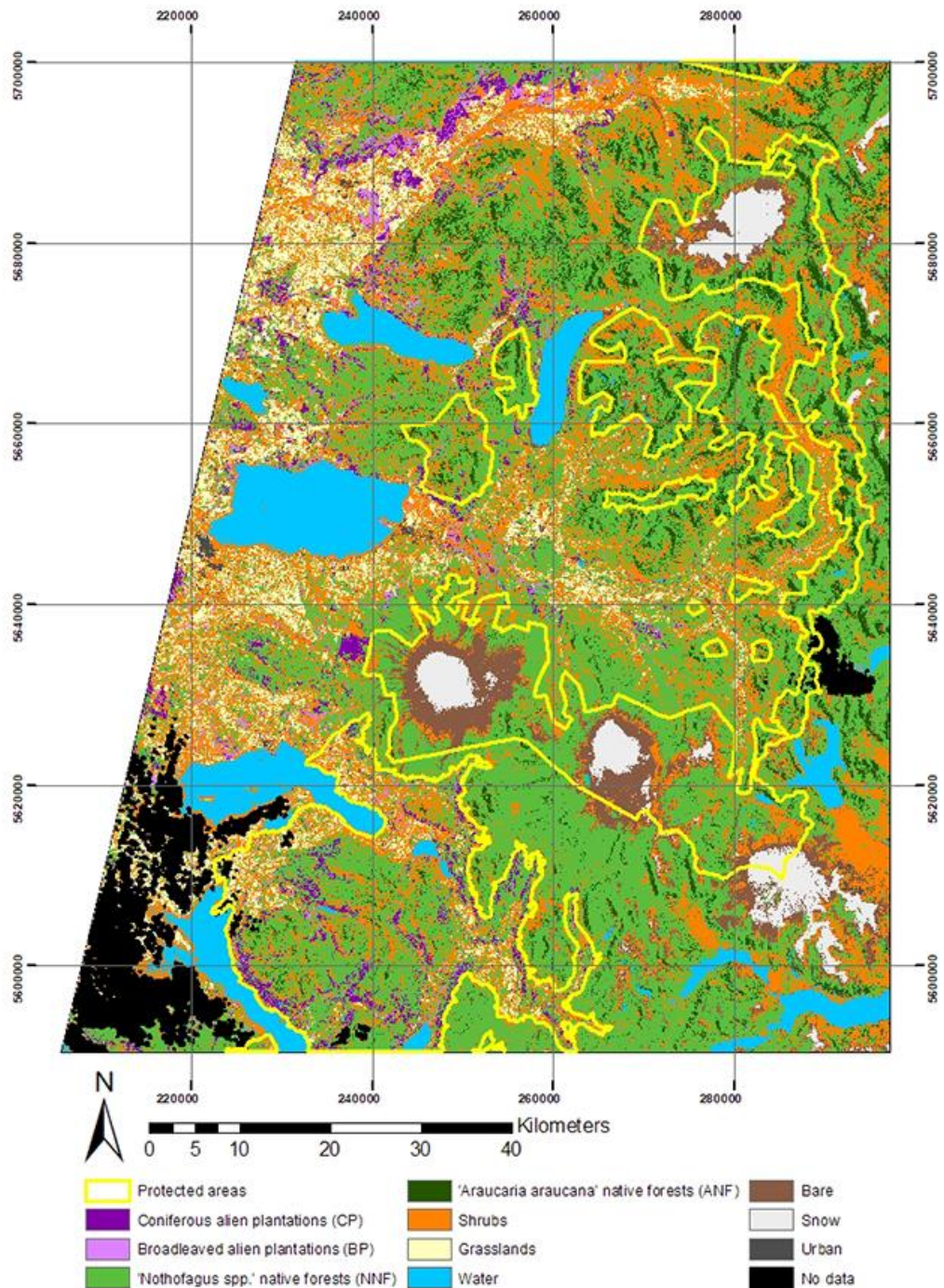
**Table 2.3** Land cover classification accuracies calculated using confusion matrices for the different image data composites.

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 class	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	



**Figure 2.2** Land cover classification of Malalcahuello using the multi-season with DEM composite.



**Figure 2.3** Land cover classification of Villarrica using the multi-season with DEM composite.

Table 2.4 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 elevation subset (valley, below 600 m) and the high elevation 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 km<sup>2</sup> are covered by coniferous alien trees, while 556 km<sup>2</sup> are broadleaved alien trees (table 2.4). These alien tree stands are located mostly in the north-west of the study area, at lower elevations and outside national park boundaries. However, some medium sized, irregular coniferous alien patches are observed inside the Malalcahuello National Park, covering 4 km<sup>2</sup> 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 elevation increases. At high elevations (> 600 m), the dominant native forests are *Nothofagus* spp. (NNF) and *A. araucana* (ANF). In fact, in the high elevation area the presence of alien trees is restricted to 75 km<sup>2</sup> for coniferous alien species, and 81 km<sup>2</sup> for broadleaved alien species. *A. araucana* forests were restricted to the eastern part of the study area, at the highest elevations.

Villarrica has a smaller overall area covered by alien trees, with 236 km<sup>2</sup> covered by coniferous alien species, and 143 km<sup>2</sup> covered by broadleaved alien species (table 2.4). Of this, 147 km<sup>2</sup> of coniferous and 112 km<sup>2</sup> of broadleaved are located in lower elevation areas, to the western part of the study area. 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 km<sup>2</sup> of coniferous alien tree cover and 23 km<sup>2</sup> 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 elevations, with *Nothofagus* spp. forests occurring mostly in the centre of the site and covering a large area.

**Table 2.4** Vegetated land cover class areas for the multi-season imagery 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 elevations (above 600 m) and *protected* are the areas inside protected areas.

Land cover class	Malalcahuello								Villarrica							
	Whole		Valley		Andes		Protected		Whole		Valley		Andes		Protected	
	Area (km <sup>2</sup> )	Area (%)	Area (km <sup>2</sup> )	Area (%)	Area (km <sup>2</sup> )	Area (%)	Area (km <sup>2</sup> )	Area (%)	Area (km <sup>2</sup> )	Area (%)	Area (km <sup>2</sup> )	Area (%)	Area (km <sup>2</sup> )	Area (%)	Area (km <sup>2</sup> )	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	2997.4	24.8	1082.5	23.1	1940.5	26.2	302.01	19.53	3513.61	40.7	667.0	22.1	2934.9	52.3	1068.1	46.7
	5	6	2	9	2	6				6	8	5	9	4	0	8
ANF	710.22	5.89	15.21	0.33	708.70	9.59	208.7	13.5	519.43	6.03	32.35	1.07	518.64	9.25	166.55	7.30
							0	0								
Shrubs	1782.49	14.79	670.53	14.3	1135.62	15.37	192.39	12.4	2104.6	24.41	947.0	31.4	1198.24	21.37	473.40	20.7
				6				4	0		4	4				4
Whole	12055	100	4668	100	7390	100	1546	100	8621	100	3012	100	5608	100	2283	100

### 2.3.2. Landscape metrics

Table 2.5 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 C.

**Table 2.5** 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 elevation subset. In the table, *whole* identifies the complete study area, *valley* corresponds to the low areas (below 600 m), *Andes* refers to the higher elevations (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 (%)	PAFRA C	CLUMPY	AI	SHDI
<b>Malalcahuello</b>	Whole	Landscape	ND	ND	ND	ND	ND	87.40	1.21
		CP	11.72	0.41	1.52	1.35	0.75	ND	ND
		BP	20.70	0.44	3.68	1.38	0.73	ND	ND
		NNF	30.15	1.89	26.2 9	1.38	0.80	ND	ND
		ANF	14.98	0.61	7.99	1.38	0.75	ND	ND

		Shrubs	58.98	0.38	4.88	1.43	0.71	ND	ND
	Valley	Landscape	ND	ND	ND	ND	ND	85.68	1.31
		CP	14.05	0.58	2.24	1.35	0.82	ND	ND
		BP	27.73	0.58	5.47	1.38	0.77	ND	ND
		NNF	33.52	1.20	15.66	1.39	0.79	ND	ND
		ANF	0.70	0.31	0.08	1.41	0.74	ND	ND
		Shrubs	70.93	0.24	2.75	1.45	0.67	ND	ND
	Andes	Landscape	ND	ND	ND	ND	ND	89.69	1.07
		CP	8.48	0.18	0.52	1.35	0.66	ND	ND
		BP	9.77	0.23	0.90	1.38	0.66	ND	ND
		NNF	25.61	2.83	40.6 4	1.38	0.82	ND	ND
		ANF	16.66	0.64	8.92	1.38	0.75	ND	ND
		Shrubs	43.05	0.57	7.72	1.41	0.76	ND	ND
	Protected	Landscape	ND	ND	ND	ND	ND	90.47	1.11
		CP	2.26	0.11	0.11	1.32	0.63	ND	ND
		BP	1.44	0.06	0.01	1.44	0.39	ND	ND
		NNF	30.46	0.90	14.8 9	1.39	0.77	ND	ND



		ANF	23.67	0.73	12.18	1.40	0.73	ND	ND
		Shrubs	28.52	0.47	7.75	1.38	0.75	ND	ND
<b>Villarrica</b>	Whole	Landscape	ND	ND	ND	ND	ND	86.81	1.15
		CP	16.84	0.21	1.35	1.41	0.65	ND	ND
		BP	35.10	0.05	0.38	1.45	0.38	ND	ND
		NNF	34.49	1.95	33.3 0	1.42	0.76	ND	ND
		ANF	38.45	0.16	2.60	1.43	0.60	ND	ND
		Shrubs	48.61	0.66	13.16	1.40	0.76	ND	ND
	Valley	Landscape	ND	ND	ND	ND	ND	85.56	1.26
		CP	19.70	0.24	1.61	1.39	0.67	ND	ND
		BP	47.04	0.08	0.72	1.47	0.51	ND	ND
		NNF	40.40	0.70	13.10	1.44	0.73	ND	ND
		ANF	11.52	0.05	0.32	1.42	0.37	ND	ND
		Shrubs	43.32	0.94	21.91	1.42	0.75	ND	ND
	Andes	Landscape	ND	ND	ND	ND	ND	87.34	1.10
		CP	14.73	0.18	1.16	1.42	0.63	ND	ND
		BP	26.85	0.03	0.14	1.44	0.28	ND	ND
		NNF	31.96	2.48	41.92	1.41	0.78	ND	ND

		ANF	47.11	0.20	3.33	1.44	0.67	ND	ND
		Shrubs	50.87	0.55	9.42	1.40	0.76	ND	ND
	Protected	Landscape	ND	ND	ND	ND	ND	88.18	1.12
		CP	30.50	0.22	2.47	1.43	0.62	ND	ND
		BP	50.61	0.03	0.09	1.48	0.35	ND	ND
		NNF	27.35	2.32	34.5 8	1.40	0.80	ND	ND
		ANF	29.01	0.17	2.06	1.42	0.65	ND	ND
		Shrubs	48.87	0.48	7.41	1.40	0.73	ND	ND

Both study areas 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 elevations 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 elevations, and the PD results are consistent, being much higher in the Andes areas of both study areas. 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 C, figure C1) shows that, even though the mean patch density (table 2.5) 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 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 C, figure C2), 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 area. 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*, 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 Malalcahuello 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) indicates that lower elevation 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, Malalcahuello shows a slightly more diverse landscape than Villarrica.

### 2.3.3. 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 Malalcahuello retaining 13 explanatory variables, and a model for Villarrica comprising 27 variables (table 2.6).

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 (Pauchard and Alaback, 2004, Gassó et al., 2009). 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 (Malalcahuello) 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 Malalcahuello 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., 2011, 2016). 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 Malalcahuello, where distance to seed sources is not present in the model. Even though Malalcahuello 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 area is being naturally invaded by these trees, even at this scale, while the establishment of new pine tree stands in Malalcahuello 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 Malalcahuello 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 away from accessible areas.

**Table 2.6** 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
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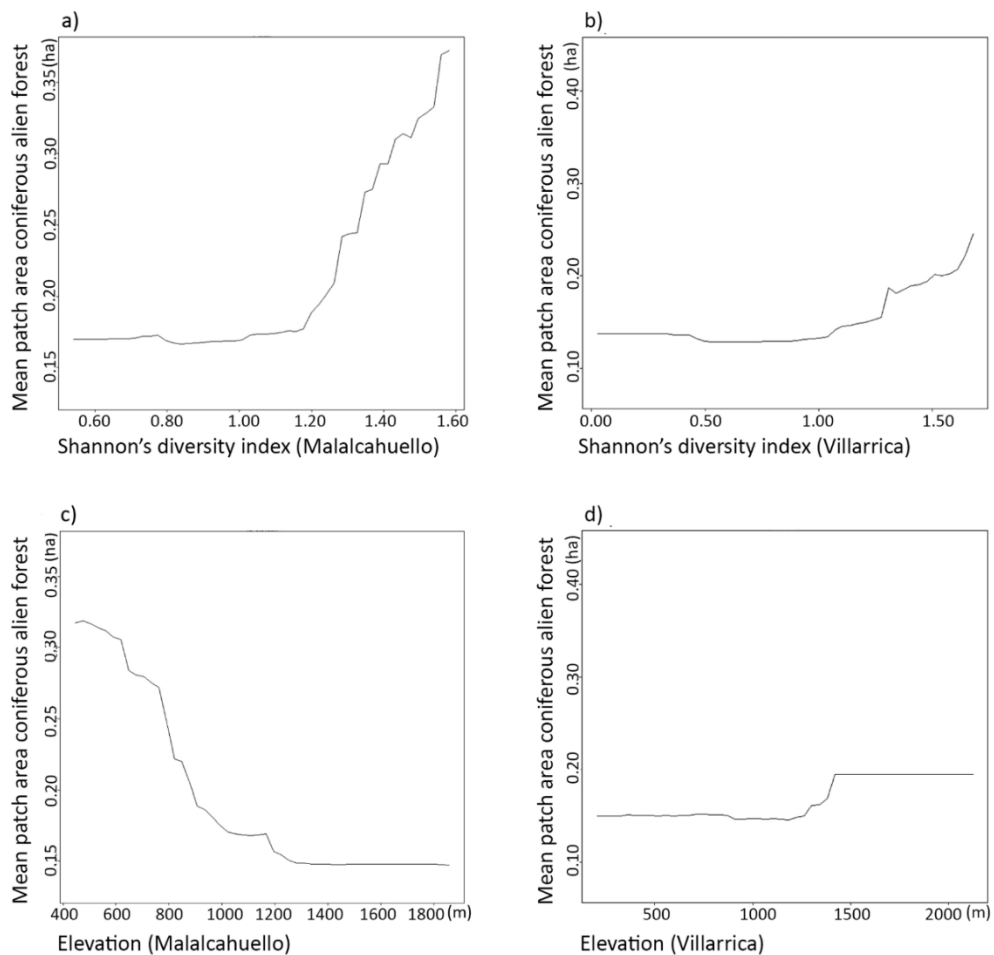
To illustrate the nature of the relationships between the explanatory and response variable, a series of partial dependence plots were generated. Only plots demonstrating a high relative importance in the random forest models and showing a meaningful relationship with the mean area of Pinaceae are shown here, with the remaining partial dependence plots presented in the appendix D. The plot for the Shannon's diversity index (figure 2.4), which is a highly relevant variable in both models (table 2.6) 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).

Elevation 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 (Pauchard and Alaback, 2004, Gassó et al., 2009). In addition, productive plantations are theoretically preferentially located in more easily accessible low and flat areas, decreasing with elevation. This hypothesis was confirmed for Malalcahuello (figure 2.4), where the main economic activities are related to forestry, agriculture and farming. However, Villarrica shows an inverse trend (figure 2.4), with slightly more coniferous alien tree cover at higher elevations. Here, lower, easily accessible areas in Villarrica are more restricted to recreational and touristic uses, relegating forestry to more remote or higher locations.

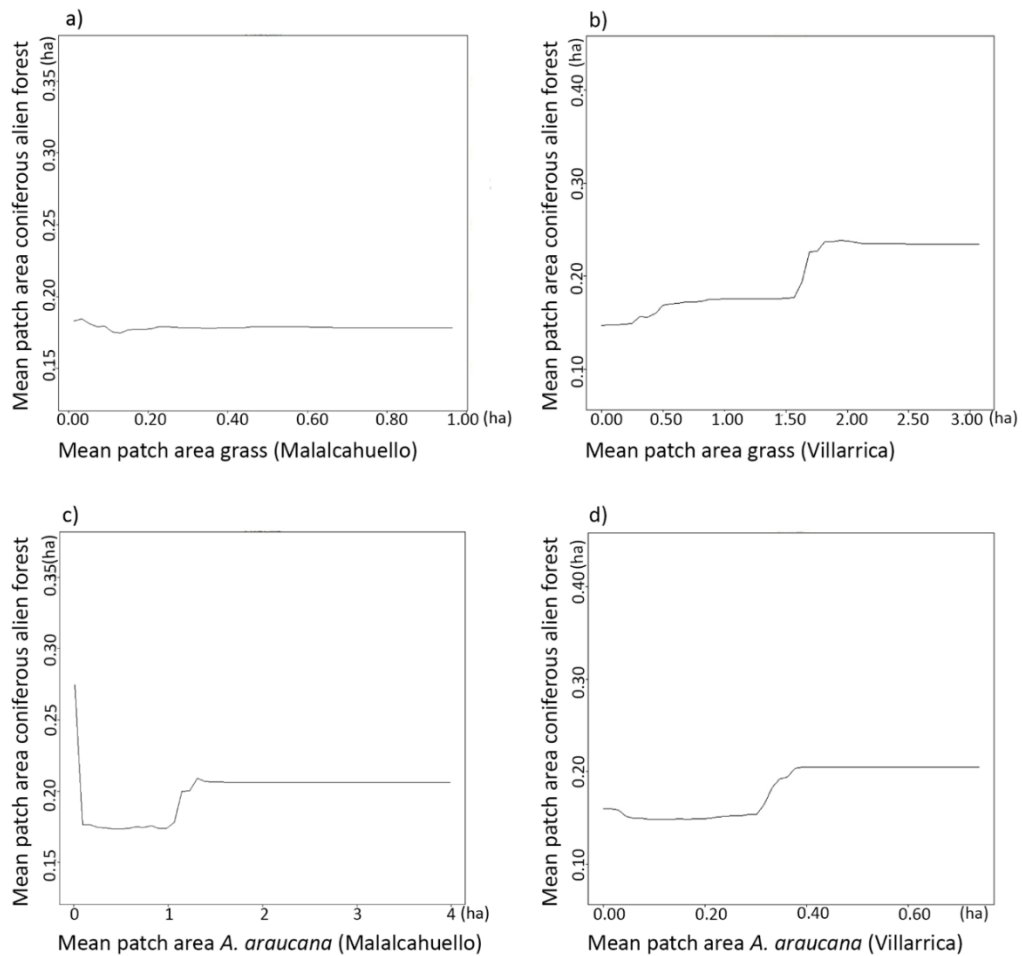
Several metrics related to grassland cover were also among the top half ranked importance variables in both models. However, the partial dependence plots did not exhibit a particular trend. In almost all of them, the area of alien pine remained constant with an increasing grassland cover, as shown in Malalcahuello in figure 2.5.



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 (Peña et al., 2008, Pauchard et al., 2014, 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 (figure 2.5). 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).



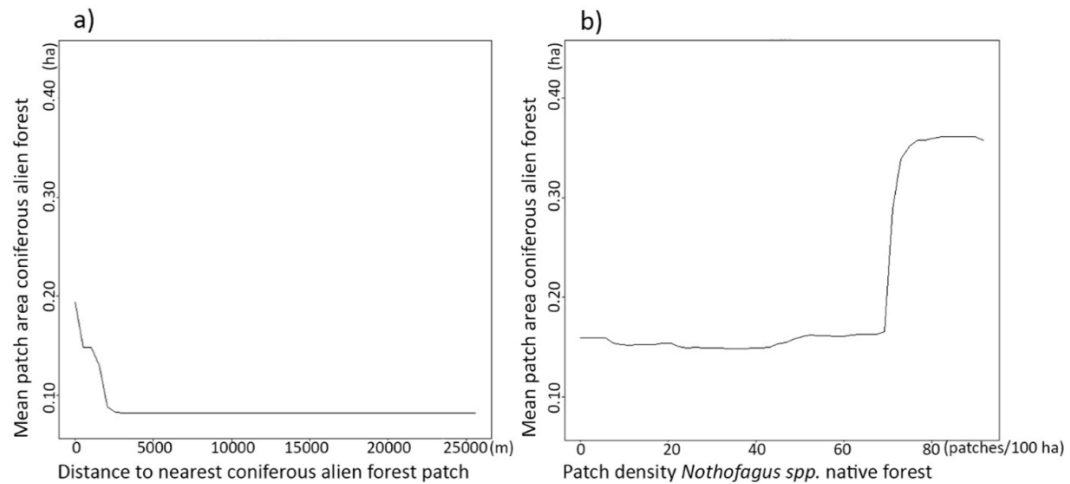
**Figure 2.4** Partial dependence plots of variables in the Malalcahuello and Villarrica models.



**Figure 2.5** Partial dependence plots of variables in the Malalcahuello and Villarrica models.

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 figure 2.6. Here, the traditional and generally accepted hypothesis that distance to seed sources is of great importance for invasion (Richardson et al., 2000, Giorgis et al., 2011, 2016, Altamirano et al., 2016, Arellano-Cataldo and Smith-Ramírez, 2016, Pauchard et al., 2016) 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 (figure 2.6) 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).



**Figure 2.6** Partial dependence plots of highly relevant variables in the Villarrica model.

#### 2.4. Discussion

The selected alien trees (*Eucalyptus* genus and Pinaceae family) were successfully mapped using Sentinel-2 moderate 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* (Peña et al., 2008, Pauchard et al., 2014).

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 m<sup>2</sup> 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, Kareiva and Neubert, 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 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, von Buch, 1965, Peña et al., 2008). 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 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 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 areas 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 (Richardson et al., 2000, Giorgis et al., 2016, Pauchard et al., 2016). 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 areas (Pauchard and Alaback, 2004). However, it is important to highlight that this method 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 elevation 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 elevations, 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, Quiroga and Gardner, 2013) and endemic tree. In its current situation, *A. araucana* could be outcompeted 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.

## 2.5. 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 method applied using freely available moderate 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 m<sup>2</sup>) 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, Kareiva and Neubert, 2004).

Within each study area, 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, Kareiva and Neubert, 2004) in the area. However, when comparing the two study areas, 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 elevations, the natural habitat of *A. araucana*. A reconsideration of the location of protected areas and the restriction of productive forest plantations to lower elevations, far from the *A. araucana* native forests, are recommended conservation measures to ensure the long-term continuity of the *A. araucana* populations.



**Chapter 3** Mapping the dynamics of alien plantation spread and native forest loss in Chile using Landsat time series analysis and Google Earth Engine

Prepared for submission to *Landscape Ecology*

## Abstract

Chile is a global biodiversity hotspot and hosts a large proportion of the southern hemisphere's temperate forests. Productive forest plantations involve deforestation of, and alien species introduction into, the Chilean Valdivian temperate forest ecosystem. This process has severely impacted the western part of Chile (the Coastal Range) and is starting to occur in the eastern part (the Andes), with forestry plantations being promoted by government subsidies since 1974. Archive Landsat satellite imagery and Google Earth Engine are used to assess land cover change over the last 40 years with a focus on alien species (*Pinaceae* and *Eucalyptus* spp.) spread and native deforestation. Plantations are highly dynamic, especially at lower elevations. 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 as a result of the end of forest subsidies in 2012. In the absence of subsidies, only large-scale plantations can engage in long rotation forestry, as smallholders look for a more stable income (provided by shorter rotation forestry). Most native forest is located at higher elevations. Although this is still relatively pristine, several abandoned alien forest plantations may be the source of biological invasion. In addition, native forest degradation as a result of *A. araucana* loss and shrub encroachment is occurring. Further investigation of this forest degradation and close monitoring of abandoned alien plantations are recommended to inform land management practices that ensure this area retains its conservation value in the future.

Key words: Biological invasions, temperate forests, change detection, Landsat, Google Earth Engine

### 3.1. Introduction

Chile is considered a global biodiversity hotspot (Myers et al., 2000) and hosts a large proportion of the temperate forests of the southern hemisphere. However, the rate of temperate forest loss in Chile has reached 4.5% per year in certain areas of the country such as the Coastal Range, which is already severely impacted by native deforestation, fragmentation and alien species introduction (Echeverría et al., 2006, Miranda et al., 2017). In addition, productive forest plantations using alien species have been steadily increasing

in Chile since the 1970s (Miranda et al., 2015) due to their high profitability, national forest policy and government subsidies encouraging plantations during the last four decades (Niklitschek, 2007, Altamirano et al., 2013).

The 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, Bone and Holzapfel, 2000, Altamirano and Lara, 2010). This, together with the continuous disturbances by productive forestry, pose a risk of biological invasion by alien trees in the ecosystem (Marvier, Kareiva and Neubert, 2004). This is concerning especially due to the singularity of its flora, of which the endemic tree *Araucaria araucana* (monkey puzzle) is the most relevant. This tree is classified as endangered in the IUCN Red List of Threatened Species (Premoli, Quiroga and Gardner, 2013) and is recognised as a Natural Monument by Chilean law due to its rarity and cultural value (Chilean Ministry of Agriculture, 1990).

The most widely used species in productive forest plantations in this area are trees from the Pinaceae family and the *Eucalyptus* genus. Both groups of species are generally highly productive, fast growing and shadow intolerant trees, characteristics that are shared by many invasive species (Dodet and Collet, 2012). Besides, the Pinaceae family has a high proportion of invasive species (12%) compared to other families (Richardson and Rejmánek, 2004). In addition, they have the ability to colonize and expand at rapid rates in almost any environment, as pollination is anemochorous and seed dispersal does not rely on mutualists (Richardson and Rejmánek, 2004).

Alien species introduction for forestry and potential biological invasion is a source of concern in Chile. Some forestry species grown in Chile are invasive or potentially invasive in the country (Bustamante and Castor, 1998, Teillier, Rodríguez and Serra, 2003, Echeverría et al., 2007). There is evidence of the spread of propagules from commercial plantations to the native forest (Peña et al., 2008), threatening the regeneration of *A. araucana* due to competition from alien saplings (Pauchard et al., 2014). The effective management of alien species needs to be rapid and targeted (Pauchard et al., 2016). A remote sensing approach is particularly appropriate to perform continuous alien tree monitoring in the Chilean forests, as the scale and abrupt topography makes ground-based techniques unfeasible. Landsat satellite imagery is especially useful to evaluate the spread of alien plantations and native forest loss, as its historical imagery archive reaches back as far as the 1970s for some locations (Wulder et al., 2016).

Previous remote sensing studies have assessed forest loss in the temperate forests of Chile using temporal series of land cover maps, but few of them try to 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 the studies by Echeverría et al. (2006) and Altamirano et al (2013). This study instead focuses on the Andean part of the country, which is still in a more pristine condition, is a relatively understudied part of the country, and has increasing presence of productive forest plantations which are spreading into the native forest in some areas (Peña et al., 2008, Langdon, Pauchard and Aguayo, 2010).

The recently released Google Earth Engine (GEE) cloud computing platform (Gorelick et al., 2017) can improve the results obtained by traditional remote sensing methods, as it allows users to perform resource intensive tasks using Google's computing capability free of cost. In addition, it can be very useful for conservation management, which is often hindered by lack of resources, as GEE eliminates the need for expensive computers or specialised software to perform computationally intensive image analyses. It enables using large scale cloud computing easily through an Internet available application programming interface and a web-based interactive development environment (Gorelick et al., 2017). GEE also includes a data catalogue of easily accessible, ready-to-use imagery, overcoming common data management complications (Gorelick et al., 2017).

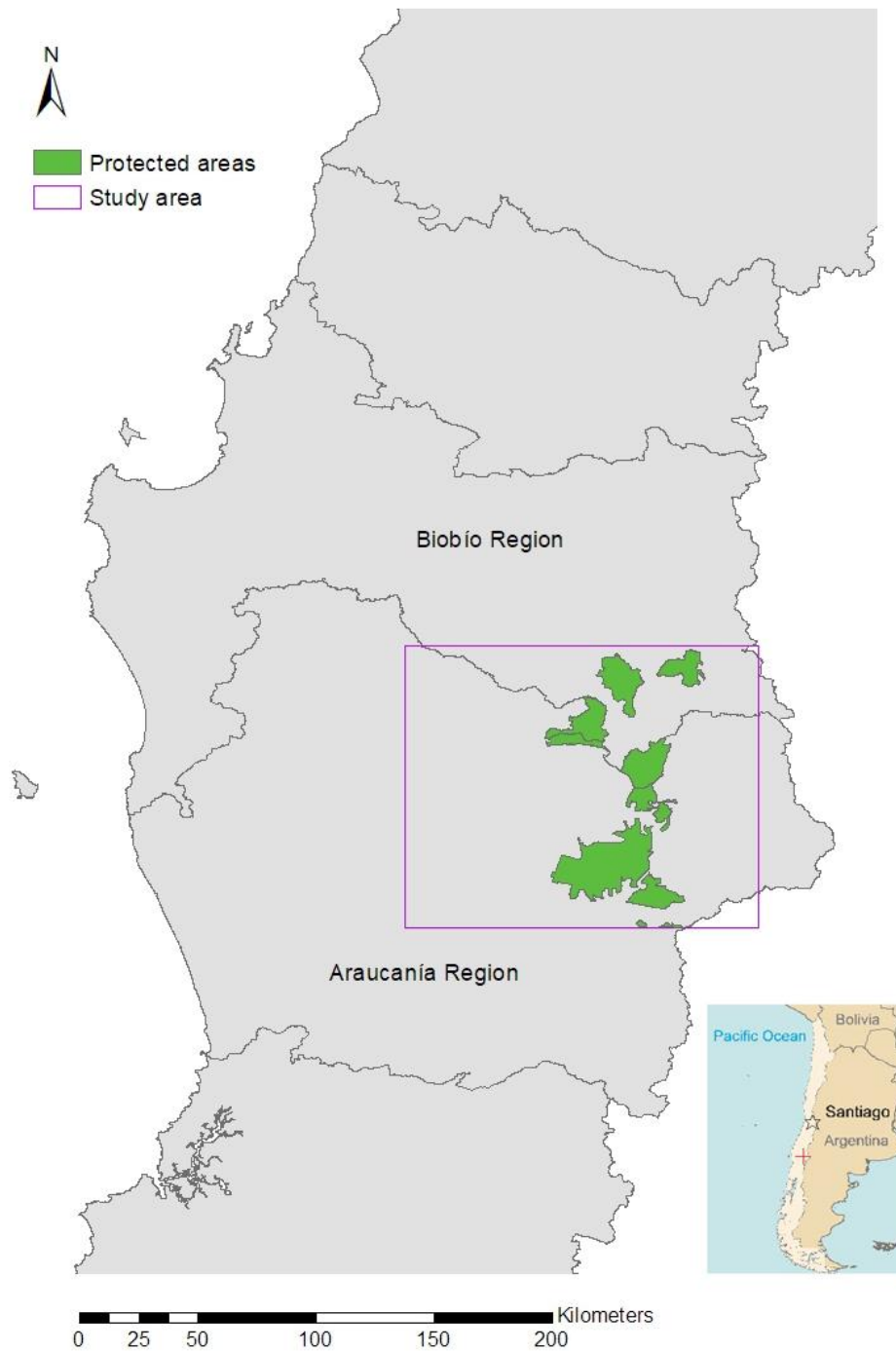
The overarching aim of this study is to investigate the dynamics of alien tree spread (potential invasion process) and native forest loss in a region of the Chilean Andes and Andean foothills over the last four decades. To achieve this aim, the following objectives are defined: 1) analyse the structural configuration of the landscape using land cover maps; 2) investigate the potential invasion process (spatial spread) of Pinaceae and *Eucalyptus* spp. and deforestation of the native forests using a change detection analysis, and; 3) compare and explain alien spread and forest loss processes.

## 3.2. Materials and methods

### 3.2.1. Study area

The study area (figure 3.1) is located in Chile, to the east of the Araucanía region (-38° latitude), in the south-central Andes. The area includes the Malalcahuello National Reserve, the Malalcahuello village and surrounds. It is a 14000 km<sup>2</sup> plot with elevations

ranging from 500 m to 2800 m. The temperate forests present in the area are composed mainly of trees of the *Nothofagus* genus and the endemic *Araucaria araucana*.



**Figure 3.1** Location of the study area. Data from the Chilean Library of National Congress (BCN) and the Geospatial Data Infrastructure (IDE) from the Chilean Ministry of National Goods (BCN, 2020, IDE Chile, 2020).

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 a nearby location in Araucanía's Coastal Range and Central Valley, 50% of the native forest loss between 1986 and 2008 was due to the establishment of new alien forest plantations (Altamirano et al., 2013). Another study on the Coastal Range in south-central Chile found the same conversion between the years 1975 and 2000 (Echeverría et al., 2006). Even though the Andean part of the country is less exploited for commercial forestry, its profitability, plus a promotional government policy that started in 1974, has encouraged the expansion of alien forest plantations everywhere in Chile. Between 1969 and 1970, more than 10 alien coniferous species were introduced in the Malalcahuello National Reserve (Peña et al., 2008). *Pinus contorta*, an alien species typically used in productive forest plantations, has been reported to be spreading into the native forest in Malalcahuello (Peña et al., 2008).

For the purpose of this study, 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 containing only protected areas was also created. The elevation of 600 m was chosen as the dividing line since this is the minimum elevation at which *A. araucana* generally grows (Premoli, Quiroga and Gardner, 2013). Most anthropogenic activity is concentrated in the valley area, where the major urbanised and agricultural areas are located.

### 3.2.2. Study species

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

The alien study species selected for this study are several members of the Pinaceae family and the *Eucalyptus* genus that are commonly used in productive forestry in Chile. Invasion is highly dependent on the ecology and growth form of the alien plants and their

interactions with the structure and the characteristics of the ecosystem (Giorgis et al., 2016). Focusing on functional groups rather than individual species is appropriate for this study, as each group (Pinaceae and *Eucalyptus* spp.) have similar ecological characteristics.

The Pinaceae group mainly comprises *Pinus radiata*, *Pinus contorta*, *Pinus ponderosa*, *Pinus sylvestris* and *Pseudotsuga menziesii*. All of these species except *P. menziesii* are included in the Global Invasive Species Database elaborated by the International Union for Conservation of Nature (IUCN) (IUCN Invasive Species Specialist Group, 2006). Although not included in the database, *P. menziesii* exhibits a series of traits, such as small seeds and short juvenile period, that are shared by other invasive trees (Dodet and Collet, 2012) and has been reported to be invading in certain areas (Simberloff, Relva and Nuñez, 2002). In addition, it is considered to be naturalized in south-central Chile (Pauchard, Langdon and Peña, 2008) and invasive in certain areas of the country (Pauchard, Langdon and Peña, 2008, Peña et al., 2008, Pauchard et al., 2014).

Among Chile's Pinaceae species, *P. radiata* and *P. contorta* are the 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). It has been reported as invading native *Nothofagus alessandrii* Espinosa forests in parts of central Chile (Bustamante and Castor, 1998), disturbing the hydrological regime in the south-central part of the country (Lara et al., 2009, Little et al., 2009). *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). In addition, it is a competitor of the endangered *Araucaria araucana* (Gundale et al., 2014, Pauchard et al., 2014) and its presence is correlated with that of the natives *A. araucana* and *Nothofagus antarctica* which form a medium to low density forest at high elevations. Most *Pinus* spp. are shade-intolerant, hence posing a similar threat to native open forests such as the high mountain *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 *Eucalyptus nitens*. These species are not formally listed as invasive but have some characteristics that are common among invasive plants, such as fast growth and low shadow tolerance (Dodet and Collet, 2012). Besides, 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).

### 3.2.3. Data sources

This study utilises the open access historical archive of Landsat satellite imagery which reaches back as far as the 1970s for some locations (Wulder et al., 2016), though imagery were not available for this study area from the 1970s. This historical archive is hosted in the GEE's image catalogue. Within this catalogue, related images, like those collected by the same sensor, are grouped together into image *collections* (Gorelick et al., 2017). This allows for easy filtering and sorting of the millions of individual images in the catalogue (Gorelick et al., 2017). In addition, the GEE cloud computing platform enables users to work with entire image collections, rather than individual images. Using collections instead of individual images provides more spectral information and increases the likelihood of generating a cloud free composite through image collection reducers. Image collection reducers use summary statistics such as the median to transform a collection into a single summary image. These summary statistics are calculated on a per pixel basis using all the images in the collection. A series of Landsat image collections spanning four decades, from the 1980s to the 2010s, were 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 TM (Thematic Mapper) sensor, and the 2010s collection was captured by the Landsat 8 OLI (Operational Land Imager) and TIRS (Thermal Infrared Sensor) instruments. Both Landsat 5 and 8 produce multispectral data at 30 m spatial resolution, so are directly comparable. The thermal band of Landsat 5 has 120 m pixels, and the two TIRS bands of Landsat 8 have 100 m pixels, but they have been resampled to 30 m. Table 3.1 summarises the spectral bands of these sensors. A preliminary classification exercise was conducted using the Landsat 8 imagery with different band combinations, one including the TIRS and coastal bands, and the other without them. Including the TIRS and coastal bands produced slightly stronger classification results. Hence, these bands were included in all the classifications performed in this study.

**Table 3.1** Spectral bands of Landsat 5 and 8 used in this study (NASA, 2020).

	<b>Landsat 5 (TM)</b>	<b>Landsat 8 (OLI + TIRS)</b>
<b>Band</b>	<b>Wavelength (<math>\mu\text{m}</math>)</b>	<b>Wavelength (<math>\mu\text{m}</math>)</b>
Coastal	-	0.44-0.45
Blue	0.45-0.52	0.45-0.51
Green	0.52-0.60	0.53-0.59
Red	0.63-0.69	0.64-0.67
NIR	0.76-0.90	0.85-0.88
SWIR 1	1.55-1.75	1.57-1.65



SWIR 2	2.08-2.35	2.11-2.29
TIRS 1	10.40-12.50	10.60-11.19
TIRS 2	-	11.50-12.51

Within each decade, targeted years 1986/87, 1998/99, 2006/07 and 2016/17 were selected, and imagery collections were generated for both summer and winter seasons (to characterise seasonal differences in landscape characteristics) for each period. These specific dates were chosen to make the dates of the last decade coincide with those used in Martin-Gallego et al. (2020), as this earlier study uses focuses on the same region in Malalcahuello. Thus, the winter and summer seasons of 2016-2017 were used for the last decade (2010s), to match the years used in Martin-Gallego et al. (2020). The rest of the datasets were chosen every 10 years from 2016-2017 decade (2006-2007, 1996-1997, 1986-1987) For each season, imagery over 4 months was collected. Earlier collections (1980s, 1990s) used images acquired over a wider time period due to the lower number of images acquired during these decades (Table 3.2). In addition, there was a lack of cloud free images for the years 1996 and 1997, so 1998/99 was the period chosen to represent the 1990s decade. These image collections were retrieved and pre-processed (see section 2.4) using GEE (Gorelick et al., 2017).

All images used were tier 1 surface reflectance data, which are the highest level Landsat data product provided by the Earth Resources Observation and Science (EROS) centre of the United States Geological Survey (USGS). Surface reflectance data are already atmospherically corrected (USGS, 2019a, 2019b). Tier 1 data reach the highest radiometric and geometric quality standard, enabling time series comparison, even among different Landsat sensors (USGS, 2019c). Table 3.2 shows a summary of the image collections used in this study.

**Table 3.2** Landsat data collections compiled from the Landsat archive using Google Earth Engine.

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

Landsat 8 (OLI + TIRS)		Summer	01/11/2016 – 28/02/2017	29
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The Landsat imagery collections form the principle data source for generating a series of land cover classifications to chart the change in land cover distributions between the 1980s and 2010s. However, elevation data from NASA’s Shuttle Radar Topography Mission (SRTM) was also used as an additional ancillary data source in the classifications as the tree species distributions display an altitudinal pattern. The large altitudinal gradient observed across the study area is expected to have an impact on landscape configuration and plant species distribution. Specifically, the 30 m resolution SRTM V3 digital elevation model (DEM) (Farr et al., 2007) was used.

Various sources of data were used for training and validating the land cover classifications. Field data were collected during two land cover survey campaigns performed in June-July 2017 and January-February 2018. A total of 146 points (locations) were visited. At each point, the following information was collected: land cover, qualitative description of the site as observed from the point; list of the tree and shrub species (both native and alien) that were present in a 5 m radius circular plot from the point; cardinal direction photographs (taken 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.

To complement the field data, other 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 vegetation cadastre created by the Chilean National Forest Corporation (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) VHR WorldView-2 satellite imagery, plus other VHR imagery accessible through Google Earth (Heilmayr et al., 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 metadata.

The 2010s decade 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, which dates from 2007 and 2008. Reference data for the earlier decades was extracted from historic VHR imagery via Google Earth, previous versions of the CONAF forest survey and present day field observation of mature forest (i.e. a mature undisturbed native forest present in the 2010s must have been present in earlier decades).

#### 3.2.4. Image pre-processing

To minimize the topographical shadows that can negatively impact the quality of the land cover classification, each image within each 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 within a collection was also cloud masked in GEE using the pixel quality assessment band (pixel\_qa) provided with the Landsat surface reflectance data products. This band provides information about each pixel's usefulness in relation to its cloud, cloud shadow, snow and ice presence (USGS, 2019b). The pixel quality assessment band can sometimes be inaccurate when the temperature differential between the cloud and the Earth's surface is either too large or too small; when there are very bright land covers such as built-up areas, sand, snow or ice; and when the clouds are too thin (USGS, 2019a, 2019b). If there was remaining cloud due to these inaccuracies, it was removed in subsequent steps of the method using temporal aggregation and manual cloud masking.

Then, temporal aggregation was performed to remove remaining cloud cover (Carrasco et al., 2019) and reduce noise in the classification. In this process, each image collection is summarised as a single image using the median value of all the images of the collection in each pixel. If the collection is large enough and there are clouds remaining in less than half of the images for a certain pixel, the median eliminates clouds. However, if there are clouds present in more than half of the images for a certain pixel, the cloud will be shown in the output median composite image. Since some of the winter collections used in this study had few images and very persistent cloudy conditions that were not identified in the pixel quality assessment band, a cloud mask was applied manually. Although some cloud was still present in the median composite images, the area of useable data is considerably increased using this method.

Finally, a two-season composite image was created for each decade. This composite included all bands belonging to the winter and summer median images and a DEM. The final Landsat 5 composites (1980s, 1990s and 2000s) had 15 bands (14 spectral bands and the elevation). The Landsat 8 composite (2010s) had 19 bands (18 spectral bands plus elevation), as the cirrus and panchromatic bands were removed from further assessment.

### 3.2.5. Land cover classification

Land cover classification was performed on the 1980s, 1990s, 2000s and 2010s composite images to assess how land cover in the study area has changed between the last four decades. A supervised random forest approach was used in GEE. Random forests are a machine learning algorithm (Breiman, 2001) that is 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 the data sources section. On average, 40 training polygons (roughly 2800 m<sup>2</sup> each) were created for each class, except for the classes that were more spatially constrained in the study area, such as permanent snow, bare soil, urban or *Eucalyptus* spp., which had around 20 training polygons each.

Table 3.3 shows the classification nomenclature used in this study. This classification system offers separation of the different types of alien and native forests, comprising 10 land cover classes, including six different vegetation classes. This system was created using the FAO (Food and Agriculture Organization) land cover classification system (Di Gregorio and Jansen, 2005), and the classification scheme used in the 2014 Chile land cover map (Zhao et al., 2016) as initial guidelines, with adaptations for the purpose of this study.

**Table 3.3** Land cover class nomenclature. (Forest class abbreviations (in brackets) are used to denote these classes in other parts of this paper.)

Land cover class		Class description
Alien forests	Coniferous plantations (CP)	Pinaceae family.
	Broadleaved plantations (BP)	<i>Eucalyptus</i> genus ( <i>E. globulus</i> and <i>E. nitens</i> ).
Native forests	<i>Nothofagus</i> spp. (broadleaved) forests (NNF)	<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.).
	<i>Araucaria araucana</i> (mixed coniferous-broadleaved) forest (ANF)	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 pillopillo</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>Poa nemoralis</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.

### 3.2.6. Land cover change analysis

The first change analysis step was simply to calculate the area of each land cover in each subset of the study area (valley, Andes, protected areas) to assess the variation between the different periods. Then, trajectories of change between land cover classes were investigated through change detection analysis, specifically post-classification comparison. Each pair of land cover classifications was overlaid and the variation between dates calculated on a per pixel basis. For this, an arbitrary value was assigned to each class (appendix E), and then the images were subtracted on a per pixel basis. The image pairs compared were the overall change between the first and the last period (1980s-2010s), and each decadal-scale change (1980s-1990s, 1990s-2000s and 2000s-2010s).

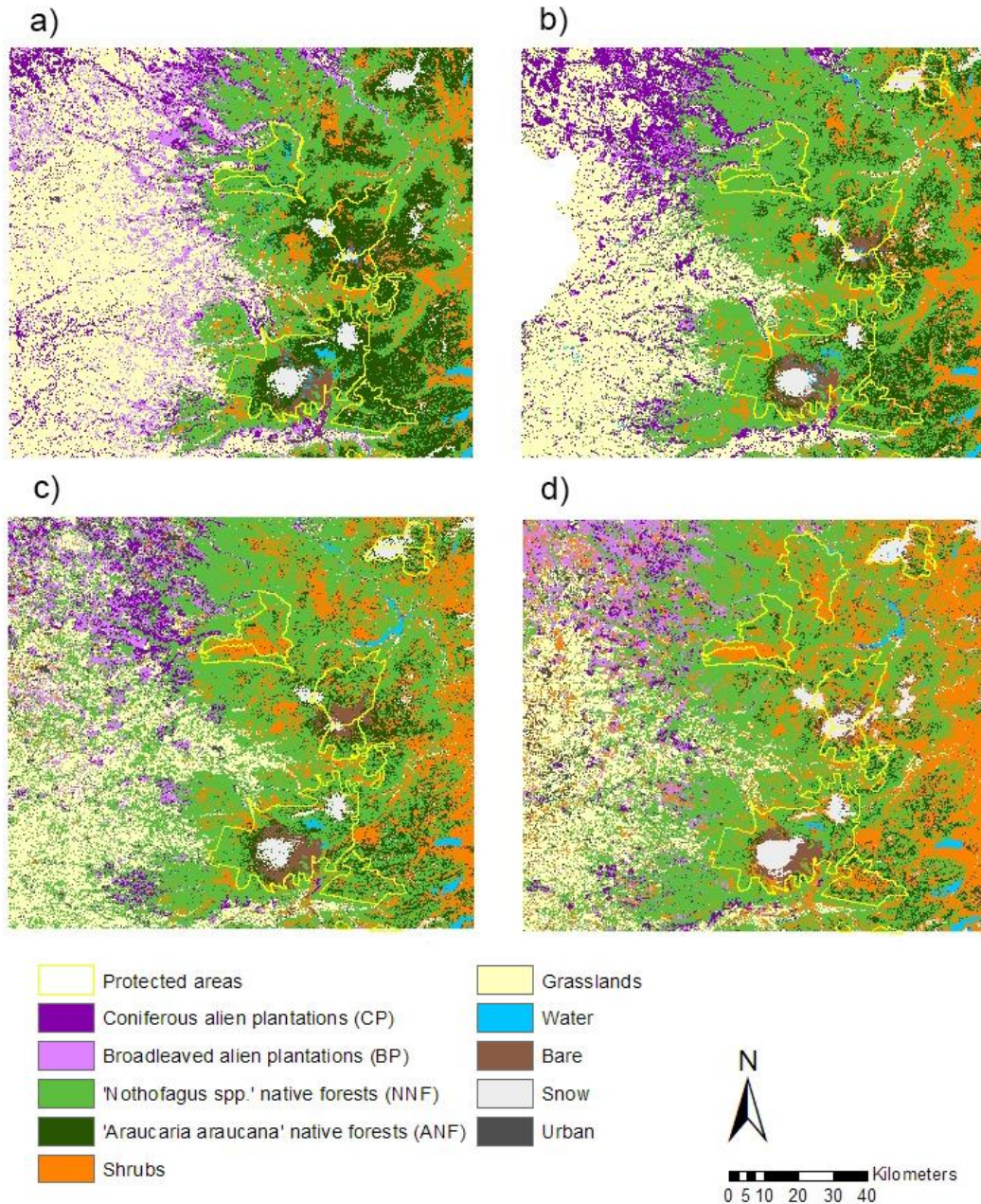
Detailed examination of the change trajectories involved the construction of a change detection matrix for each pair and each study area subset. Since the protected areas have changed over the 30 year span of the study, the land cover area inside protected areas was calculated using only the protected areas extent in the earlier of the pair of classifications being compared to ensure that the protected area was indeed protected for the entire time-period. Finally, maps of change were created for the vegetated land cover classes. These maps show specifically where alien and native forest gain and loss have occurred. A simplified classification system was used for the change detection matrix and change maps

to limit error that could arise through confusion between forest sub-classes. Thus, the vegetated land cover classes used for this part are alien forest plantations (including both coniferous and broadleaved), native woody species (encompassing *A. araucana* and *Nothofagus* spp. native forests and shrublands) and grasslands.

### 3.3. Results

#### 3.3.1. Land cover classifications

Figure 3.2 shows the land cover maps generated for the four time periods. A clear differential distribution of land covers is observed in all decades: most plantations are located to the west of the study area, where the valley (lower elevation) is located. In contrast, the Andes (on the east) contain the protected areas and have a dominance of native vegetated classes. This is unsurprising, as most urban settlements are located in the valley. Most agriculture and silviculture also take place in the valley which is more accessible due to the lower elevations, the absence of steep slopes and the proximity to urban areas.



**Figure 3.2** Land cover maps in a) 1980s decade, b) 1990s decade, c) 2000s decade, and d) 2010s decade.

Table 3.4 shows the classification accuracies for the four decades. The full confusion matrices can be found in appendix F. Generally, alien plantations are well separated from native forested areas. There is some confusion between the two alien plantation classes. Also, shrubs, *Nothofagus spp.* and *Araucaria araucana* forests are sometimes confused with each other. This is not surprising, as *A. araucana* forests are rarely monospecific, having *Nothofagus spp.* trees intermixed and, at high elevations, they appear as open forest with an understorey of shrubs. In addition, 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 separate at 30 m resolution. For this reason, the change detection analysis was performed using a simplified classification system that collapses coniferous and broadleaved plantations together in the *alien* class, and the three native woody species classes in the *native woody species* class.

**Table 3.4** Classification accuracies of the different composites. (CP = coniferous plantations, BP = broadleaved plantations, NNF = nothofagus native forests, ANF = araucaria native forests, alien = alien plantations, native = native woody species, PA = producer's accuracy, UA = user's accuracy, OA = overall accuracy.)

		1980s		1990s		2000s		2010s	
		PA	UA	PA	UA	PA	UA	PA	UA
Land cover classes	CP	0.77	0.80	0.87	0.83	0.78	0.89	0.83	0.78
	BP	0.89	0.47	0.62	0.75	0.89	0.80	0.76	0.9
	NNF	0.98	0.93	0.95	0.69	0.95	0.63	0.98	0.57
	ANF	0.93	0.78	0.62	0.79	0.79	0.87	0.61	0.98
	Shrubs	0.82	0.95	0.67	0.72	0.78	0.81	0.80	0.71
	Grass	0.93	0.9	0.69	0.74	0.67	1	0.84	1
	Water	0.94	0.89	1	0.96	1	1	0.96	1
	Bare	0.66	0.96	0.84	1	0.93	0.96	0.93	0.96
	Snow	1	0.92	1	1	1	1	1	1
	Urban	0.63	1	0.91	0.67	0.95	0.95	0.88	1
	OA	0.86		0.80		0.85		0.85	
Simplified land cover classes	Alien	0.95	0.81	0.93	0.97	0.97	0.99	0.89	0.93
	Native	0.98	0.94	0.96	0.92	0.98	0.86	1	0.90
	Grass	0.93	0.90	0.70	0.74	0.67	1	0.84	1
	Water	0.94	0.90	1	0.96	1	1	0.96	1
	Bare	0.66	0.96	0.84	1	0.93	0.96	0.93	0.96
	Snow	1	0.92	1	1	1	1	1	1



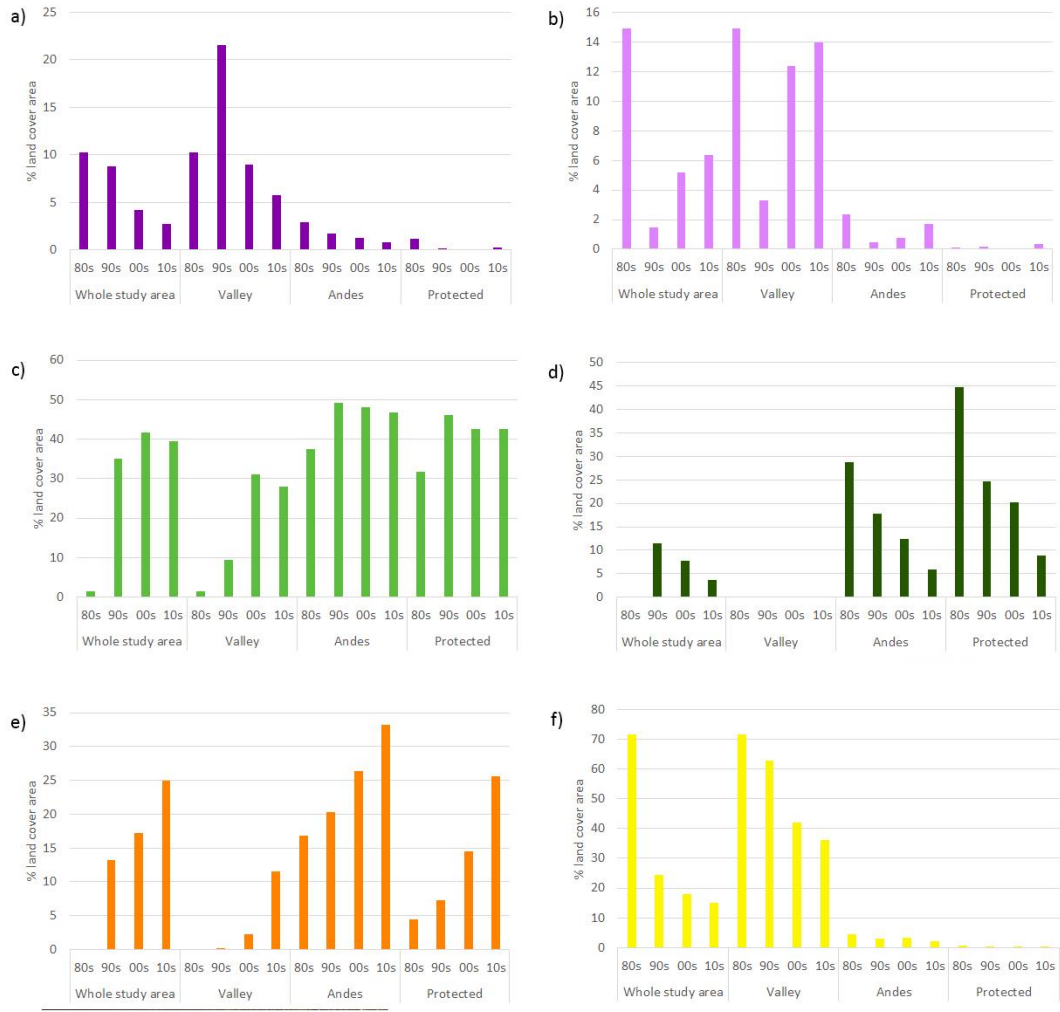
	Urban	0.63	1	0.91	0.67	0.95	0.95	0.88	1
	OA	0.91		0.92		0.94		0.94	

Figure 3.3 presents the percentages of vegetated land cover areas over the study period. In general, coniferous alien plantations have been steadily decreasing since the 1980s. Broadleaved alien plantations, on the other hand, experienced a sharp decrease in the first period (1980s-1990s) followed by an increasing trend until the 2010s. 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 years, being often harvested at 15 years (Riesco Muñoz, 2007, Salas et al., 2016) and Pinaceae rotations range between 18 and 35 years (Cantero, Espinel and Sáenz, 1995, Salas et al., 2016), although these rotations will vary depending on the environmental conditions, the site and the specific species. These species are fast growing, and most of the plantations observed in the 1980s image could have been established during the 1970s. This way, their rotation periods explain the decreases in *Eucalyptus* spp. between the 1980s and the 1990s and the sharpest decrease in Pinaceae land cover, which occurred between the 1990s and the 2000s. In addition, the fact that forest subsidies ended in 2012 (Chilean Ministry of Agriculture, 1998) may have promoted land cover changes from productive forestry to other land uses, especially in the less profitable estates.

Native *Nothofagus* spp. forest is one of the most dominant land cover classes in all subsets of the landscape, but it shows different dynamics at higher elevations (Andes subset) and lower elevations (valley subset). The Andes and protected areas show a relatively stable amount of *Nothofagus* spp. forest. In the valley, *Nothofagus* spp. stands were a minority class in the 1980s and the 1990s. However, in the last two decades, they are the second most dominant class in the valley, only surpassed by grasslands, which have been on a constant decline since the beginning of the study period. These *Nothofagus* spp. forest stands in the valley are probably a result of forestry activities to produce wood for the local communities.

The endemic *Araucaria arucana* forests have decreased in extent in all time periods and subsets of the study area, despite having been declared a Natural Monument in 1976 and laws forbidding its felling in almost all cases. Since 1990, it can only be removed for authorised scientific, public works or national defence purposes (Chilean Ministry of

Agriculture, 1990). Accompanying the decrease in *A. araucana* forests, there has been an increase in shrublands. As mentioned before, high elevation *A. araucana* forests are generally open and have an understorey of shrubs. The increase in shrublands observed could be a result of confusion between the two classes, or an indicator of a decrease in *A. araucana* canopy cover.



**Figure 3.3** Overall vegetation land cover change over the study period.

### 3.3.2. Land cover change analysis

As mentioned in section 3.2.6., the land cover change analysis was performed using a simplified classification system to limit error. The classes used for this analysis were alien forest plantations (including both coniferous and broadleaved), native woody species (encompassing *A. araucana* and *Nothofagus* spp. native forests and shrublands) and grasslands.

Overall inter-class change between the first (1980s) and the last (2010s) periods is presented in table 3.5 as a change matrix. The full matrices including non-vegetated classes and displaying change in km<sup>2</sup> for all periods can be found in appendix G. Between the 1980s and the 2010s, most of the alien forest cover shifted to native woody species (1479.22 km<sup>2</sup>, 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). 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<sup>2</sup> 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<sup>2</sup> 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 1980s and the 2010s. An area of 8534.07 km<sup>2</sup> (48.68 % of the whole study area) was native forest in the 1980s and 2010s. 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 in the classification.

During the first period (1980s-1990s, table 3.5, figure 3.4) alien plantations remained (888.40 km<sup>2</sup>, 5.28%) or were converted to native forest (686.41 km<sup>2</sup>, 4.08%). This conversion from alien plantation to native woody species was the dominant trajectory in the Andes and within protected areas. Before the beginning of the large-scale Chilean forest industry, several test plantations were established in the country. More than 10 alien coniferous trees were introduced in the Malalcahuello National Reserve between 1969 and 1970 (Peña et al., 2008). These plantations are now abandoned and surrounded by a matrix of native forest. It is possible that some of these plantations were removed after the tests, or that some of them were gradually displaced by native forest covers. It is important to highlight that the first period was the one with more conversion from alien plantation to grassland. This change could indicate forest clearing for farming, or a result of clear-cut plantations at the end of the silvicultural rotation. Depending on the species, rotations can be as long as 18-35 years for Pinaceae (Cantero, Espinel and Sáenz, 1995, Salas et al., 2016), meaning that new plantations established during the 1990s would not be cut until the 2010s or 2020s. In fact, the 2000s-2010s period had the second highest rate of conversion from alien plantation to grassland. However, the length of Pinaceae rotations indicates that a longer time series would be required to gain more insight on the dynamics of forestry related land cover change.

In the second period (1990s-2000s, table 3.5, figure 3.4) most alien plantations remained (836.15 km<sup>2</sup>, 4.97% of the whole study area), although a large part (709.70 km<sup>2</sup>, 4.22%) changed to native forests. A large proportion of grasslands (1018.05 km<sup>2</sup>, 6.05%) were converted to native forests, especially in the valley subset. This confirms 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 first period, in the Andes and protected areas, alien plantations mainly changed to native forest cover.

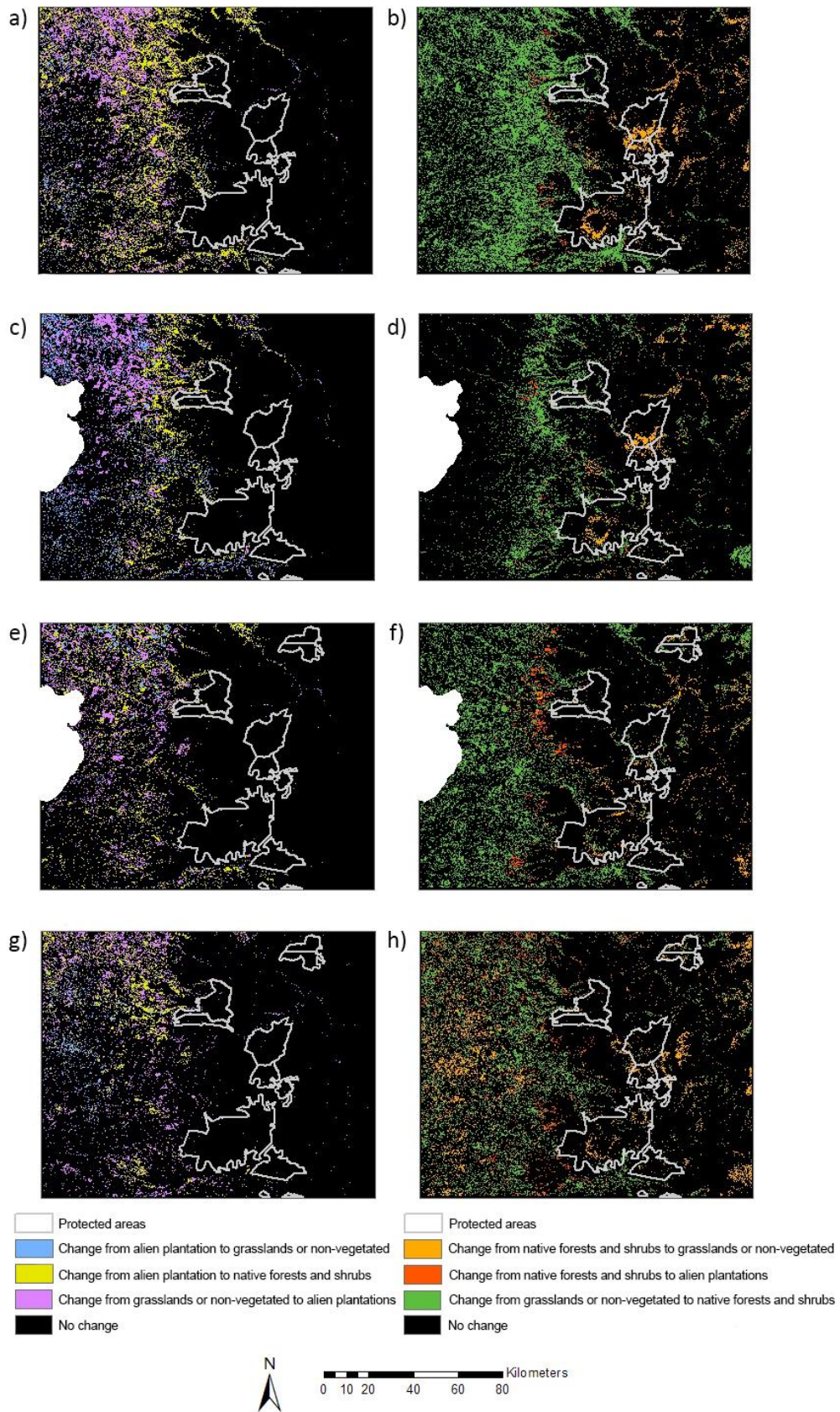
The last period (2000s-2010s, table 3.5, figure 3.4) showed that a large portion of alien plantations remained (871.23 km<sup>2</sup>, 4.97% of the whole study area). Part of them were converted to native forests, but this conversion was not as marked as in earlier decades. A larger proportion of native forests than in earlier decades shifted to grasslands, especially in the valley, where 6.62% of the area (442.54 km<sup>2</sup>) shifted from native forests to grasslands. This indicates that many native plantations have

been clear-cut in recent decades, suggesting again native species forestry activities. Also, the last decade showed more conversion from native forests to alien plantations in all subsets. This was expected in the valley as a result of conversions between different forest plantations, but it is a worrying discovery for the Andes and protected areas, as it could mean that there are more new alien plantations being established in the more pristine part of the study area or that there is invasion occurring.

**Table 3.5** Inter-class land cover change between the 1980s and the 2010s expressed as relative change in each class. The percentages are calculated with respect to the total area of each subset (whole study area, valley, Andes and protected areas). (AP = alien plantations, NWS = native woody species.)

		2010s land cover class (%)		
		AP	NWS	Grass
	1980s land cover class (%)			
Whole study area	Alien plantations	3.06	8.44	1.11
	Native woody species	0.57	48.68	0.53
	Grass	5.36	10.01	13.28
Valley	Alien plantations	6.39	15.46	2.69
	Native woody species	0.14	1.46	0.05
	Grass	12.93	21.81	32.91
Andes	Alien plantations	1.01	4.10	0.13
	Native woody species	0.84	77.81	0.82
	Grass	0.68	2.72	1.16
Protected areas	Alien plantations	0.12	1.12	0.01
	Native woody species	0.31	72.17	0.09
	Grass	0.03	0.56	0.12
		1990s land cover class (%)		
		AP	NWS	Grass
	1980s land cover class (%)			
Whole study area	Alien plantations	5.28	4.08	3.22
	Native woody species	0.39	51.86	0.51
	Grass	4.39	2.49	20.27
Valley	Alien plantations	12.44	5.14	8.37
	Native woody species	0.32	1.36	0.14
	Grass	11.72	2.96	53.49
Andes	Alien plantations	1.34	3.50	0.39
	Native woody species	0.43	79.67	0.70
	Grass	0.35	2.23	1.97
Protected areas	Alien plantations	0.12	1.10	0.02
	Native woody species	0.20	75.03	0.06
	Grass	0.01	0.55	0.16
		2000s land cover class (%)		
		AP	NWS	Grass
	1990s land cover class (%)			
	Alien plantations	4.97	4.22	0.56
	Native woody species	0.97	56.59	0.95

Whole study area	Grass	2.98	6.05	14.77
Valley	Alien plantations	12.44	10.01	1.34
	Native woody species	1.08	7.84	0.62
	Grass	8.04	15.48	38.18
Andes	Alien plantations	0.86	1.03	0.14
	Native woody species	0.90	83.43	1.14
	Grass	0.20	0.86	1.89
Protected areas	Alien plantations	0.03	0.27	0.01
	Native woody species	0.06	74.29	0.17
	Grass	0.01	0.09	0.12
		2010s land cover class (%)		
	2000s land cover class (%)	AP	NWS	Grass
Whole study area	Alien plantations	4.97	3.18	1.00
	Native woody species	2.35	59.17	2.92
	Grass	1.30	4.91	10.82
Valley	Alien plantations	11.42	6.83	2.52
	Native woody species	4.27	21.71	6.62
	Grass	3.01	10.40	26.07
Andes	Alien plantations	0.99	0.93	0.06
	Native woody species	1.17	82.29	0.63
	Grass	0.25	1.52	1.42
Protected areas	Alien plantations	0.04	0.05	0.00
	Native woody species	0.41	72.35	0.11
	Grass	0.01	0.19	0.10



**Figure 3.4** Forest change maps for a) dynamics of alien plantation loss and expansion 1980s-2010s, b) dynamics of native forest loss and expansion 1980s-2010s,



c) dynamics of alien plantation loss and expansion 1980s-1990s, d) dynamics of native forest loss and expansion 1980s-1990s, e) dynamics of alien plantation loss and expansion 1990s-2000s, f) dynamics of native forest loss and expansion 1990s-2000s, g) dynamics of alien plantation loss and expansion 2000s-2010s, and h) dynamics of native forest loss and expansion 2000s-2010s.

### 3.4. Discussion

This study sought to investigate the dynamics of alien tree spread and native forest loss over 40 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. 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).

The analysis of the structural configuration of the landscape (the first objective) showed, as expected, a clear spatial organisation of the landscape with two different subset areas based on their elevation: the Andes (at higher elevations) and the valley (at lower elevations). Logically, most human activity (and the land covers and uses related to this) is located in the valley, which is more accessible and contains the major urban settlements. A similar landscape structure was found by Martin-Gallego et al. (2020) in the same study area. Although this general landscape structure was noticeable in all decades, the results show that the alien Pinaceae and *Eucalyptus* spp. and native *Nothofagus* spp. forested classes are highly dynamic, being converted from one to another frequently over 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 *Eucalyptus* 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 wood and firewood for the local communities. Also, *Nothofagus* spp. is sometimes used for sustainable forestry. Rotation periods range between 18 and 35 years for Pinaceae, between 10 and 20 for *Eucalyptus* spp., and between 25 and 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 (1980s) and the last (2010s), 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. The high dynamism and long silvicultural rotation periods require longer time series to model alien spread and forest loss processes accurately. Future work should expand this study by monitoring forest change in upcoming decades. Although the Landsat imagery archive is a valuable tool to analyze the dynamics of conversion among alien and native forests and shrubs, as demonstrated in this study, future work could benefit from other imagery such as Sentinel-2, as its higher in spatial resolution and spectral capabilities could improve forest land cover separation in future studies.

Government subsidies for productive forestry started in the 1970s and ended in 2012. This has likely driven smallholders to shift from Pinaceae to species with a shorter rotation period such as *Eucalyptus* spp. and from industrial forestry (Pinaceae and *Eucalyptus* spp.) to sustainable systems or short rotation forestry destined for a more local market (*Nothofagus* spp.). This interpretation is further confirmed by the changes observed in the last period (2000s-2010s), where less than 5% of native forest has changed to alien forest cover in the valley. However, it is important to remark that native deforestation in favour of productive forestry is still occurring in the area. The rate of conversion in the last period is lower than the rates observed in the first period, although this has increased compared to the 1990s-2000s period. This is especially concerning in the Andes, where change from native to alien species has also increased in the last period.

Forest subsidies kick-started the forestry industry in Chile and maintained it as widespread practice for almost four decades, but their end in 2012 marked the start of a new transition period in the forestry sector (and the landscape). The constant decrease in grassland cover and the increase of forest in the valley suggests that the region has specialized in forestry, with farming and agriculture activities being displaced to other areas of the country. However, in the next decades with the end of forest subsidies, smallholders whose income depends on a small area of land will have to shift from long rotation wood forestry to other activities that are more consistently profitable in the short-term. These will include short rotation forestry (such as *Eucalyptus* spp. or *Nothofagus* spp.) or other, radically different activities such as sustainable forestry or tourism. Thus, the end of subsidized forestry will stabilise the market (Matsumoto, 1998, Femenia, 2015), with the less profitable

estates (due to their size or inaccessibility) naturally shifting from industrial monoculture forestry to other uses. This phenomenon, however, will not affect extensive plantations owned by large, highly profitable forestry companies, which could potentially keep deforesting the native forest (there is a rise in native to alien conversion within the Andes in the last period). In the absence of subsidies, the landscape will diversify as smallholders change their productive activities and Pinaceae productive plantations will become larger in general. Future monitoring in the upcoming decades will be essential to confirm if this is the case and the alien species introduction for forestry slows down, thus reducing the number of propagule foci and pressure on the native forest and decreasing the risk of biological invasion. This future work could benefit from other satellite sensors such as Sentinel-2, which does not have a historical archive as long as the Landsat mission, but has finer spatial resolution and the potential to provide more accurate separation between vegetated land covers due to its spectral capabilities (Martin-Gallego et al., 2020).

The change detection analysis (the second objective), which aimed to assess the spread or invasion process of Pinaceae and *Eucalyptus* spp. and the deforestation of the native forests, has shown that, overall between the 1980s and the 2010s, there has not been a net spread of alien forest cover at the expense of native forest in the area. However, conversion from native forest to alien has been accelerating through the decades. This suggests that forest is a highly dynamic class as a result of intense silvicultural activity, especially in the valley area. Comparing the first decade (1980s) with the last (2010s) conceals this dynamism. Similar results have been reported earlier in other parts of Chile by Altamirano et al. (2013). Besides, this method does not differentiate between undisturbed native forests and native plantations. A loss of undisturbed native forest could be concealed by native plantations established later, as both would be classified as the same native forest class. Being able to separate these two types of native forest would provide a more accurate measure of forest loss. Future work should aim to separate undisturbed natural forests from plantation forests, potentially using VHR or drone based imagery and texture or shape measures to enhance separation of undisturbed native forest from native plantations.

These trends confirm that the end of forest subsidies has triggered a change in forestry practices. Even though the comparison of alien spread and forest loss does not show a clear link between alien forest cover expansion and native forest

deforestation (the third objective) at Landsat's resolution, it is important to remark that roughly 110 km<sup>2</sup> of the alien forest stands that were present in the Andes in the 1980s still remain today. These are probably abandoned plantations, as some test plantations were introduced in the early 1970s (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). This monitoring could use other, higher resolution imagery such as from VHR satellite or drone-based imagery to detect invasive patches.

A concerning result of this study was the rise in shrub cover in the Andes and within protected areas. This could be a result of confusion in the classification, as the shrubs land cover class is sometimes confused with the native forest classes. Shrubs are confused with *Nothofagus* spp., especially with young early successional forests, because these forests 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 are thus also potentially confused with the shrubland class. Despite these sources of confusion between shrubs and native forests, the rise in shrub cover is still concerning, as the more recent classifications are the most accurate and also the ones that show a larger proportion of shrubs. Besides, conversion from native forest to shrub has been documented before in nearby areas (Zamorano-Elgueta et al., 2015, Miranda et al., 2017). This, together with the decrease of *A. araucana* forest in all decades, is a sign of forest degradation that requires further investigation, as it can occur either as a result of disturbances, or because of changing environmental conditions such as those brought about by climate change. In addition, shrub encroachment in the naturally open *A. araucana* forest could prevent its regeneration, jeopardizing the conservation policies that have been put in place for this endemic, culturally relevant tree.

It is important to mention that climate change has likely influenced land cover change in the area over the last decades. Increasing temperatures and decreasing

rainfall have been occurring in the area and are projected to occur in the future (Kitzberger et al., 2016, González et al., 2018, Urrutia-Jalabert et al., 2018). This has already had a negative impact on the productivity of rainfed crops and grasslands (Garreaud et al., 2017). This is corroborated by the results obtained in the 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).

The degradation of the *A. araucana* forest could also be influenced by climate change, which could be further reducing the already limited high mountain habitat of *A. araucana*. In addition, increasing temperatures will allow alien trees to colonise higher elevations, entering in direct competition with *A. araucana*. Further, a warmer and drier climate will increase the frequency of forest fires (González, Veblen and Sibold, 2005, González et al., 2018). 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, which are expanding, 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 (especially native 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). 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).

Other ecosystem services such as hydrological regulation, erosion control, carbon capture and storage, and provision of recreation and cultural value are also impacted by a highly dynamic forest cover. The type of forest present in a catchment impacts the water cycle. A fragmented and constantly changing forest cover will modify the water supply ecosystem service and other recreational and cultural ecosystem services related to water such as fishing. A study in the Valdivian Coastal Range and the Valdivian Andes, two regions close to the study areas, found a positive correlation between native forest presence and runoff coefficient during the dry season. The opposite was observed for *Pinus radiata* and *Eucalyptus* spp. plantations, as these species have a high transpiration rate (Lara et al., 2009). *Eucalyptus* spp. plantations reduce runoff by 75%, and *Pinus* spp. plantations, by 40% (Farley, Jobbágy and Jackson, 2005). A decreased forest coverage or forest stand density in an early successional native forest increased streamflow in the Chilean Valdivian Coastal Range and Valdivian Andes (Lara et al., 2009). These modifications of the hydrological regime alter water habitats (turbidity, oxygen content, invertebrate population, temperature, etc.), which in turn affect the abundance of fish. A positive correlation between trout abundance and native forest cover in the catchment has been reported in the Chilean Valdivian Coastal Range and Valdivian Andes (Lara et al., 2009). Productive forestry, then, negatively impacts the recreational use of rivers and in general has lower economic and recreation potential for tourism (Nahuelhual et al., 2013). In addition, industrial forest plantations subject to clearcutting promote soil loss (Oyarzun and Peña, 1995). Erosion is here a consequence of the unsustainable management practices which leave the soil bare for long periods of time. The high forest dynamism found in this study, with frequent changes in type of tree cover, is probably contributing to erosion in the area. Finally, although plantations have a high rate of carbon sequestration due to their quick growth

(Mukul, Herbohn and Firn, 2016), their increase in Chile did not increase the carbon pool in forest biomass (Heilmayr, Echeverría and Lambin, 2020). This is because more diverse systems such as undisturbed native forest tend to have a larger carbon stock (Matos et al., 2020).

Consequently, from a management perspective, it is of utmost importance to preserve native forest patches with a high structural complexity and species richness, as they support specific biodiversity, act as seed sources for regenerating native forests nearby, and provide a wider range of ecosystem services. It is also key to preserve early successional native forests and allow them to develop into a more complex system. 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.

### 3.5. Conclusion

This study has demonstrated that continuous monitoring of forests is crucial for sustainable land planning, as forests, both undisturbed natural forests and plantation forests, are a highly dynamic class. This high dynamism is not unique to this part of Chile, as similar results have been found in other parts of the country (Altamirano et al., 2013). This dynamism, together with the long silvicultural rotation periods require long time series to model the alien 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 the native forest. Remote sensing methods, such as the one used in this study, allow continuous monitoring of forest dynamics in extensive and inaccessible areas such as the Andes in a timely and cost-effective manner. This is essential in countries such as Chile, which hosts a large amount of threatened biodiversity but suffers a lack of funding for conservation (Waldron et al., 2013).

There is a clear separation by elevation in the landscape of the Malalcahuello area in all decades, as reported in earlier studies (Martin-Gallego et al., 2020). The lower elevation subset (the valley) is a highly disturbed and dynamic area and hosts most productive forestry and other anthropogenic land uses. The higher elevation subset (the Andes), is still relatively untouched, with native forests as the dominant land cover. Although right now alien forest stands are located mostly in the valley area, the presence of a few abandoned alien forest plantations embedded in the native

forest in the Andes could pose a danger 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 out of the scope of this study but should be further investigated, as the conservation value of the area will diminish in the future if this phenomenon continues. These issues also require close monitoring and management of native forests to prevent irreversible invasion due to the abandoned alien stands and forest degradation due to shrub encroachment.

The high dynamism of the forested classes has probably led to an increase in relative abundance of early successional native forest and plantations (both alien and native). Successional forest takes a long time to develop and hosts a similar biodiversity and ecosystem service potential to undisturbed native forest. Consequently, it is key, from a management perspective, to preserve undisturbed native forest remnants and promote species and structural complexity in successional forests.

Finally, the results of this study suggest that socioeconomic factors have a great impact in the landscape configuration. 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.



**Chapter 4** Monitoring alien invasive Pinaceae in Chilean temperate forests using object-based image analysis of WorldView-3 imagery

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## Abstract

The Chilean temperate forest, a global biodiversity hotspot, is currently under severe threat from biological invasions resulting from ongoing alien species introduction for productive forestry. Despite concern for the ecological consequences of this invasion process, broad-scale ground-based invasive species monitoring is not achievable due to logistical and budgetary constraints. Here, a remote sensing based method using very high resolution WorldView-3 imagery and object-based image analysis in Google Earth Engine is used to map early invasion of Pinaceae within a native forest matrix, with an overall accuracy ranging from 0.77 to 0.92 over four study areas. Further landscape structure assessment is performed to describe invasion in relation to native forest fragmentation and abundance. The results indicate that invasion is occurring at a rapid rate in three of the four study areas. Distance to seed sources is a major driver of invasion, but its relevance in the invasion process depends on the site, and other drivers. Alien Pinaceae trees and native *Araucaria araucana* trees exhibit a similar pattern of occupancy. This poses a risk for *A. araucana* populations, as fast growing Pinaceae could outcompete *A. araucana*. The landscape assessment indicates that fragmentation and invasion are closely linked, confirming the results of previous studies. In addition, in one site, native forest degradation and shrub encroachment are observed, potentially promoting further invasion.

Key words: Biological invasions, remote sensing, WorldView, temperate forests, object-based image analysis

### 4.1. Introduction

Temperate forests are scarce and increasingly threatened by high rates of annual forest loss (Echeverría et al., 2006, Hansen, Stehman and Potapov, 2010), land use change (Ellis, 2011) and biological invasion by alien trees (Essl, Milasowszky and Dirnböck, 2011, Richardson and Rejmánek, 2011). The Chilean temperate forests follow these global trends, having reached a 0.3% annual rate of forest loss between

1986 and 2011 (Heilmayr et al., 2016). The main contributors to this forest change are expansion and invasion of shrublands and alien Pinaceae (Miranda et al., 2017). As such, central and south Chile are considered a threatened global biodiversity hotspot (Myers et al., 2000).

The Valdivian temperate forest of Chile is considered a biogeographical island, as it has evolved in isolation due to the barrier effects of the Andes and the Pacific Ocean (Altamirano and Lara, 2010). This isolation makes the Valdivian temperate forest especially vulnerable to alien species introduction and biological invasions (Alpert, Bone and Holzapel, 2000). It also has very specific flora due to its isolated evolution (Alpert, Bone and Holzapel, 2000), including the *Araucaria araucana* tree. This is an endemic tree classified as endangered in the IUCN Red List of Threatened Species due to its declining population (Premoli, Quiroga and Gardner, 2013). It is also a culturally relevant tree, especially among indigenous communities (Herrmann, 2006), and its felling is prohibited due to its Natural Monument status (Chilean Ministry of Agriculture, 1990).

Despite their importance, deforestation, forest degradation and fragmentation of the Chilean temperate forests have been occurring since the 1970s, when the developing forestry industry started promoting the establishment of large-scale productive plantations (Echeverría et al., 2012, Miranda et al., 2015). Alien pine trees belonging to the Pinaceae family are commonly used in commercial forestry plantations, and they are considered invasive or potentially invasive in Chile (Bustamante and Castor, 1998, Teillier, Rodríguez and Serra, 2003, Echeverría et al., 2007) and other parts of the world (Pauchard and Alaback, 2004, Dziki et al., 2016). Also, the degree of fragmentation of the Chilean temperate forests makes them more vulnerable to biological invasions (Bustamante and Simonetti, 2005, Riitters et al., 2018), with Pinaceae trees a particular threat. Between the years 1962 and 1990, the Chilean Forestry Institute implemented a species introduction program to diversify the forestry sector. As part of this program, a series of test plantations, both alien and native, were introduced throughout the country (Loewe Muñoz and Murillo Bravo, 2001, Peña et al., 2008). Invasion from one of these test plantations, which is now abandoned, into the surrounding native forest has been observed previously in Chile (Peña et al., 2008), with seedlings competing with *A. araucana* regeneration (Pauchard et al., 2016). Relevant drivers of these invasions are the availability of seed

sources (alien plantations) and propagule pressure (Richardson et al., 2000, Gundale et al., 2014).

Ideally, these alien trees should be closely monitored and promptly managed to avoid biological invasions (Pauchard et al., 2016, Nuñez et al., 2017), however in practice this is highly challenging in environments such as the Chilean Andes due to the large geographical extent of these regions and their extreme topography. In these environments, the degree of invasion is often simply unknown due to a lack of monitoring resources and capabilities. Remote sensing holds significant potential for large-scale alien tree species detection and mapping, producing quantitative information on alien tree distributions over broad areas. This is extremely important for alien species control and management, as this requires targeted action to mitigate against sustained biological invasions. These actions and management decisions must, in turn, be based on robust information detailing the occurrence and extent of the invasion processes that are occurring. Freely available medium spatial resolution satellite imagery such as that from the Landsat or Sentinel-2 missions has been used previously for alien species mapping (Matongera et al., 2017, Ndlovu et al., 2018, Calleja et al., 2019, Martin-Gallego et al., 2020). However, it is not possible to detect early stages of invasion using moderate resolution imagery, as only the larger homogeneous alien patches can be mapped (Martin-Gallego et al., 2020). The spatial resolution of these sensors is insufficient to detect the smaller patches that are indicative of the initial stages of invasion. VHR satellite imagery such as that obtained from WorldView-3, which has 1.24 m multispectral and 0.31 m panchromatic spatial resolutions at nadir (Satellite Imaging Corp, 2020), has great potential for early invasion detection. WorldView imagery has been successfully used for alien species detection in heterogeneous urban landscapes (Jombo et al., 2020), forested environments (Malahlela, Cho and Mutanga, 2015, Peerbhay et al., 2016, Alvarez-Taboada, Paredes and Julián-Pelaz, 2017), riparian habitats (Fernandes et al., 2014), and arid environments (Robinson et al., 2016, Adam, Mureriwa and Newete, 2017).

Pixel-based land cover classification approaches, whereby each pixel within an image is assigned a categorical identifier based on their spectral values compared to a set of reference 'training' pixels of established land cover types, are widely used with moderate resolution imagery. However, conventional pixel-based classification approaches often yield noisy 'salt and pepper' results, particularly when applied to

VHR imagery such as Worldview (Blaschke, 2010). Object-based methods mitigate this problem by also considering the contextual information (i.e. shape, texture, compactness) within a pixel neighbourhood to create meaningful objects (Benz et al., 2004, Blaschke, 2010, Bradley, 2014). By operating at the object level, pixel misclassifications are reduced. Unlike pixel-based approaches, an object-based method for image classification has two steps. First, the image is segmented into objects, and then each object is classified using training data (Benz et al., 2004). However, these processes are often highly computationally intensive, limiting the analysis that can be performed in some scenarios. In this context, the GEE cloud computing platform (Gorelick et al., 2017) plays a key role, as it allows users to perform resource intensive tasks using Google's computing capability free of cost. GEE enables use of large scale cloud computing in a straightforward way through an Internet available application programming interface and a web-based interactive development environment (Gorelick et al., 2017). In addition, it includes an archive of easily accessible, ready-to-use data, overcoming common data management complications (Gorelick et al., 2017). By utilising the computation capacity and functionality of GEE, users are able to perform object-based analyses without the need of expensive computers or specialised software, overcoming these previous limitations.

The overall aim of this study is to detect alien Pinaceae trees using satellite remote sensing and OBIA and describe their biological invasion in relation to landscape fragmentation, distance to seed sources and native forest spatial configuration. The specific objectives are: 1) Identify and map alien Pinaceae presence and colonisation within the native forest matrix; 2) assess the spatial configuration of Pinaceae trees in the landscape to evaluate their degree of invasion, and; 3) describe the relationship between Pinaceae invasion and landscape fragmentation, distance to seed sources and native forest spatial configuration.

## 4.2. Materials and methods

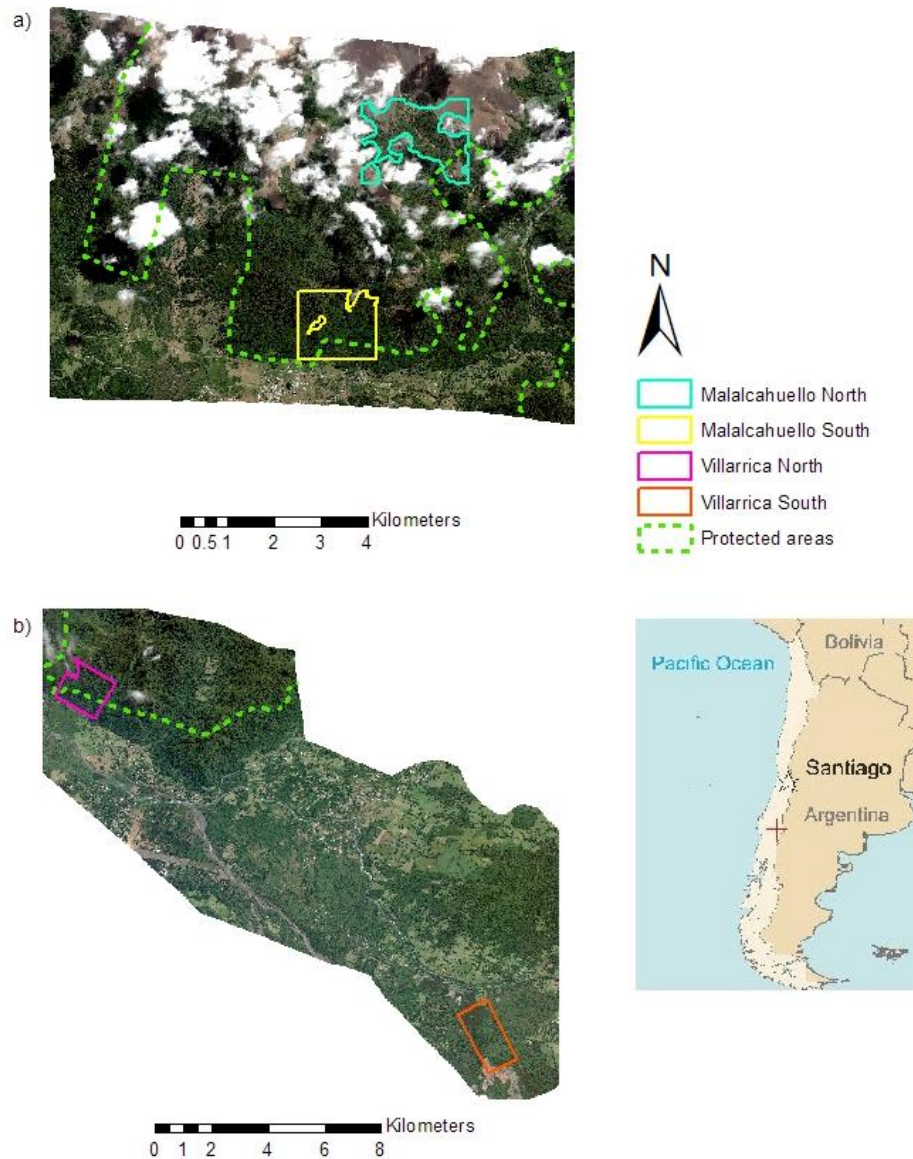
### 4.2.1. Study areas

This investigation covers four study areas located in the Chilean Andes (figure 4.1), to the east of the Araucanía region (IX administrative region) at -38° latitude. They have a similar mosaic of land covers, including Valdivian temperate forest and alien

Pinaceae plantations. Two of the study areas (Malalcahuello North and Malalcahuello South) are in the Malalcahuello National Reserve. The other two are located close to Pucón village. One of them (Villarrica North) is within the Villarrica National Reserve and the other (Villarrica South) is nearby, close to both Villarrica National Reserve and Villarrica National Park. The specific locations and shapes of the study areas were selected based on previous knowledge about alien tree locations (Peña et al., 2008, Martin-Gallego et al., 2020) and cloud cover limitations.

The Malalcahuello North study area (236 ha) is at 1450 m elevation, and contains a mosaic of native Valdivian forests including endangered *Araucaria araucana* high elevation open forests, mixed *A. araucana* and *Nothofagus* spp. forests (Smith, 2002), and high mountain shrubs and grasses. Test Pinaceae plantations established during 1969 and 1970 are also present within Malalcahuello National Reserve, and although abandoned, have become a potential invasion threat to the endangered *A. araucana* (Peña et al., 2008) forests. The Malalcahuello South study area (229 ha) is at 1000 m elevation and contains a mosaic of native *Nothofagus* spp. forest with some patches of mixed *A. araucana* and *Nothofagus* spp. forest. There are also abandoned Pinaceae plantations within this site.

The Villarrica North study area (175 ha) is at 450 m elevation, and is dominated by native *Nothofagus* spp. and shrublands, and also contains an extensive abandoned Pinaceae plantation. This abandoned plantation could be a test plantation, as several Pinaceae and *Eucalyptus* spp. plantations were established near Villarrica from 1950 onwards (Kunstmann, 1965, von Buch, 1965). The Villarrica South study area (202 ha) is at 460 m elevation, and contains a mix of native *Nothofagus* spp. forest and shrubs, and several plantations that are actively used for timber production.



**Figure 4.1** a) Malalcahuello study areas location overlaid on the WorldView-3 imagery, and b) Villarrica study areas location overlaid on the WorldView-3 imagery.

#### 4.2.2. Study invasive species

The alien tree species assessed were from the Pinaceae family which are often used in productive plantations in Chile. These include *Pinus radiata*, *Pinus contorta*, *Pinus ponderosa*, *Pinus sylvestris* and *Pseudotsuga menziesii*. These trees are very productive, establish quickly and easily, have fast growth rates and most are shadow intolerant. All these characteristics are shared by many invasive species (Dodet and Collet, 2012). Most Pinaceae used in commercial plantations are considered 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). In addition, one of the most widely used trees, *Pinus radiata*, has been reported as an invasion threat internationally, including for native forests in Australia (Calviño-Cancela and van Etten, 2018).

Since Pinaceae are generally shadow intolerant trees, they colonise open areas, such as the treeless steppes in Patagonia and the naturally open high mountain *A. araucana* forest (Peña et al., 2008, Langdon, Pauchard and Aguayo, 2010, Pauchard et al., 2016). Pinaceae trees tend to create closed canopies, which can prevent the regeneration of *A. araucana*, which is also shade intolerant (Pauchard et al., 2014). The only relatively shade tolerant tree of the Pinaceae group considered in this study is *Pseudotsuga menziesii*. This tree can regenerate under a canopy, particularly when it has been thinned or pruned (Peña and Langdon, 2007, Pauchard, Langdon and Peña, 2008). Nevertheless, it has been included in the Pinaceae group because it has a very similar spectral signature to the other *Pinus* species and cannot be accurately spectrally separated from remote sensing imagery.

#### 4.2.3. Data sources

High spatial resolution (1.60 m pixel size) WorldView-3 ortho-ready standard level satellite imagery was used for this analysis. This data product is pre-radiometrically and sensor corrected. Table 4.1 displays the WorldView-3 spectral bands used for the analysis. The imagery acquisition dates were 28 December 2017 for the Malalcahuello image, and 21 December 2017 for the Villarrica image.

**Table 4.1** Spectral bands of WorldView-3 (Satellite Imaging Corp, 2020).

Band	Wavelength (nm)
Coastal	400 – 450
Blue	450 – 510
Green	510 – 580
Yellow	585 – 625
Red	630 – 690
Red edge	705 – 745
Near infrared 1	770 – 895
Near infrared 2	860 – 1040

To perform an image classification, reference data corresponding to locations of known land cover types are required to form training datasets on which to perform the classification, and validation datasets to assess the accuracy of the classification.



To build these training and validation datasets, data from various sources were used. Field data was collected during two field campaigns in June-July 2017 and January-February 2018. During these field campaigns, land cover surveys were conducted at 146 locations surveyed in Malalcahuello, and 132 in Villarrica. At each location, the GPS location was recorded, a qualitative description of the area was made, the woody species present within a 5 m radius plot were listed, and several photos were taken (looking north, east, south, west and, sometimes upwards photos showing the forest canopy cover). To supplement the field data, further reference information was obtained from: 1) visual interpretation of the satellite imagery aided by the geolocated field photos; 2) a 2014 30 m resolution land cover map of Chile with an overall accuracy of 80% (Zhao et al., 2016); 3) the national vegetation survey conducted by the Chilean National Forest Corporation (CONAF) (CONAF, 2017) with 0.5 ha minimum mapping unit from the years 2008 and 2013; and 4) reference high resolution aerial imagery available via Google Earth from multiple imagery acquisition dates. For training and validation purposes, Google Earth high resolution imagery was used only where there was clear indication that landscape change had not occurred between the dates of the Google Earth imagery acquisition and the acquisition dates of the imagery being classified.

#### 4.2.4. Pre-processing

As mentioned in section 4.2.3., the ortho-ready standard WorldView-3 imagery used in this study is pre-radiometrically and sensor corrected. A further orthorectification pre-processing step was performed, where the imagery was corrected to improve its positional accuracy using rational polynomial coefficients (RPC) to compensate for distortions resulting from topographical variability. Finally, cloud and cloud shadow were masked using histogram thresholding.

#### 4.2.5. Image segmentation

As introduced in section 4.1., an object-based approach is used to classify the WorldView-3 imagery, as this has been shown to outperform pixel-based approaches when using high-resolution imagery (Benz et al., 2004, Blaschke, 2010, Blaschke et al., 2014). The first step of the object-based image classification is image segmentation, which was performed using the GEE cloud computing platform (Gorelick et al., 2017). The simple non-iterative clustering (SNIC) super-pixel

segmentation algorithm was used (Achanta and Ssstrunk, 2017), as it outperforms other algorithms in memory use and processing time (Mahdianpari et al., 2018). Although relatively new, the SNIC algorithm has been used for wetland mapping in Canada (Mahdianpari et al., 2018), land cover mapping for ecosystem service assessment in Greece (Verde et al., 2020) and urban mapping in China (Tu et al., 2020).

SNIC is a bottom-up approach that begins by creating a regular grid of centroids (seeds), and then assigns each pixel to a super-pixel cluster depending on its normalized distance from the centroids in the five-dimension space of colour and spatial coordinates. Two of these dimensions are the spatial coordinates and the other three are the colour coordinates in the CIELAB colour space (lightness, position between red and green, and position between blue and yellow). All pixels are ordered in priority queues according to their connectivity to the clusters. When a pixel is added to a cluster, an updated centroid value and queue is calculated for each cluster (Achanta and Ssstrunk, 2017, Mahdianpari et al., 2018, Tu et al., 2020, Verde et al., 2020).

A number of parameters need to be specified to run SNIC, these being seed distance, segment compactness, connectivity and neighbourhood size (Achanta and Ssstrunk, 2017, Tu et al., 2020). To tune these parameters to achieve the best segmentation, this study relied on the visual assessment of the clusters against the original imagery. This qualitative approach has been used in many applications of OBIA previously, including Ma et al. (2017), Tassi and Vizzari, (2020) and Verde et al. (2020). The parameters specified for all the study areas were seed distance (size) = 10, segment compactness = 1, connectivity = 8 and neighbourhood size = 256. Seed distance (size) = 10 creates a 10-pixel regular grid, allowing the creation of fairly small objects without excessive computing power demand. This value was selected after some testing and visual assessment of the resulting clusters (Tassi and Vizzari, 2020). Compactness influences the cluster shape (Tassi and Vizzari, 2020) by accounting for spatial distance. Compactness = 0 disables spatial distance weighting, creating irregular clusters. Increasing values of compactness tend to create more square clusters. Compactness = 1 creates segments that are not square but without completely disabling spatial distance weighting. Connectivity = 8 considers pixels connected if one or their edges or corners touch. Neighbourhood size = 256 creates a tile neighbourhood large enough to avoid boundary artefacts. To improve the

segmentation, an iterative approach was used to place the seeds (centroids) in optimal locations. This process involves calculating the standard deviation of the pixel values within each cluster, and identifying the worst outlier pixel in each cluster to be used as new seed (centroid) points to grow new clusters (Gorelick, 2018, Tassi and Vizzari, 2020).

#### 4.2.6. Object-based classification

Image classifications were performed in GEE using a supervised random forest classifier (Breiman, 2001) on the objects. The random forests machine learning algorithm works as an ensemble model, using the results from many models to compute a result that is always more accurate than that of any of the isolated models (Horning, 2010; Rodriguez-Galiano et al., 2012). In a review of supervised object-based land cover image classification, the random forests algorithm was shown to have the best performance (Ma et al., 2017). For each of the four study areas (Malalcahuello North, Malalcahuello South, Villarrica North and Villarrica South), approximately 180 training polygons were generated using the reference data outlined in section 4.2.3. An independent accuracy assessment was subsequently performed comparing the values of the classified image to a set of referenced points. Approximately 280 points were used for the accuracy assessment of each study area, and they were referenced using the data described in section 4.2.3.

Table 4.2 shows the classification nomenclature used in this study. This system was created using the FAO (Food and Agriculture Organization) land cover classification system (Di Gregorio and Jansen, 2005), the classification scheme used in the 2014 Chile land cover map (Zhao et al., 2016), and a previous study focusing on these study areas (Martin-Gallego et al., 2020), with subsequent adaptations for the purpose of this study.

**Table 4.2** Land cover class nomenclature.

<b>Land cover class</b>	<b>Class description</b>
Alien Pinaceae (coniferous) plantations and invasion	Pinaceae family ( <i>Pinus radiata</i> , <i>Pinus contorta</i> , <i>Pinus ponderosa</i> , <i>Pinus sylvestris</i> , <i>Pseudotsuga menziesii</i> )
Native <i>Nothofagus</i> spp. (broadleaved) forests	<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</i> )

	<i>trichosperma</i> and other native broadleaved trees.)
Native <i>Araucaria araucana</i> (coniferous) forests	<i>A. araucana</i> within a matrix of <i>Nothofagus</i> spp. or high mountain shrubs depending on the elevation.
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 pillo</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>Poa nemoralis</i> , <i>Nothofagus obliqua</i> and <i>Luma apiculata</i> . Also, high areas dominated by mountain flora.
Bare	Rocky outcrops, bare soils in rotation agricultural grassland, sandy lake beaches or high mountain areas above the limit of vegetation.
Urban	Cities, smaller urban settlements, or impervious surfaces such as asphalt, concrete and roof materials.

#### 4.2.7. Landscape metrics

Landscape metrics analysis was performed to assess whether alien invasion is linked to the structural configuration of the landscape, especially proximity to other plantations and fragmentation (Richardson et al., 2000, Marvier, Kareiva and Neubert, 2004, León Cordero et al., 2016) in this environment. To perform the analysis, Pinaceae plantations were first identified in the land cover classifications; these were assumed to be potential seed sources for invasion. Then, landscape metrics were calculated in a series of nested 100 m buffers around the plantations. These landscape metrics quantify the geometric spatial configuration of patches of different land cover types within a landscape, and were calculated using Fragstats 4.2 (McGarigal et al., 2002, McGarigal, 2014).

To identify the Pinaceae plantations present in each study area, a patch level landscape analysis was performed in Fragstats using the metrics patch area and contiguity index. Patch area measures the area of each patch in hectares. Contiguity

index is a shape metric that equals zero for a one-pixel patch and increases up to 1 as connectedness increases. It assesses the contiguity of pixels to provide an index of patch boundary configuration (McGarigal, 2014). For a Pinaceae patch to be considered a plantation (i.e. a seed source), it had to fulfil the following criteria: patch size > 1 ha, contiguity index > 0.85 and classified as Pinaceae in the CONAF forest surveys of 2013 and 2007. Finally, the plantations were digitised manually in the selected locations, aided by visual interpretation of the high-resolution WorldView-3 data and Google Earth imagery. They were digitised rather than determined automatically from the imagery to represent the regular shape of the original plantation footprint more accurately. Automatic extraction of plantation objects might include areas of invasion around the edge of (i.e. stemming from) the original plantation. In addition, plantations in the two Malalcahuello sites and Villarrica North were verified during the field campaigns to these areas, further corroborated by interpretation of high resolution aerial imagery. The plantations in Villarrica were verified using high resolution aerial imagery.

Next, a series of nested 100 m buffers from the plantation polygons were created for each site. Class level and landscape level Fragstats analysis was performed for each buffer around the plantation patches. The class level metrics were calculated for each vegetated land cover class. The landscape level metrics used are outlined in table 4.3. These metrics were selected based on their previous use in literature to describe land cover spatial configuration patterns relevant to vegetation studies in Chile (Echeverría et al., 2006, 2008, 2012, Altamirano et al., 2013, Molina et al., 2015, Hernández et al., 2016, Martin-Gallego et al., 2020), and on their robustness to assess landscape fragmentation. An ideal fragmentation index should be correlated with aggregation and as independent as possible of class area to avoid underestimating the connectivity of the less widespread habitats (Neel, McGarigal and Cushman, 2004).

**Table 4.3** Landscape metrics used in this study and their 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 per hectare (McGarigal, 2014).
	Mean patch area	AREA_MN	AREA_MN is the mean of the areas (in hectares) of all

			patches within a class (McGarigal, 2014).
	Largest patch index	LPI	LPI assesses dominance as the percentage of the total area of the landscape that is comprised by the largest patch of the class (McGarigal, 2014).
	Percentage of landscape	PLAND	PLAND quantifies abundance of a class. It is the percentage of the landscape covered by a specific class (McGarigal, 2014).
	Perimeter area fractal dimension	PAFRAC	PAFRAC assesses the shape complexity of patches, whatever their extent (McGarigal, 2014). A Euclidean geometry has a value of 1, increasing gradually to 2 as the perimeter gets more complex (McGarigal, 2014). It is correlated with aggregation and not very correlated with area (Neel, McGarigal and Cushman, 2004, Wang, Blanchet and Koper, 2014).
	Clumpiness index	CLUMPY	CLUMPY measures aggregation. 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).
	Shannon's diversity index	SHDI	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 with class richness (McGarigal, 2014). A closely related diversity index has been

			previously used in Chile by Molina et al. (2016).
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Finally, the Pinaceae plantation locations were compared against the landscape configuration assessed from the classification of the WorldView-3 imagery to explain the pattern of invasion. The metrics of patch density, mean patch area and dominance of Pinaceae were plotted against distance to plantations to assess the effect of distance to seed sources (plantations) on the level of invasion. Then, dominance of Pinaceae and patch density and mean patch area of all the vegetated land covers were plotted against distance to plantations to detect relationships between Pinaceae invasion and presence and abundance of other types of vegetation. Finally, dominance of Pinaceae and landscape level metrics were plotted against the distance to plantations to evaluate the relation between fragmentation and Pinaceae invasion.

### 4.3. Results

#### 4.3.1. Pinaceae invasion mapping

A land cover classification was produced for each study area. The overall accuracies of the classifications were 0.80 for Malalcahuello North, 0.77 for Malalcahuello South, 0.86 for Villarrica North and 0.92 for Villarrica South. Table 4.4 shows the accuracies for the vegetated classes for each study area. The complete confusion matrices can be found in appendix H. Grasslands, bare soil and urban land covers achieved high accuracies (producer's and user's accuracy were up to 100%). Alien Pinaceae (producer's accuracy was up to 96%; user's accuracy, up to 88%) and native *Nothofagus* spp. (producer's accuracy was up to 97%; user's accuracy, up to 95%) and *A. araucana* (producer's accuracy was up to 65%; user's accuracy, up to 78%) were also well classified. *A. araucana* were sometimes confused with Pinaceae, as they are both coniferous trees with very similar spectral signatures. This confusion appeared mostly in Malalcahuello North, which is a very high elevation area where both *A. araucana* and Pinaceae grow in low to medium density, appearing as isolated trees or very small patches within a matrix of shrubs. Shrubs were sometimes confused with *Nothofagus* spp. This is not surprising, as both *Nothofagus* spp. and the shrub species are broadleaved, and some small or stunted *Nothofagus* spp. are included in the shrubs class.

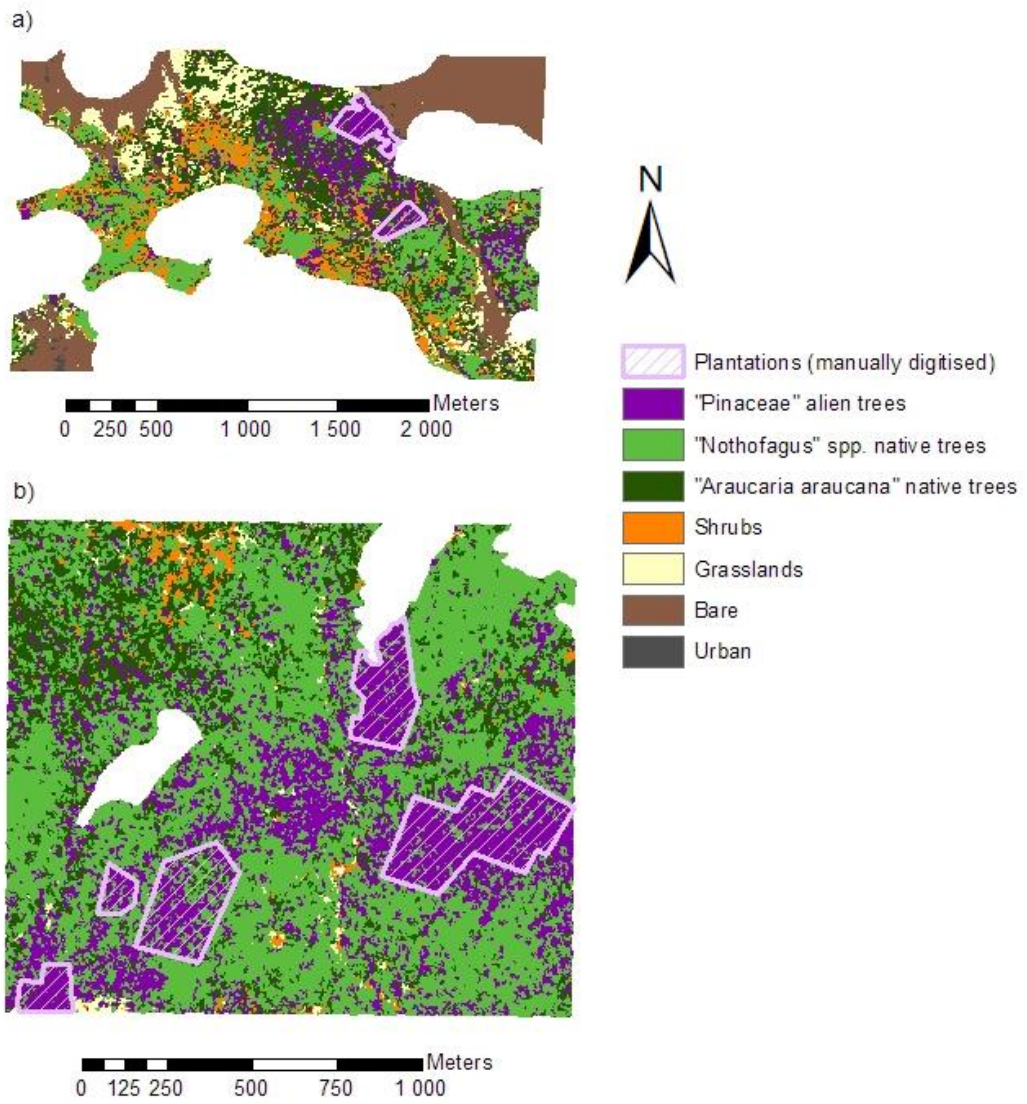
**Table 4.4** Land cover classification user's and producer's accuracies for the different study areas. (PA = producer's accuracy and UA = user's accuracy.)

Land cover classes	Malalcahuello North		Malalcahuello South		Villarrica North		Villarrica South	
	PA	UA	PA	UA	PA	UA	PA	UA
Pinaceae	0.74	0.70	0.88	0.88	0.94	0.81	0.96	0.86
<i>Nothofagus</i> spp.	0.97	0.76	0.85	0.68	0.84	0.95	0.84	0.87
<i>A. araucana</i>	0.65	0.78	0.65	0.70	-	-	-	-
Shrubs	0.61	0.85	0.48	0.83	0.74	0.74	0.84	0.95
Grasslands	0.97	0.97	0.88	0.88	0.96	1.00	1.00	1.00

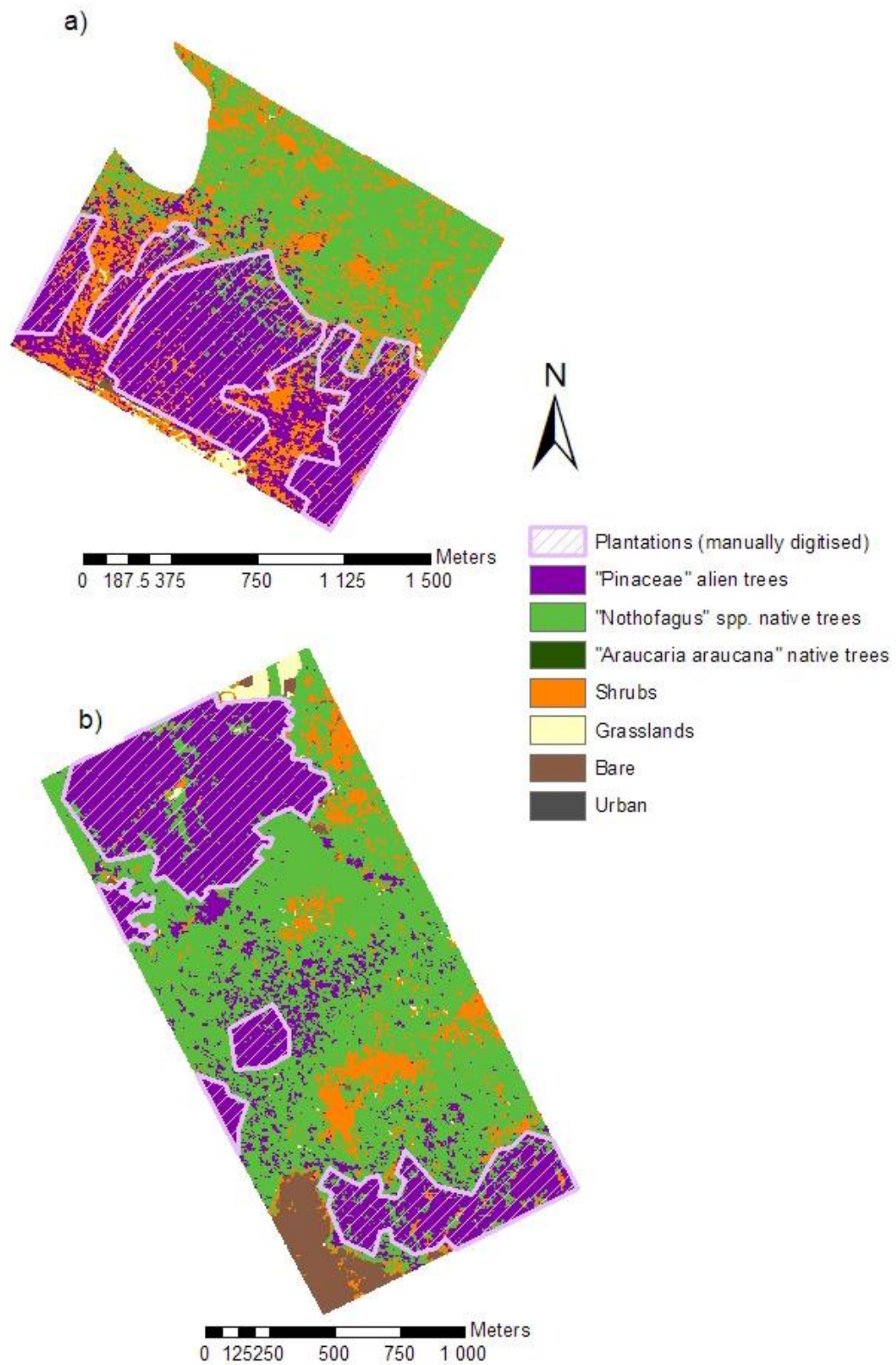
Figures 4.2 and 4.3 show the land cover maps for the study areas. These land cover maps were used in the next stage of the method for the landscape configuration analysis. Malalcahuello North is a high elevation area within a National Reserve. Currently, there are no new forestry plantations being established in the area as this is a conservation site. However, the land cover map shows an abandoned Pinaceae plantation in the northeastern part of the study area. This abandoned plantation was visited during field work and several patches of Pinaceae invasion into the surrounding *A. araucana* and *Nothofagus* spp. forest were observed. This invasion is displayed in the land cover map as small Pinaceae patches in the north-central part of the map. This invasion has also been documented by Peña et al., (2008) in an earlier study conducted in this area. The *A. araucana* forest in Malalcahuello North is naturally open, and Pinaceae tends to colonize open areas to close the canopy. This way, Pinaceae colonization can compromise future regeneration of *A. araucana*, which is a shade intolerant tree. Malalcahuello South is a lower elevation area situated on the edge of the National Reserve, close to Malalcahuello Village. As it is a lower elevation area, there is less presence of *A. araucana*. Pure, open *A. araucana* forest is not found in this area. Instead, the dominant native forest is *Nothofagus* spp. with some interspersed individual *A. araucana* trees. The largest individual *A. araucana* trees are detected by this method, as they form extensive crowns. Several large Pinaceae patches are detected. Since these are inside the National Reserve, it seems clear they are abandoned plantations; records regarding old plantations are simply not generally available. Small Pinaceae patches are observed within the native *Nothofagus* spp. forest matrix. This is likely to be Pinaceae trees invading native forest gaps.



Villarrica North is located on the edge of the Villarrica National Reserve, near Pucón village. The northeastern part of the study area is located inside the protected area. A very large Pinaceae forest can be observed just outside the boundary of the protected area, with presence of small invasive patches inside the National Reserve. The area inside the Reserve has *Nothofagus* spp. forest and native shrubs. Earlier studies have found that the chief driver of native forest loss is conversion to shrubs, which accounts for 45% of the net native Valdivian forest loss in Chile (Miranda et al., 2017). Conversion to Pinaceae is the second most important contributor to native forest loss (Miranda et al., 2017). Villarrica South is outside protected areas and comprises a mix of alien Pinaceae plantations and native *Nothofagus* spp. and shrubs. Unlike the rest of the study areas, the Pinaceae plantations in Villarrica South are being actively managed, with periodical clear-cuts. For example, upon visual examination of the study area on Google Earth historical high resolution imager, it is observed that the bare soil patch to the southwest of the study area was vegetated at the beginning of 2016, and the two small Pinaceae plantations to the west of the area were completely felled in 2019.



**Figure 4.2** Land cover classification of a) Malalcahuello North, and b) Malalcahuello South.



**Figure 4.3** Land cover classifications of a) Villarrica North, and b) Villarrica South.

#### 4.3.2. Landscape configuration

Pinaceae plantations were identified in all study areas, as observed in figures 4.2 and 4.3. Some plantations in Malalcahuello South and Villarrica North contain a few small *Nothofagus* spp. and shrubs patches within them. This may be a result of classification confusion, however, since plantations in these study areas are abandoned, it could also be a case of native vegetation regeneration within the *Pinaceae* plantation. There is also some *Nothofagus* spp. presence in the Pinaceae plantations in Villarrica South. In this case, it is not confusion, but a patch of native forest within a topographically challenging area. Since Pinaceae felling is performed using machinery, it requires flat terrain. Areas inaccessible due to abrupt topography are left with their natural vegetation. These native patches are clearly observable in the WorldView-3 image and the land cover maps in the northern part of Villarrica South.

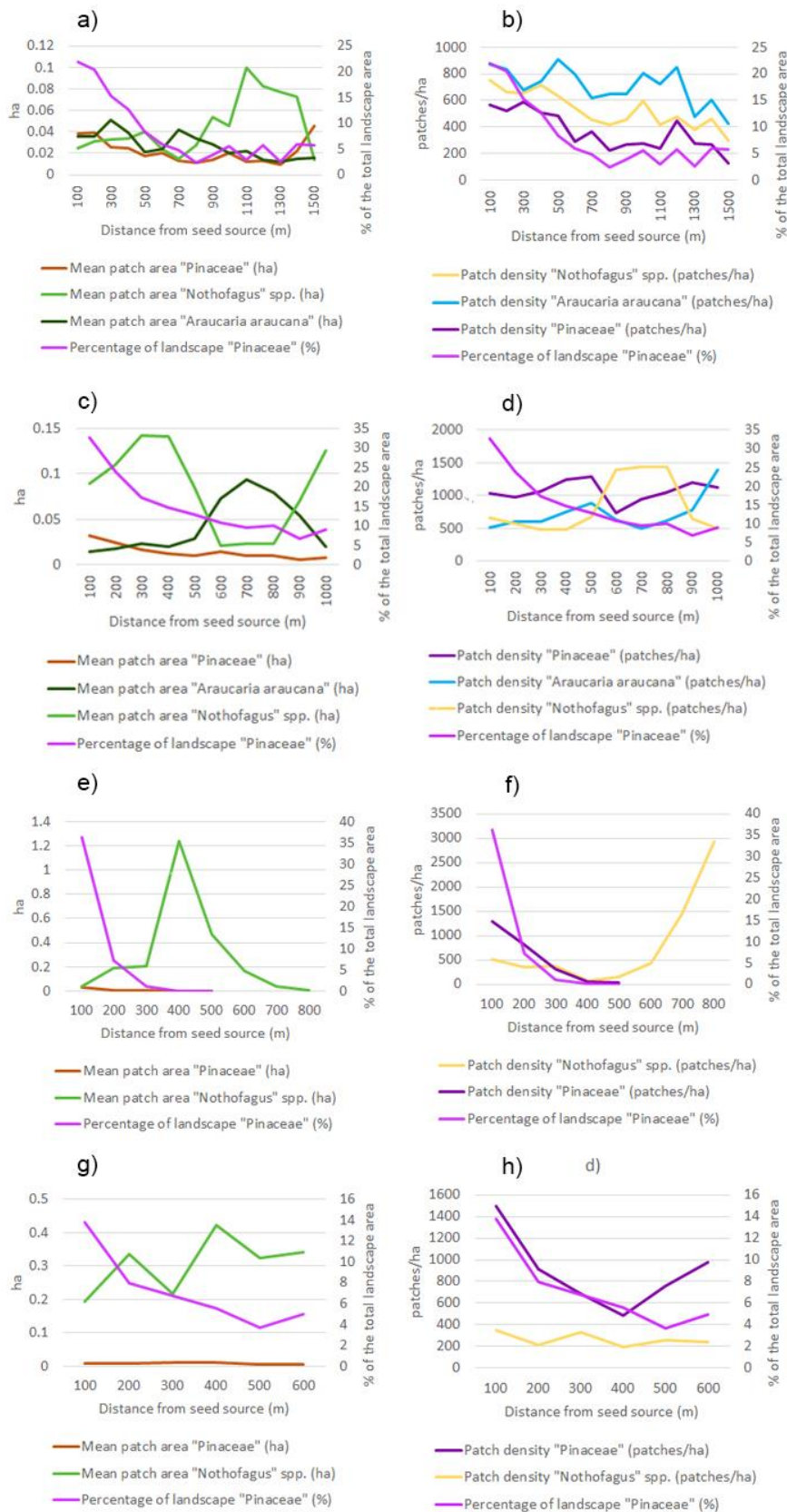
Figure 4.4 shows class level landscape metric results for Pinaceae. In Malalcahuello North, Malalcahuello South and Villarrica South, there were Pinaceae trees in every 100 m buffer analysed (1500 m for Malalcahuello North, 1000 m for Malalcahuello South, and 600 m for Villarrica South). The results for Malalcahuello North, with Pinaceae present at 1500 m from the plantations, corroborates a previous study in the same area, which found *Pinus contorta* trees up to 1200 m from the seed source (Peña et al., 2008). Only Villarrica North had Pinaceae trees only up to 500 m from the plantation (seed source), despite having similar initial patch density and mean patch area to the other study areas. This difference could be due to the landscape configuration and topography of Villarrica North. In this study area, the plantation is located on the lower slopes of a steep hill. Propagules can only establish in the uphill direction, as there is a road downhill preventing spread. In previous *Pinus contorta* invasion studies in Chile, it has been found that the dominant invasion direction was downhill, as the topography determines the dominant wind direction in very mountainous terrains due to the presence of katabatic winds (Peña et al., 2008). This landscape configuration has probably limited the establishment of new Pinaceae as far away from the plantation as in the other study areas. Also, the dominant Pinaceae species in this site is *Pseudotsuga menziesii*, which can regenerate under canopy, impossible to detect using optical satellite remote sensing.

Only Malalcahuello North and Villarrica North showed a steady decrease in Pinaceae patch density, percentage of landscape and mean patch area with distance from seed sources. This is the typical invasion pattern, where most regeneration appears close to the seed source and decreases gradually with distance (Pauchard, Langdon and Peña, 2008). In Malalcahuello North there is an outlier at 1500 m from the plantation, where mean patch area increases sharply. This could be a classification error, a highly invaded area, or an unidentified former plantation. On the other hand, Malalcahuello South and Villarrica South do not show this clear pattern of decreasing Pinaceae with distance. The highest Pinaceae dominance (percentage of landscape covered by Pinaceae) is found close to plantations. However, patch densities and mean patch areas vary erratically. This suggests that there might be more important drivers of invasion than distance to seed sources in these sites. Alternatively, it could be a result of incomplete seed source mapping due to classification errors or due to Pinaceae patches located just outside the study area boundaries, which could be influencing the area within. Presence of forest degradation or gaps expressed as patch density, mean patch area or dominance of shrublands and grasslands did not show a relation with patch densities and mean patch areas of Pinaceae. Other drivers determining the invasion could be topography such as in Villarrica North, or human intervention.



**Figure 4.4** Class level landscape metrics for Pinaceae cover in a) and b) Malalcahuello North; c) and d) Malalcahuello South; e) and f) Villarrica North; g) and h) Villarrica South.

Native forests and their relationship with Pinaceae presence (as represented by the percentage of the landscape covered by Pinaceae, their mean patch area and patch density) are shown in figure 4.5. Malalcahuello North is a high elevation landscape dominated by open *A. araucaria* forests and dense *Nothofagus* spp. forests with some *A. araucana* present within them in the lower part of the study area. As mentioned earlier, Pinaceae exhibits the typical pattern of invasion here, with decreasing presence with distance from the plantation (seed source). *Nothofagus* spp. decreases in patch density with decreasing Pinaceae presence but increases sharply in mean patch area. As the landscape is less invaded by Pinaceae, it gives way to large patches of *Nothofagus* spp. forest. However, this is not happening with *A. araucana*. With decreasing invasion, there is more patch density of *A. araucana*, but its mean patch area decreases slightly. This correlation between Pinaceae and *A. araucana* presence has been reported previously for this study area (Peña et al., 2008). Malalcahuello South is a lower elevation area that is mostly dominated by dense *Nothofagus* spp. forest, with some presence of *A. araucana* intermixed. Here, the invasion pattern is not as clear, as even though the percentage of landscape and the mean patch area covered by Pinaceae decreases with distance from the plantation (seed source), the patch density shows a more confusing pattern. Mean patch area of native forest does not show a clear pattern either, though patch density of *Nothofagus* spp. increases as Pinaceae dominance decreases. Then, patch density of *Nothofagus* spp. decreases again as patch density of *A. araucana* increases. In this site, a positive relation between patch density of Pinaceae and *A. araucana* can be observed, although not as markedly as in Malalcahuello North.

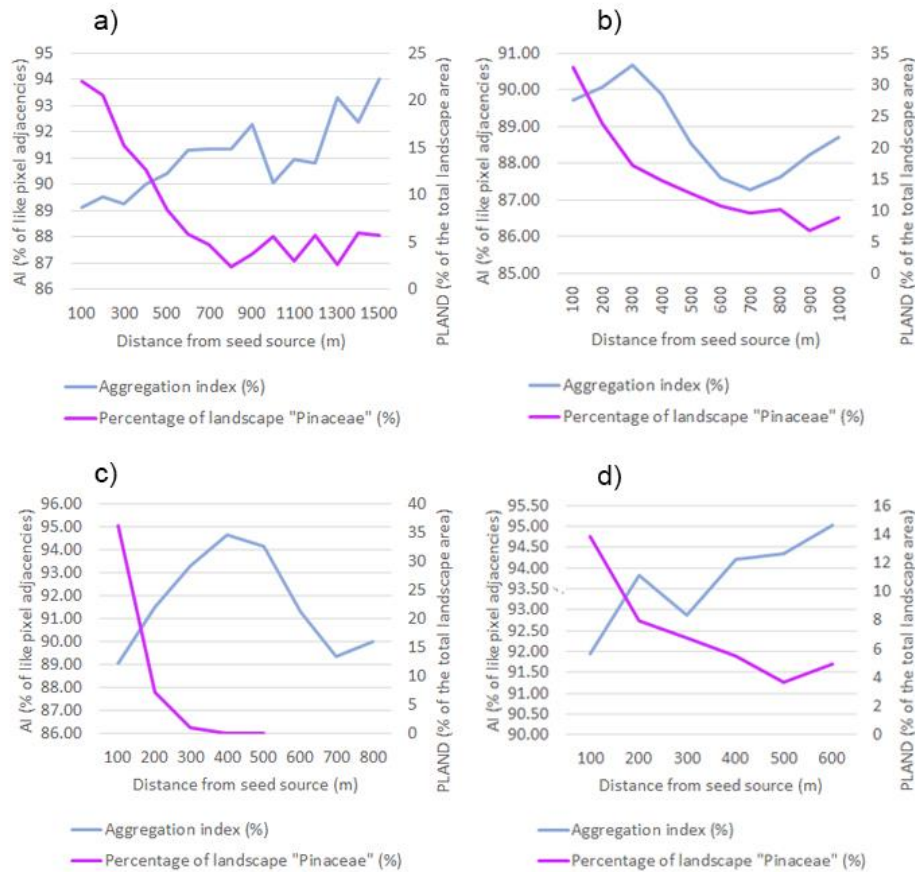


**Figure 4.5** Class level landscape metrics results for native forest in a) and b) Malalcahuello North; and c) and d) Malalcahuello South; e) and f) Villarrica North; and g) and h) Villarrica South.



The study areas in Villarrica do not show native forest patterns as clearly as the ones found in Malalcahuello. An earlier, broad scale landscape model explaining Pinaceae presence in Malalcahuello and Villarrica found a more complicated landscape in Villarrica, with more variables influencing Pinaceae presence (Martin-Gallego et al., 2020). Villarrica North showed a sharp increase in *Nothofagus* spp. mean patch area and patch density as Pinaceae disappeared. This is probably again a result of the topography, as Pinaceae are forced to travel uphill in this site, and previous studies have found Pinaceae invading mostly downhill (Peña et al., 2008). The low value of mean patch area of *Nothofagus* spp. in the furthest buffer from the plantation indicates an increase in native forest fragmentation that is unrelated to Pinaceae invasion. This is further confirmed by a steady increase in percentage of landscape covered by shrubs, which become the dominant class in the most distant buffers, covering almost 70% of the landscape in the last 100 m buffer (appendix I). In Villarrica South, *Nothofagus* spp. mean patch area increases as Pinaceae presence decreases, but its patch density remains constant. However, its mean patch area is relatively high in general compared to the other study areas. This, together with the low and stable patch density suggests that native forest fragmentation is not occurring in this site.

The landscape level metric aggregation index (figure 4.6) has minimum values (i.e. maximum levels of disaggregation) closest to plantations in Malalcahuello North, and Villarrica North and South. These results agree with the widespread theory that habitat fragmentation and invasion are closely linked (Marvier, Kareiva and Neubert, 2004, León Cordero et al., 2016). Malalcahuello South and Villarrica North exhibit an increasing landscape fragmentation in the most distant buffers, suggesting that there are other disturbance sources fragmenting the landscape. Not all landscape metrics showed meaningful results. The most pertinent findings are presented here, with the full output of landscape assessment given in appendix I.



**Figure 4.6** Landscape level aggregation index for a) Malalcahuello North, b) Malalcahuello South, c) Villarrica North, and d) Villarrica South.

The following abbreviations are used: AI = aggregation index, PLAND = percentage of landscape.

#### 4.4. Discussion

The first objective of this study was to identify and map Pinaceae trees within the matrix of native *Nothofagus* spp. and *A. araucana* forests. The OBIA of VHR WorldView-3 data using SNIC segmentation yielded more detailed and more accurate vegetation maps than previous studies performed in the area using pixel-based methods and moderate resolution Sentinel-2 data (Martin-Gallego et al., 2020). A very high level of detail was achieved, detecting patches as small as 2 m<sup>2</sup>, which in some cases represent individual trees. Therefore, the object-based method applied in this study is suitable for Pinaceae invasion detection. Other types of satellite imagery such as Sentinel-2 or Landsat do not have a fine enough spatial resolution to detect early Pinaceae invasion, instead detecting only large homogeneous patches that are often plantations (Martin-Gallego et al., 2020).

However, VHR satellite data can be costly, often prohibitively so for continuous monitoring. Drone imagery could be a cost-effective alternative, but its ability to separate very spectrally similar trees such as *A. araucana* and Pinaceae should be tested in future work. In addition, the availability of the GEE cloud computing platform makes the OBIA approach a viable option for land managers, who often lack the computing resources necessary to use OBIA to inform their decisions.

The second objective was to assess the spatial configuration of Pinaceae trees and evaluate their degree of invasion. Alien trees were present in all 100 m buffers in Malalcahuello North (furthest buffer = 1500 m), Malalcahuello South (furthest buffer = 1000 m) and Villarrica South (furthest buffer = 600 m). The results for Malalcahuello North agree with a previous study, which found Pinaceae regeneration up to 1200 m away from plantations (Peña et al., 2008). In Villarrica North, there were buffers up to 800 m away from the plantations (seed sources), but Pinaceae only reached 500 m. Between 1962 and 1990, the Chilean Forestry Institute implemented a forestry species introduction program to diversify the forestry sector. As part of this program, a series of test plantations, both alien and native, were introduced throughout the country (Loewe Muñoz and Murillo Bravo, 2001, Peña et al., 2008). In Malalcahuello, this program took place between 1969 and 1970, and in Villarrica, between 1965 and 1967 (Loewe Muñoz and Murillo Bravo, 2001, Peña et al., 2008). Considering this time frame and the distances reached by Pinaceae in the sites with abandoned plantations (Malalcahuello North, Malalcahuello South and Villarrica North), it is found that Pinaceae have moved 32 m per year in Malalcahuello North, 21 m per year in Malalcahuello South and 10 m per year in Villarrica North. These are very concerning results, since the threshold for a plant species with seed reproduction to be considered invasive is 2 m per year (Richardson et al., 2000). In this environment, Pinaceae is highly invasive, even in the less favourable conditions (uphill in Malalcahuello South and Villarrica North).

The pattern of invasion was typical for Malalcahuello North and Villarrica North, with a steady decrease in Pinaceae patch density, dominance and mean patch area with increasing distance from seed sources. However, although in Malalcahuello South and Villarrica South Pinaceae dominance did decrease with increasing distance from plantations, the patch densities and mean patch areas did not show a clear pattern. This could be a result of incomplete seed source mapping due to classification errors or seed sources outside the study area boundaries influencing

the area within. However, this inconsistency could also suggest that distance to seed sources is interacting with other drivers of invasion. This mix of invasion drivers has been reported before for Pinaceae in this region (Taylor et al., 2016). For Villarrica South, one of these other drivers is likely to be human intervention, as the plantations of this study area are being actively exploited, felled periodically. In Malalcahuello North, which has a dense matrix of *Nothofagus* spp., the presence of canopy gaps or areas of degraded forest can play a significant role in Pinaceae establishment, as most of the species in this group are only able to regenerate in open conditions. Only *Pseudotsuga menziesii* is capable of regenerating under canopy, but it prefers slightly open areas (Peña and Langdon, 2007, Pauchard, Langdon and Peña, 2008).

The landscape level metric aggregation index confirmed the widespread theory that links fragmentation and biological invasions (Marvier, Kareiva and Neubert, 2004, Bustamante and Simonetti, 2005, León Cordero et al., 2016, Riitters et al., 2018). This theory suggests that fragmented landscapes are more vulnerable to biological invasions. Most of the study areas exhibit the lowest level of landscape aggregation near plantations.

The third objective of this study was to describe the relationship between Pinaceae invasion and native forest spatial configuration and fragmentation. The landscape configuration assessment found that presence of Pinaceae had a negative relation with *Nothofagus* spp. forests, but a positive one with *A. araucana*. This positive relation between Pinaceae and *A. araucana* was reported earlier in this site by Peña et al. (2008). As it tends to be located in the same areas as *A. araucana*, Pinaceae poses a threat to the regeneration of *A. araucana* (Pauchard et al., 2014). Since Pinaceae trees have a tendency to create canopies, they could prevent the regeneration of the shade intolerant *A. araucana*. In addition, this study found that the native *Nothofagus* spp. forest is suffering a degradation process in some areas (Villarrica North), as its dominance decreases to give way to shrubs. Shrubs are an important contributor to native forest loss in Chile, accounting for 45 % of the loss (Miranda et al., 2017). In these degraded areas, the landscape also appears more fragmented, as indicated by the landscape level aggregation index.

#### 4.5. Conclusion

Invasion monitoring and rapid targeted management have proven to be an effective tool for invasive species control (Pauchard et al., 2016, Nuñez et al., 2017). This monitoring is costly and time-consuming using traditional, ground-based methods, especially in areas with challenging topography such as the Chilean Andes. This study presented a robust and repeatable method of assessing alien tree invasion over large areas using WorldView-3 VHR optical remote sensing data and the freely available GEE cloud computing platform to perform OBIA. The accuracy (0.77 – 0.92 overall classification accuracy) and level of detail achieved (2.56 m<sup>2</sup>) allowed the detection of both alien plantations, and biological invasion by these alien forestry species into the native *Nothofagus* spp. and *A. araucana* forests. By using the GEE cloud computing platform, hardware and data management limitations are overcome. This allows land managers to use this method to inform conservation decisions in a reasonably quick and easy way and protect the valuable Valdivian temperate forests effectively.

This study determined that this part of the Valdivian temperate forest is suffering Pinaceae invasion and forest degradation with shrub encroachment in some areas. Invasion is happening at a rapid rate, and forest degradation could promote further invasion. The landscape assessment results confirmed the theory that increasing fragmentation and biological invasions are closely linked. Although distance to seed sources is a major driver of invasion, this study found that there are other drivers playing an important role, such as topography or human intervention. This should be assessed in future modelling work. Also, future multitemporal work using VHR imagery would further enable identification of the drivers of invasion.

Alien Pinaceae and native *A. araucana* are both shade intolerant trees and show a similar pattern of occupancy. This could put the long-term conservation of the endemic *A. araucana* at risk, as Pinaceae grow fast in open areas (such as the naturally open *A. araucana* forest) and could outcompete *A. araucana* seedlings by rapidly creating a closed canopy. This could compromise the long-term conservation of *A. araucana*. This process is already happening within National Reserves, where test Pinaceae plantations were abandoned half a century ago, diminishing their conservation value. Invasive Pinaceae management should be a priority measure to

avoid the loss of the characteristic landscape of high mountain open *A. araucana* forests.

## Chapter 5 Conclusions

### 5.1. Summary of the thesis

Alien species and biological invasions in South America are relatively understudied (Speziale et al., 2012). This research project has increased our understanding through rigorous study of a region of temperate forests in the south-central Chilean Andes. The main aim of this research was to understand the patterns of alien tree expansion, invasion, and occupancy in the Valdivian temperate forests over a range of spatial scales; with a goal to inform forest and conservation management and enhance the sustainability of the forestry sector in Chile. Mapping and understanding the distribution patterns of alien species are a fundamental first step for invasion risk modelling, early invasion detection and rapid response (Bradley, 2014). This is key information for a successful alien species control programme (Pauchard et al., 2016, Nuñez et al., 2017). This overall research aim has been addressed through three research chapters:

1. Alien tree plantations were mapped at a broad scale using Sentinel-2 imagery, and the spatial distribution of alien tree stands was modelled in relation to landscape configuration, as reported in chapter 2.
2. The historical alien tree plantation pattern and forest change dynamics were investigated using archived Landsat imagery, as reported in chapter 3.
3. Early alien invasion was mapped at the local scale using finer spatial resolution WorldView-3 imagery and OBIA, as reported in chapter 4. Also, here, the relationship between invasion, fragmentation, distance to seed sources and landscape configuration was further investigated at the local scale.

The main findings of the research chapters are:

1. Moderate resolution satellite imagery such as Landsat or Sentinel-2 is useful for broad-scale monitoring of deforestation and conversion to alien stands (chapters 2 and 3). Finer resolution imagery such as WorldView-3 is well suited for early tree invasion detection using OBIA (chapter 4).

2. Socioeconomic factors have great impact on landscape configuration. The cessation of subsidized forestry in 2012 marked an inflection point in land cover change dynamics. Smallholders shifted to short rotation forestry (including alien *Eucalyptus* spp. for fibre and native *Nothofagus* spp. for local uses) or other land uses, while large companies kept engaging in long rotation alien Pinaceae forestry (chapter 3).
3. Alien Pinaceae and native *A. araucana* tend to occupy the same locations (chapter 2), and Pinaceae are invading the naturally open *A. araucana* forest (chapter 4).
4. Early Pinaceae invasion is linked to fragmentation, distance to seed sources, topography and human intervention (chapter 4).
5. Incidentally, this study also found that native forest degradation is occurring. Native *A. araucana* forests are fragmenting (chapter 2 and 3) and decreasing in areal coverage as shrubs encroach (chapter 3 and 4). This degradation could make the forest more vulnerable to invasions in the future.

## 5.2. Satellite remote sensing for alien tree species monitoring

Before attempting to undertake any risk modelling or management planning, accurate mapping of the distribution of alien species is essential. Many of these distribution studies are performed using herbarium records. Remote sensing image analysis may be a superior approach, avoiding common bias in ground-based methods such as those related with choosing collection locations (Bradley, 2014). In addition, continuous forest change monitoring is essential for sustainable land management to make native forest conservation and productive forestry compatible. In this context, the first objective of this thesis was to evaluate the capabilities of different satellite remote sensing data sources to detect alien tree stands accurately.

Using moderate resolution imagery such as Landsat or Sentinel-2 is the most cost-effective way of mapping and monitoring forest over large areas. Landsat, with its imagery archive, is especially useful for retrospective monitoring. However, for present and future monitoring, Sentinel-2 is recommended, as its finer spatial resolution and superior spectral capabilities (including four red edge bands) show



greater potential to distinguish different types of forest accurately. In this study, thematically detailed land cover maps were achieved using Sentinel-2. Grasslands, shrublands and four types of forest were separated accurately. In this sense, the maps created here represent an advancement on previous research, as most earlier vegetation mapping efforts are based on (i.e. limited to) plant functional types (Ustin and Gamon, 2010, Bradley, 2014). Species-specific detection would be ideal, but it is still uncommon. Sentinel-2 was able to separate the *A. araucana* species, the *Eucalyptus* genus and the Pinaceae family accurately. Moderate resolution imagery is key to understand invasion risk at advanced stages of invasion at the landscape scale (Bradley, 2014) and to inform landscape scale management actions, which are a cost effective way of protecting against biological invasions (Marvier, Kareiva and Neubert, 2004).

However, Sentinel-2 can only detect relatively large land cover patches, being limited to plantations or large naturalised alien stands. Finer spatial resolution imagery is required to detect early stages of invasion. In this study, VHR WorldView-3 imagery detected early Pinaceae invasion. Quick, targeted management is an effective tool for invasive species control (Pauchard et al., 2016, Nuñez et al., 2017), and it may be achievable via continuous monitoring with VHR satellite imagery. The downside to VHR satellite imagery is its cost. Continuous monitoring would be economically unfeasible using this type of imagery, especially given the relatively limited funding for conservation in Chile (Waldron et al., 2013). Researchers also lack the financial resources to purchase large amounts of VHR data (Pettorelli et al., 2014), hindering advancement in invasion science. Making remote sensing data freely available for conservation studies would be a major step to facilitate this progress (Pettorelli et al., 2014). In this context, the recent project by Planet is highly relevant, as it will provide cost free global access to VHR satellite imagery of tropical forests to support conservation efforts. Since October 2020, analysis ready basemaps updated every month can be downloaded from Planet's online satellite imagery platform. This initiative has been possible thanks to the partnership between Planet, Norway's Ministry of Climate and Environment, Kongsberg Satellite Services, and Airbus (O'Shea, 2020).

### 5.3. Alien tree cover at different scales

The second objective of this research was to assess the current extent of alien tree cover and potential invasion in the Valdivian temperate forest at different spatial scales. The broad scale land cover maps, which are limited to large alien forest stands and plantations, reveal a clear altitudinal pattern in the distribution of alien tree cover. This pattern has been consistent over the last 40 years. Lower elevation areas (valley) have more land cover diversity and fragmentation. They also host most of the alien tree cover. Most native forests are located in higher elevation areas (Andes), although, worryingly, there were some alien tree patches at high elevation, and even within protected areas. Due to the difficult access, there is generally less human disturbance and less alien tree cover at high elevations. Altamirano and Lara (2010) observed a similar pattern of forest degradation and alien tree cover, with most of disturbances occurring at medium to low elevations and near roads. Especially interesting is the role of roads, which have often been identified as a major vector of invasion (Pauchard and Alaback, 2004, Riitters et al., 2018). However, the broad scale invasion models developed in this research did not find roads to be a relevant invasion driver. This could be due to the lack of reliable road data at the scale of the analysis. Narrow forest roads and paths may not be included in the dataset used for modelling. Distance to seed source and propagule pressure are other major drivers of invasion (Richardson et al., 2000, Giorgis et al., 2011, 2016, Gundale et al., 2014, Altamirano et al., 2016, Arellano-Cataldo and Smith-Ramírez, 2016, Pauchard et al., 2016), but here they were not consistently identified as relevant in the broad scale invasion models. At the resolution used (10 m pixel size), many of the patches detected are a result of plantation rather than natural invasion, as the smallest patches are not detected. Consequently, in some cases, the pattern of invasion is masked by the pattern of plantation. However, distance to seed source and propagule pressure showed a relationship with alien tree cover in the very fine resolution study. This relationship was not consistent for all metrics in all study areas, suggesting that other invasion drivers could dominate over distance to seed sources. Also, these inconsistencies could be a result of undetected alien patches or alien patches located just outside the study area boundaries. This stresses the importance of choosing the right scale of observation for biological invasion assessment. If too coarse a scale is used, patterns of natural invasion may be masked by other, anthropogenic factors driving alien spread. The appropriate scale depends

on the stage of invasion and the growth form of the alien species. In the case of alien tree invasions in the south-central Chilean Andes, moderate resolution imagery is too coarse to identify natural invasion, but could help informing large-scale management decisions by identifying high risk or high conservation value areas.

#### 5.4. Alien tree expansion and invasion

The third research objective was to assess the expansion of alien tree cover and plantations and invasion over time. The broad-scale temporal study using moderate resolution imagery (chapter 3) revealed that forest classes are highly dynamic, especially in the lower elevation areas (valley). This high forest dynamism has been detected elsewhere in Chile by Altamirano et al. (2013). As mentioned earlier, at this scale, most of the alien patches detected are plantations. Consequently, the trajectories of land cover change are heavily influenced by anthropogenic factors. No net expansion of alien tree cover at the expense of native forest was found over the study period. It should be noted, though, that undisturbed native forests, early successional native forests, and native plantations were not separated. This means that there may still be a net loss of undisturbed native forests (converted to alien plantation) with their relatively high conservation value, while early successional native forests and native plantations increase in areal extent at the expense of other land covers. This, together with the fact that conversion to alien tree cover is accelerating and has been reported as a relevant deforestation driver in other locations (Zamorano-Elgueta et al., 2015), indicates that undisturbed native forests need to be more strictly protected to avoid severe deforestation and conversion to productive activities.

The broad-scale modelling (chapter 2) and local scale analysis (chapter 4) revealed alien Pinaceae patches within native *A. araucana* forests, indicating invasion. *A. araucana* forests are naturally open, with the trees growing at a low density with an understorey of native shrubs and grasses. Pinaceae are good competitors in open or treeless environments such as these. In fact, Pinaceae invasion has been reported in the location studied in this project by other authors (Peña et al., 2008) and in other open or treeless environments (Rundel, Dickie and Richardson, 2014, Pauchard et al., 2016). Alien species can reduce the biodiversity of a site through competition and modification of the habitat (Bolch et al., 2020). In the case of Pinaceae in *A. araucana* forests, they can create a closed canopy and prevent *A. araucaria* regeneration due

to the shadowing effect, as *A. araucana* requires full sun exposure. On the other hand, the native *Nothofagus* spp. forests are not as invaded. These forests are dense, with closed canopies and a multi layered structure. Some invasion under canopy could be occurring, especially by *Pseudotsuga menziesii*, the only relatively shade tolerant invasive tree in this area (Peña and Langdon, 2007, Pauchard, Langdon and Peña, 2008). This invasion was not observable in this study using optical remote sensing.

A process of native forest degradation and shrub encroachment was observed in the temporal study (chapter 3) and was confirmed by the very fine resolution assessment (chapter 4). Conversion from native forest to shrubs has been previously reported in nearby locations (Altamirano and Lara, 2010, Miranda et al., 2015, Zamorano-Elgueta et al., 2015). Degradation of the native forest is often attributed to selective thinning (also known as high grading) for firewood and timber by the local communities (Gea-Izquierdo et al., 2004, Altamirano and Lara, 2010, Salas et al., 2016). However, recent studies have found that periods of severe drought can trigger processes of forest degradation through mortality events and species replacement in temperate forests (Suarez and Kitzberger, 2008, Batllori et al., 2020). Severe drought periods are increasingly common as a result of climate change. In fact, between 2010 and 2015, central and south-central Chile suffered a continuous sequence of years with rainfall deficits up to 45%. This period was termed the Central Chile Mega Drought (Garreaud et al., 2017). This dry period caused complex vegetation changes and altered the fire regime (Garreaud et al., 2017, González et al., 2018). The immediate effect of these droughts was the shift from temperate (mesic) forest species to xeric species (well adapted to drier conditions) (Batllori et al., 2020). In this way, *Nothofagus* forests were converted to shrubs or other tree species (Batllori et al., 2020). An earlier study in Patagonian forests found that drought favoured native *Aristotelia chilensis* over *Nothofagus dombeyi* due to its higher ecological plasticity and tolerance to dryness (Suarez and Kitzberger, 2008). In the south-central Chilean Andes, native (mesic) forest is being converted to shrubs, either by selective thinning, as a result of severe drought or, more likely, as a result of the synergistic effect of both. In addition, it is important to highlight that many species of the *Pinus* and the *Eucalyptus* genera are xeric, as their native ranges are within arid or semi-arid regions and can withstand warm temperatures (Li and Wang, 2003, Moran et

al., 2017). Consequently, they will probably be favoured by native forest degradation and drought.

### 5.5. Alien tree cover and landscape configuration

The fourth objective was to relate the presence of alien tree stands and degree of invasion to landscape spatial configuration and fragmentation at different scales. At broad scales, the most important drivers of alien tree cover were landscape heterogeneity and elevation. Alien tree patches tend to be located in heterogeneous, fragmented and easily accessible areas. Previous studies have linked alien species presence with fragmentation and accessibility (Altamirano and Lara, 2010, Altamirano et al., 2016) in other areas of Chile. As mentioned before, at this scale, many of the alien tree patches detected are plantations. Consequently, it is intuitive that heterogeneous (with a diversity of land covers and uses) and accessible areas host most of the alien tree patches. However, a link between *A. araucana* and alien tree invasion was also found at this scale, although it had less importance than landscape heterogeneity and elevation in the models. This suggests that some advanced invasion is also detected at this scale. At local scales, early invasion was linked mainly to distance to seed sources, although topography also played a role in alien dispersal. Uphill trajectories and barriers such as roads seemed to limit dispersal. Lastly, *A. araucana* forests are more invaded than *Nothofagus* spp. forests, as reported earlier by Peña et al. (2008). This is not surprising, as *A. araucana* forests are naturally open and alien Pinaceae are good competitors in open areas (Rundel, Dickie and Richardson, 2014, Pauchard et al., 2016).

### 5.6. Management recommendations

The last objective was to provide evidence-based management recommendations to improve forest and conservation management. According to the World Wildlife Fund (WWF) Chile, in their *Strategic plan for the Valdivian ecoregion*, the extensive net of protected areas in Chile only covers 11.7% of the Valdivian temperate forest ecoregion. In addition, most of these protected areas are at high elevations (above 600 m). This means that certain forest types such as *Nothofagus* spp. forests, which are typical of lower elevations, are under protected (WWF Chile, 2011). WWF emphasizes the importance of using landscape ecology and biological conservation criteria when selecting suitable areas for protection. The land cover maps generated

in this thesis showed relatively high fragmentation and presence of alien species within the current protected areas, diminishing their conservation value. In addition, a large proportion of *A. araucana* is located outside protected areas. These results, together with the WWF Chile recommendations, suggest that protected area boundaries should be reconsidered, as some high conservation value areas seem to be unprotected. A previous study in the Chilean Andes highlighted the need to improve the system used to determine protected areas in the country (Altamirano and Lara, 2010). Satellite imagery, especially Sentinel-2, should be used for land planning and selecting new areas for protection.

Several protected areas are included within the study areas of this thesis. Their main aims according to their management plans are summarised here. Containing part of the major hydrological catchment of the Araucanía region, the Malalcahuello National Reserve has the protection of soils and the hydrological regime among its main objectives. The preservation of the protected *A. araucana* as well as the sustainable management of the *Nothofagus* spp. are also considered. The Malalcahuello National Reserve should provide cultural and recreational services for tourism and the socioeconomical development of the area (CONAF, 1996). The creation of the Villarrica National Park aimed to conserve the biodiversity, landscape, geomorphology and hydrological regime of the area (CONAF, 2013). The Tolhuaca National Park was created for biodiversity conservation, preservation and improvement of the Malleco river catchment, landscape protection, and for tourism, recreation and environmental purposes (CONAF, 2014a). The main objectives of the Conguillío National Park are the preservation of *A. araucana* and the landscape beauty, environmental protection of the area, and provision of a range of educational and recreational services (CONAF, 2006). The Huerquehue National Park has the conservation of *Nothofagus* spp. and mixed *A. araucana* and *Nothofagus* spp. forests among its main aims. It also aims to preserve the geomorphology and the Toltén river catchment. Cultural and recreational uses are also considered among the aims (CONAF, 1999). Finally, the Nalcas and China Muerta National Reserves were established to guarantee the commerce for native wood and conserve native tree species, and the landscape (CONAF, 2014b, 2014c).

Regardless of the specific objectives, all these protected areas are structured in different zones depending on the degree of protection and type of use. The *intangible* zone contains unique, fragile, and relatively pristine ecosystems or

environmental features that deserve full protection. Only non-destructive scientific and conservation activities are allowed, and vehicles are forbidden. The *primitive* zone also contains areas with unique ecosystems and environmental features that have received minimal human disturbance. These areas, however, are relatively resilient and could tolerate low intensity public use. Vehicles are prohibited. The *restoration* zone includes degraded areas or areas with presence of alien species that require specific management to stop the degradation process or to restore them. The *public use* zone includes natural or modified areas of outstanding landscape beauty or with recreational resources (ski, fishing, etc.). These areas allow vehicles and dense use, but always in a sustainable way. The *resource management* zone allows extractive activities and natural resource use, but sustainably. Aside from these main zones, other minor zones like archaeological, cultural, and special use sites can be found in protected areas (CONAF, 1996, 1999, 2006, 2013, 2014a, 2014c, 2014b). It is important to highlight that not all protected areas contain all types of zones.

Most of the *A. araucana* and mixed *A. araucana* and *Nothofagus* spp. forests are included in the intangible or primitive areas. However, this thesis has demonstrated that there is presence of Pinaceae trees within protected areas and in the proximity of *A. araucana*. This occurred mostly into high elevation, naturally open *A. araucana* forests, as most Pinaceae are adapted to grow in full sun (heliophytes). Pinaceae invasion has been reported in the same area (Peña et al., 2008) and in other open or treeless areas (Rundel, Dickie and Richardson, 2014, Pauchard et al., 2016). These Pinaceae locations should be reassessed and included in the restoration zone for active management for eradication, as they could significantly alter the structure and species composition of the pure and mixed *A. araucana* forests. Currently, restoration zones are almost exclusively used for eroded or burnt areas, even though the management of alien species is within the objectives of restoration zones. The Pinaceae trees should be removed gradually through selective thinning and accompanied by native species reforestation. Clearcuttings should never be used to remove large Pinaceae stands, as this would promote erosion and alter the hydrological regime (Salas et al., 2016).

Sometimes, resource management zones are adjacent to primitive or intangible zones. This should be avoided, as resource management zones are disturbance areas and could affect the edge of the primitive and intangible zones. For example, certain resource management zones currently contain large Pinaceae stands that could act

as seed sources for invasion into directly adjacent primitive and intangible zones. Transition or buffer zones should be placed between primitive or intangible and resource management zones to minimise impact on the most sensitive and pristine ecosystems of the primitive and intangible zones. Perhaps, public use zones could also be used for this buffering purpose.

Considering the findings in this thesis, Sentinel-2 is recommended for forest inventory and continuous monitoring of deforestation, forest degradation, and conversion to alien forest cover. This imagery accurately detects plantations and heavily invaded areas, which can be very useful for invasion risk modelling (Shaw, 2005). Sentinel-2 data could help in identifying high risk and high conservation areas, such as the naturally open *A. araucana* forests, degraded sites and native forest near productive alien plantations. These areas should be prioritised for targeted alien species monitoring and control. VHR imagery could be acquired for these priority areas to detect early invasions and eradicate them swiftly and effectively (Pauchard et al., 2016, Nuñez et al., 2017). Remote sensing should inform land management to place alien forestry plantations only in low risk areas.

Forestry plantations should employ sustainable management practices, especially in the surrounds of protected areas. First, a limitation on the size of clear-cut areas should be established to lessen the environmental impact of intensive forestry on the soil, the hydrological regime and biodiversity, as previously suggested by Salas et al. (2016). Compliance with these clear-cut area limits could be easily assessed using remote sensing. Second, mixed species plantations should be used instead of monocultures, as they provide a more heterogeneous and structurally complex habitat that enhances biodiversity and act as a permeable matrix between undisturbed native forest patches, increasing habitat connectivity (Hartley, 2002, Díaz et al., 2005, Brockerhoff et al., 2008, Lindenmayer, Hobbs and Salt, 2013). This is especially important for the valley (low elevation) areas, which are highly fragmented and contain a low proportion of undisturbed native forest according to the results of this thesis. Third, islands of native forest within the productive forestry matrix of the valley should be protected to enhance biodiversity conservation (Horák et al., 2019). Finally, undermanaged monoculture forestry plantations are a fire hazard. Their high density and flammability, coupled with rising temperatures and decreased rainfall, could trigger extreme wildfires (Gómez-González, Ojeda and Fernandes, 2018). Consequently, they should be forbidden near protected areas and



appropriate buffer areas should be established around them. These buffer areas should have a reduced vertical and horizontal biomass continuity to stop fire spread. The reduction of biomass could be achieved through pruning, thinning, or using grazing animals (Lovreglio, Meddour-Sahar and Leone, 2014, Mena et al., 2016). Avoiding catastrophic fires is especially important in the Valdivian temperate forest, as the native tree species are not as well adapted to post-fire regeneration and resprouting as Pinaceae and *Eucalyptus* spp. are (Drake, Molina and Herrera, 2012, Kitzberger et al., 2016).

The end of forestry subsidies in 2012 marked a turning point for Chilean forestry, especially for smallholders. Small plantations have ceased to be profitable, and smallholders are shifting to short rotation forestry or other land uses that provide a more stable income. Mixed species native *Nothofagus* spp. plantations are recommended as an alternative for smallholders. Many Chilean communities depend on firewood for heating during winter and other uses, and native *Nothofagus* spp. forests are severely degraded due to uncontrolled selective thinning (Gea-Izquierdo et al., 2004, Altamirano et al., 2010). *Nothofagus* spp. plantations would take the pressure away from native forests, which should be more strictly protected, and provide a sustainable income for these smallholders. In addition, sustainable *Nothofagus* spp. forestry would provide habitat for wildlife and improve native forest connectivity throughout the landscape. Agroforestry or silvopastoral systems using *Nothofagus* spp. are also recommended. These systems combine agriculture, forestry and livestock raising, using ecosystem resources more efficiently and being more environmentally sustainable (Rigueiro-Rodríguez et al., 2008). Also, the combination of activities increases income stability for the farmer. Agroforestry systems with *Nothofagus* spp. are already used in parts of Patagonia, and have reached higher pasture productivity than open grassland systems (Sánchez-Jardón et al., 2010, Gargaglione, Peri and Rubio, 2014, Peri et al., 2016).

## 5.7. Limitations and future directions

Despite the great advantages of remote sensing methods for alien species mapping and modelling, there are also some limitations. First, freely available moderate spatial resolution imagery is not able to detect early invasion, although it can be useful for broad scale management and targeting priority areas. VHR imagery can detect early invasion, but its cost is a key limitation, especially given the lack of

funding for conservation in Chile (Waldron et al., 2013). Initiatives such as Planet's public-private partnership to provide VHR imagery for forest monitoring will be key to overcome these funding limitations (O'Shea, 2020). In addition, drones, also known as UAVs (unmanned aerial vehicles), also hold great potential to overcome this limitation, as they can provide VHR imagery of local areas at low cost (Grenzdörffer, Engel and Teichert, 2008, Paneque-Gálvez et al., 2014, Tang and Shao, 2015). Future research should investigate the ability of drone imagery to separate different types of native and alien forest covers. Another common constraint for remote sensing is computing power, as image analyses are resource-intensive processes. Nowadays, however, this limitation is offset via powerful cloud computing platforms such as GEE (Gorelick et al., 2017) that only require an Internet connection to perform complex image analyses using very large data sets. The availability of reliable ground reference data is another limitation for remote sensing, especially when making historical maps. Familiarity with the field, expert photointerpretation skills and ancillary datasets such as aerial imagery and virtual globe image archives such as Google Earth can help in this regard.

The detection of early invasions using very fine resolution land cover maps was limited to targeted, relatively small areas, as image analyses using VHR data are computationally intensive, even using cloud computing platforms. In addition, the use of optical remote sensing imagery limited the detection to only the first layer of vegetation (the top layers of the forest canopy). Some invasion can occur in the understorey of the dense, multi-layered *Nothofagus* spp. forests, but this is undetectable with the methods used in this research project. This is important because one of the alien Pinaceae species, *Pseudotsuga menziesii*, can regenerate under canopy (Peña and Langdon, 2007, Pauchard, Langdon and Peña, 2008). Future work should investigate the potential of active sensors such as light detection and ranging (LiDAR) or synthetic aperture radar (SAR) for understorey alien species detection, which is still relatively uncommon (Bolch et al., 2020). The use of moderate resolution imagery such as Sentinel-2 or Landsat is also limited by its inability to separate alien plantations from advanced alien invasion when the precise coordinates of the plantations are not known. Future investigations should try to overcome this limitation by including shape, texture or ancillary datasets in the analyses. Hyperspectral imagery could be tested for individual Pinaceae and

*Eucalyptus* species identification, providing species level information on invasiveness of different productive forestry species.

VHR imagery has proven to be very effective for early invasion detection. Future work should use these datasets to investigate the drivers of invasion in greater detail, including environmental variables such as slope, aspect or dominant wind direction. Combinations of remote sensing data could also help understand the drivers of invasion. Early invasion detected with VHR imagery could be coupled with the forest structural characteristics estimated using LiDAR data (Sumnall, Hill and Hinsley, 2016, Marselis et al., 2020) to model the effect of canopy structure on invasion. A deeper understanding of invasion drivers could enable predictive modelling, which would be extremely valuable for conservation and alien species control. Finally, regarding forest management, future work should aim to optimise the production using sustainable systems such as agroforestry or mixed species forestry, instead of large-scale monocultures.

The land cover maps and models generated in this thesis, could be directly used to select high invasion risk areas and improve protected area management using geospatial analysis. According to the results obtained in this thesis, open *A. araucana* forests with nearby presence of alien tree stands should be prioritised as high invasion risk areas. Also, the assessments suggest that fragmented landscapes with presence of alien species should also be considered high invasion risk areas. High forested land cover dynamism is often a result of human intervention (plantation establishment and harvesting). Consequently, these highly dynamic areas should also be considered high risk. Finally, any large alien patch within the boundaries of a protected area should be closely monitored, as the analysis in chapter 4 detected rapid Pinaceae invasion within a native forest matrix. Invasion within protected areas would undermine their objectives and conservation value.

## 5.8. Contribution of this research

The fundamental relevance of this doctoral research project is twofold. First, scientifically, a greater understanding of alien tree species spatial distribution and invasion has been achieved, contributing to the advancement of invasion science in Chile and South America, where biological invasions have traditionally been understudied. This project has corroborated that alien tree presence and invasion is linked to landscape fragmentation. It has also been demonstrated that

naturalisation and invasion can occur in relatively undisturbed forests if these have the environmental requirements for the survival and reproduction of the alien species. In this case, the main requirement is light availability, a condition fulfilled by the high mountain *A. araucana* forests. Consequently, a concern about Pinaceae invasions into the native *A. araucana* forest has been raised, as the long-term regeneration of this endemic and culturally relevant tree could be compromised. In addition, an ongoing native forest degradation process has been identified in the area.

Second, from the forest management perspective, the information provided in this thesis can be a cornerstone for the design of new policies and management strategies that minimise the risk of invasion stemming from productive plantations. This way, the highly profitable Chilean forestry industry could become more sustainable, as should be required for a global biodiversity hotspot. The methods adopted and developed here could be used to enhance and update traditional forest surveys in a relatively simple and effective way. This could help in designing successful conservation and invasion control strategies in Chile and elsewhere.

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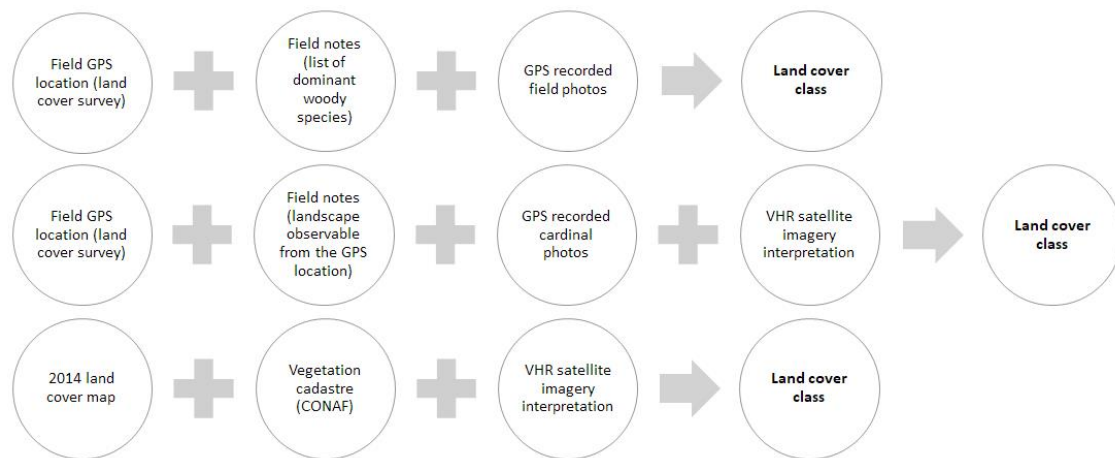
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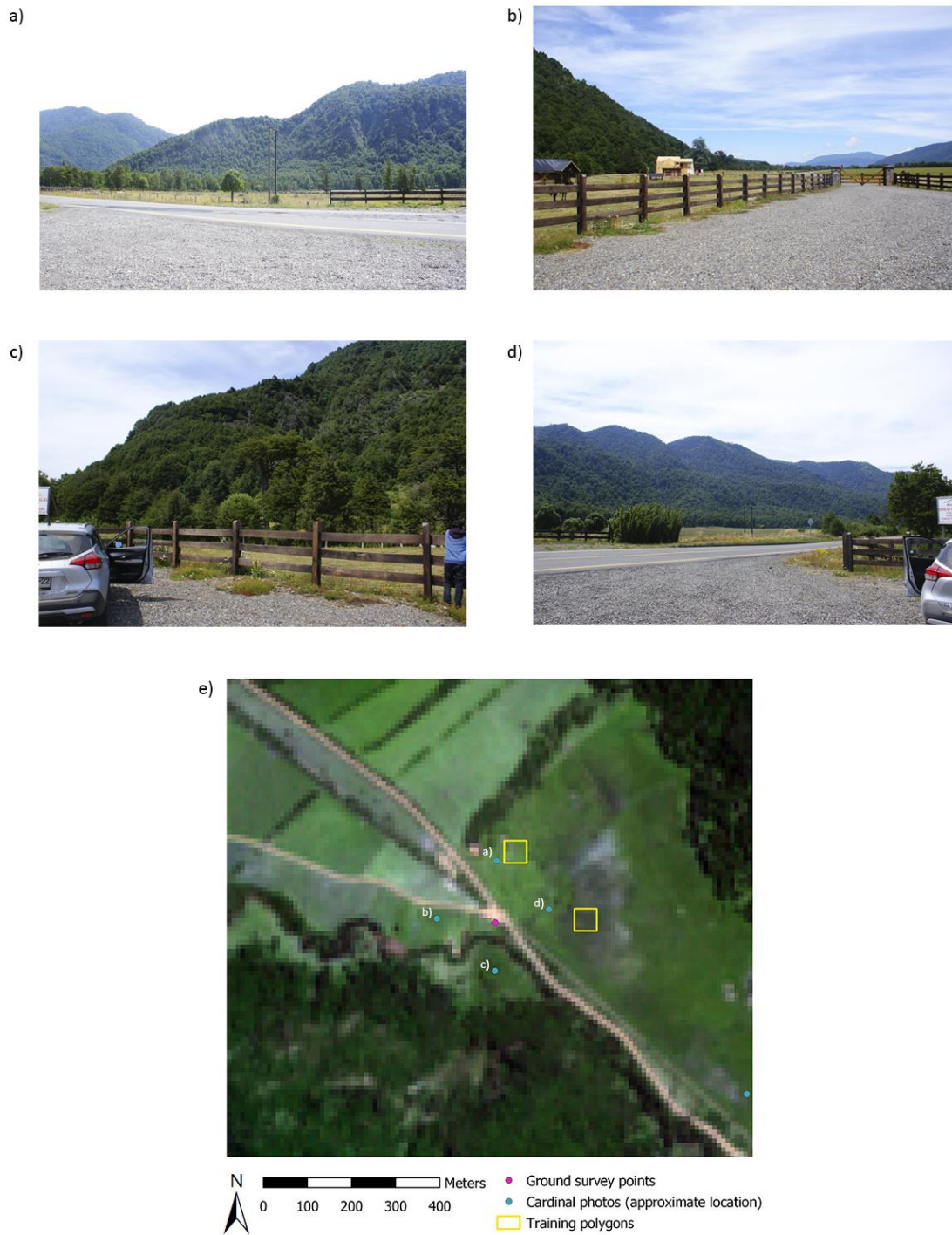
## Appendices

## Appendix A

### Land cover survey and classification details



**Figure A1** Dataset ensembles used to determine qualitative land cover classes from the different datasets. To determine a land cover class in a location, all the datasets within an ensemble need to agree. Depending on the location and data availability, a different ensemble was employed.



**Figure A2** Example of cardinal photos, ground survey locations, and training polygons (grassland): a) north facing photo, b) west facing photo, c) south facing photo, d) east facing photo, and e) location map and training polygons.



**Figure A3** Example of cardinal and canopy cover photos, ground survey locations, and training polygons (native *Nothofagus* spp. forest): a) north facing photo, b) west facing photo, c) south facing photo, d) east facing photo, e) canopy cover photo, and f) location map and training polygons.

## Appendix B

### Confusion matrices

**Table B1** Confusion matrix for the winter composite for Malalcahuello. 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, PA = producer's accuracy and UA = user's accuracy.

	CP	BP	NNF	ANF	Shrubs	Grass	Water	Bare	Snow	Urban	TOTAL	UA
CP	47	1	0	1	0	0	0	0	0	0	49	0.96
BP	2	38	0	1	0	2	0	0	0	0	43	0.88
NNF	10	5	44	33	12	2	5	0	0	0	111	0.40
ANF	1	2	11	34	8	0	1	0	0	0	57	0.60
Shrubs	0	1	5	14	20	21	0	0	0	0	61	0.33
Grass	0	0	1	2	3	36	0	0	0	0	42	0.86
Water	0	0	2	0	0	0	36	0	0	0	38	0.95
Bare	0	0	0	4	1	0	5	28	0	0	38	0.74
Snow	0	0	0	4	3	0	0	0	46	0	53	0.87
Urban	0	0	0	3	0	2	0	0	0	42	47	0.89
TOTAL	60	47	63	96	47	63	47	28	46	42	539	
PA	0.78	0.81	0.70	0.35	0.43	0.57	0.77	1	1	1		

Overall	0.69											
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**Table B2** Confusion matrix for the summer composite for Malalcahuello. 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, PA = producer's accuracy and UA = user's accuracy.

	CP	BP	NNF	ANF	Shrubs	Grass	Water	Bare	Snow	Urban	TOTAL	UA
CP	44	1	0	0	0	0	0	0	0	0	45	0.98
BP	4	36	0	0	0	0	0	0	0	0	40	0.90
NNF	11	8	53	38	5	0	0	0	0	0	115	0.46
ANF	0	2	10	43	4	0	0	0	0	0	59	0.73
Shrubs	1	0	0	14	38	15	0	0	0	0	68	0.56
Grass	0	0	0	0	0	48	0	0	0	0	48	1.00
Water	0	0	0	1	0	0	39	0	0	0	40	0.98
Bare	0	0	0	0	0	0	8	28	0	0	36	0.78
Snow	0	0	0	0	0	0	0	0	46	0	46	1.00
Urban	0	0	0	0	0	0	0	0	0	42	42	1.00
TOTAL	60	47	63	96	47	63	47	28	46	42	539	
PA	0.73	0.77	0.84	0.45	0.81	0.76	0.83	1	1	1		
Overall	0.77											

**Table B3** Confusion matrix for the multi-season composite for Malalcahuello. 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, PA = producer's accuracy and UA = user's accuracy.

	CP	BP	NNF	ANF	Shrubs	Grass	Water	Bare	Snow	Urban	TOTAL	UA
CP	43	4	0	3	0	0	0	0	0	0	50	0.86
BP	3	35	0	3	0	0	0	0	0	0	41	0.85
NNF	13	5	58	36	2	0	0	0	0	0	114	0.51
ANF	1	3	3	40	6	0	0	0	0	0	53	0.75
Shrubs	0	0	2	10	36	11	0	0	0	0	59	0.61
Grass	0	0	0	0	1	52	0	0	0	0	53	0.98
Water	0	0	0	1	0	0	40	0	0	0	41	0.98
Bare	0	0	0	3	2	0	7	28	0	0	40	0.70
Snow	0	0	0	0	0	0	0	0	46	0	46	1.00
Urban	0	0	0	0	0	0	0	0	0	42	42	1.00
TOTAL	60	47	63	96	47	63	47	28	46	42	539	
PA	0.72	0.74	0.92	0.42	0.77	0.83	0.85	1	1	1		
Overall	0.78											

**Table B4** Confusion matrix for the multi-season with DEM composite for Malalcahuello. 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, PA = producer's accuracy and UA = user's accuracy.

	CP	BP	NNF	ANF	Shrubs	Grass	Water	Bare	Snow	Urban	TOTAL	UA
CP	47	6	0	0	0	0	0	0	0	0	53	0.89
BP	3	36	0	0	1	0	0	0	0	0	40	0.90
NNF	10	5	57	23	3	0	0	0	0	0	98	0.58
ANF	0	0	6	58	11	0	0	0	0	0	75	0.77
Shrubs	0	0	3	14	32	11	0	0	0	0	60	0.53
Grass	0	0	0	0	0	51	0	0	0	0	51	1
Water	0	0	0	1	0	0	40	0	0	0	41	0.98
Bare	0	0	0	0	0	0	7	28	0	0	35	0.80
Snow	0	0	0	0	0	0	0	0	46	0	46	1
Urban	0	0	0	0	0	1	0	0	0	42	43	0.98
TOTAL	60	47	66	96	47	63	47	28	46	42	542	
PA	0.78	0.77	0.86	0.60	0.68	0.81	0.85	1	1	1		
Overall	0.81											

**Table B5** Confusion matrix for the winter composite for Villarrica. 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, PA = producer's accuracy and UA = user's accuracy.

	CP	BP	NNF	ANF	Shrubs	Grass	Water	Bare	Snow	Urban	TOTAL	UA
CP	47	5	5	1	1	0	0	0	0	0	59	0.80
BP	6	9	0	1	0	0	0	0	0	0	16	0.56
NNF	3	1	112	13	21	9	2	2	0	0	163	0.69
ANF	0	0	10	13	0	0	0	0	0	0	23	0.57
Shrubs	2	0	13	6	21	22	0	3	0	2	69	0.30
Grass	0	0	0	0	1	65	0	0	0	0	66	0.98
Water	0	0	0	0	0	0	52	0	0	0	52	1
Bare	0	0	2	1	2	1	0	28	1	0	35	0.80
Snow	0	0	0	0	2	0	0	0	40	0	42	0.95
Urban	0	0	0	0	0	0	0	5	0	39	44	0.89
TOTAL	58	15	142	35	48	97	54	38	41	41	569	
PA	0.81	0.60	0.79	0.37	0.44	0.67	0.96	0.74	0.98	0.95		
Overall	0.75											

**Table B6** Confusion matrix for the summer composite for Villarrica. 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, PA = producer's accuracy and UA = user's accuracy.

	CP	BP	NNF	ANF	Shrubs	Grass	Water	Bare	Snow	Urban	TOTAL	UA
CP	46	1	2	0	0	0	0	0	0	0	49	0.94
BP	4	11	5	0	0	0	0	0	0	0	20	0.55
NNF	5	3	125	16	2	0	0	2	0	0	153	0.82
ANF	0	0	7	18	0	0	0	0	0	0	25	0.72
Shrubs	2	0	3	1	45	24	0	6	0	1	82	0.55
Grass	0	0	0	0	0	73	0	0	0	0	73	1.
Water	0	0	0	0	0	0	50	0	0	0	50	1
Bare	0	0	0	0	1	0	4	24	1	0	30	0.80
Snow	0	0	0	0	0	0	0	2	37	0	39	0.95
Urban	1	0	0	0	0	0	0	4	3	40	48	0.83
TOTAL	58	15	142	35	48	97	54	38	41	41	569	
PA	0.79	0.73	0.88	0.51	0.94	0.75	0.93	0.63	0.90	0.98		
Overall	0.82											

**Table B7** Confusion matrix for the multi-season composite for Villarrica. 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, PA = producer's accuracy and UA = user's accuracy.

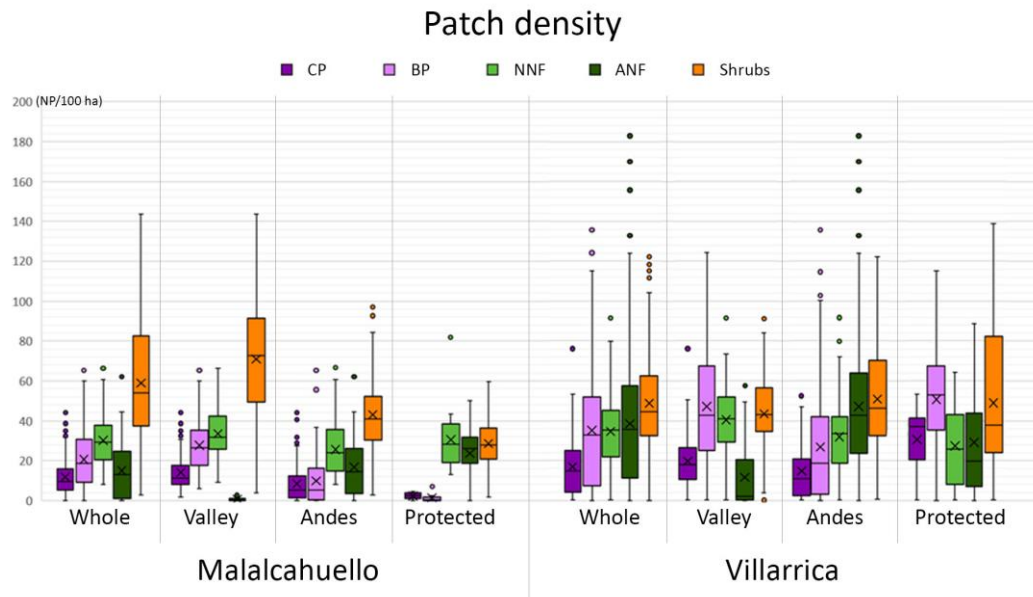
	CP	BP	NNF	ANF	Shrubs	Grass	Water	Bare	Snow	Urban	TOTAL	UA
CP	49	4	2	0	0	0	0	0	0	0	55	0.89
BP	3	10	2	1	0	0	0	0	0	0	16	0.63
NNF	3	1	127	14	7	0	0	2	0	0	154	0.82
ANF	0	0	8	19	0	0	0	0	0	0	27	0.70
Shrubs	2	0	2	1	38	19	0	5	0	0	67	0.57
Grass	0	0	0	0	1	78	0	0	0	0	79	0.99
Water	0	0	0	0	0	0	54	0	0	0	54	1
Bare	0	0	1	0	2	0	0	25	0	0	28	0.89
Snow	0	0	0	0	0	0	0	0	40	0	40	1
Urban	1	0	0	0	0	0	0	6	1	41	49	0.84
TOTAL	58	15	142	35	48	97	54	38	41	41	569	
PA	0.84	0.67	0.89	0.54	0.79	0.80	1	0.66	0.98	1		
Overall	0.85											

**Table B8** Confusion matrix for the multi-season with DEM composite for Villarrica. 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, PA = producer's accuracy and UA = user's accuracy.

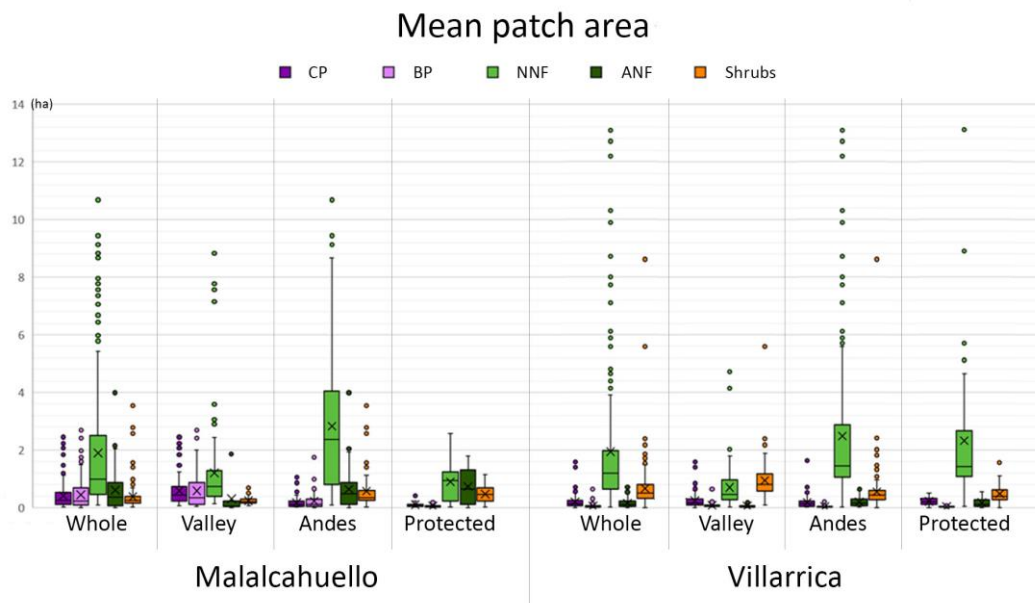
	CP	BP	NNF	ANF	Shrubs	Grass	Water	Bare	Snow	Urban	TOTAL	UA
CP	44	2	2	0	0	0	0	0	0	0	48	0.92
BP	8	11	5	0	0	0	0	0	0	0	24	0.46
NNF	2	2	121	12	5	0	1	2	0	0	145	0.83
ANF	0	0	5	23	0	0	0	0	0	0	28	0.82
Shrubs	3	0	8	0	39	12	1	4	0	0	67	0.58
Grass	0	0	0	0	0	85	0	0	0	0	85	1
Water	0	0	0	0	0	0	52	0	0	0	52	1
Bare	1	0	1	0	4	0	0	30	0	0	36	0.83
Snow	0	0	0	0	0	0	0	0	41	0	41	1
Urban	0	0	0	0	0	0	0	2	0	41	43	0.95
TOTAL	58	15	142	35	48	97	54	38	41	41	569	
PA	0.76	0.73	0.85	0.66	0.81	0.88	0.96	0.79	1	1		
Overall	0.86											

## Appendix C

### Landscape metrics distributions



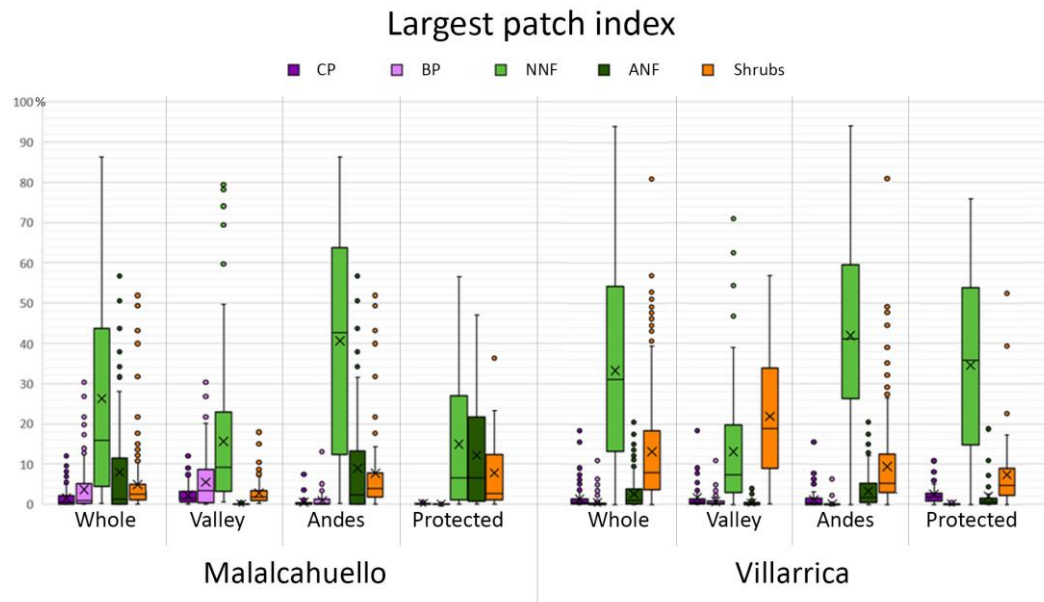
**Figure C1** Patch density values for each study area and subset. In the image, NP/100 ha stands for number of patches per 100 hectares, whole is the complete study area, valley is the low elevation subset, Andes is the high elevation subset and protected corresponds to the protected areas. The land cover classes are abbreviated as follows: CP = coniferous plantations, BP = broadleaved plantations, NNF = *Nothofagus* spp. native forests, ANF = native *A. araucana* forest.



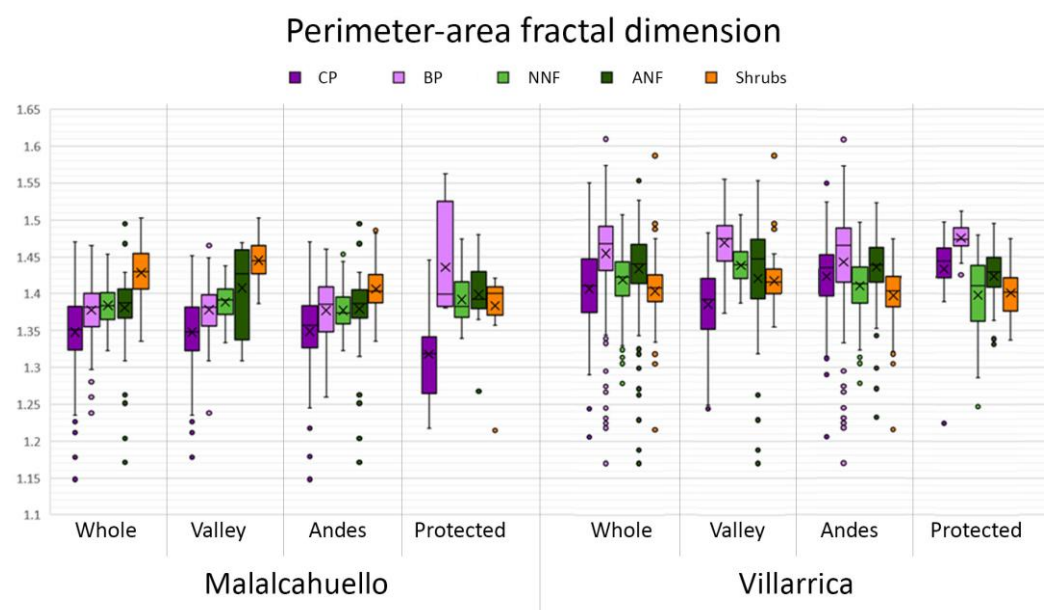
**Figure C2** Mean patch area for each study area and subset. In the image, whole is



the complete study area, valley is the low elevation subset, Andes is the high elevation subset and protected corresponds to the protected areas. The land cover classes are abbreviated as follows: CP = coniferous plantations, BP = broadleaved plantations, NNF = *Nothofagus* spp. native forests, ANF = native *A. araucana* forest.

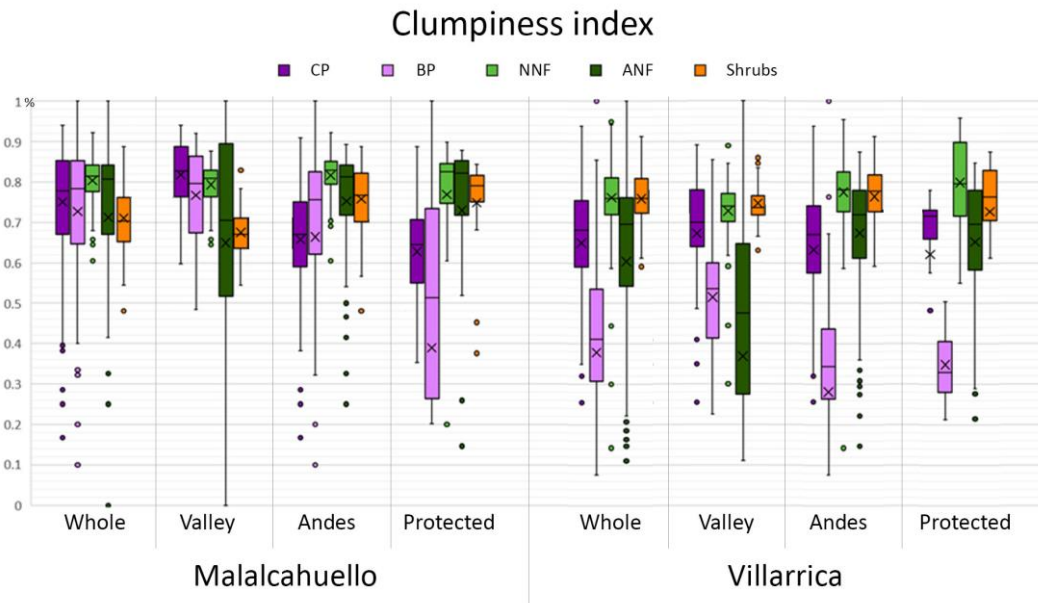


**Figure C3** Largest patch index for each study area and subset. In the image, whole is the complete study area, valley is the low elevation subset, Andes is the high elevation subset and protected corresponds to the protected areas. The land cover classes are abbreviated as follows: CP = coniferous plantations, BP = broadleaved plantations, NNF = *Nothofagus* spp. native forests, ANF = native *A. araucana* forest.

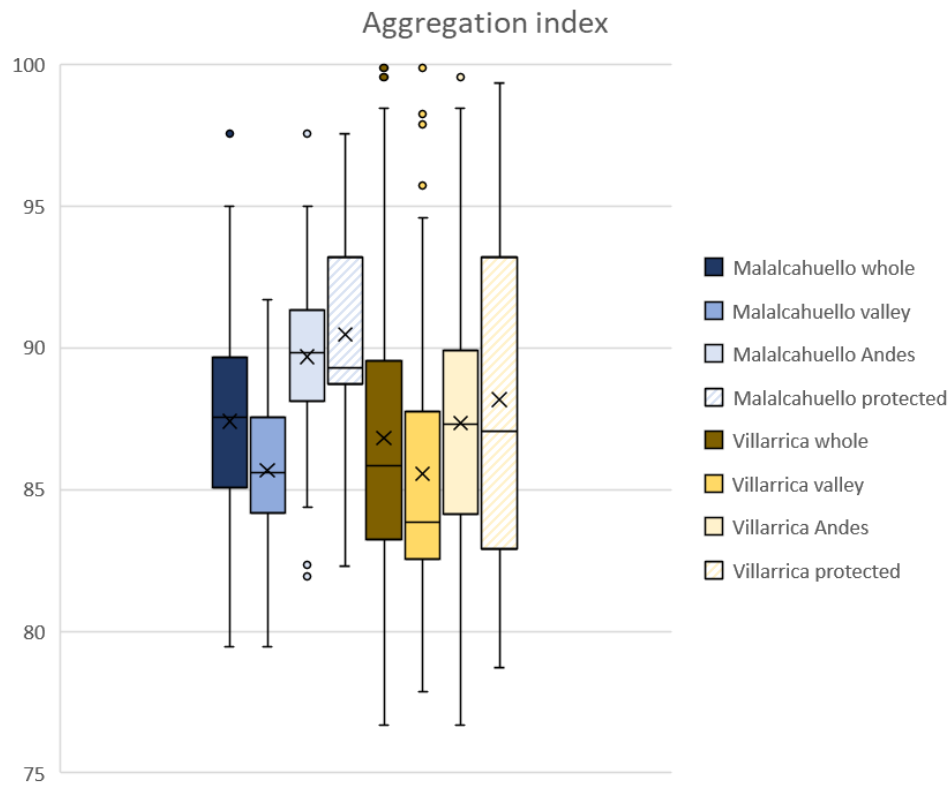


**Figure C4** Perimeter-area fractal dimension for each study area and subset. In the

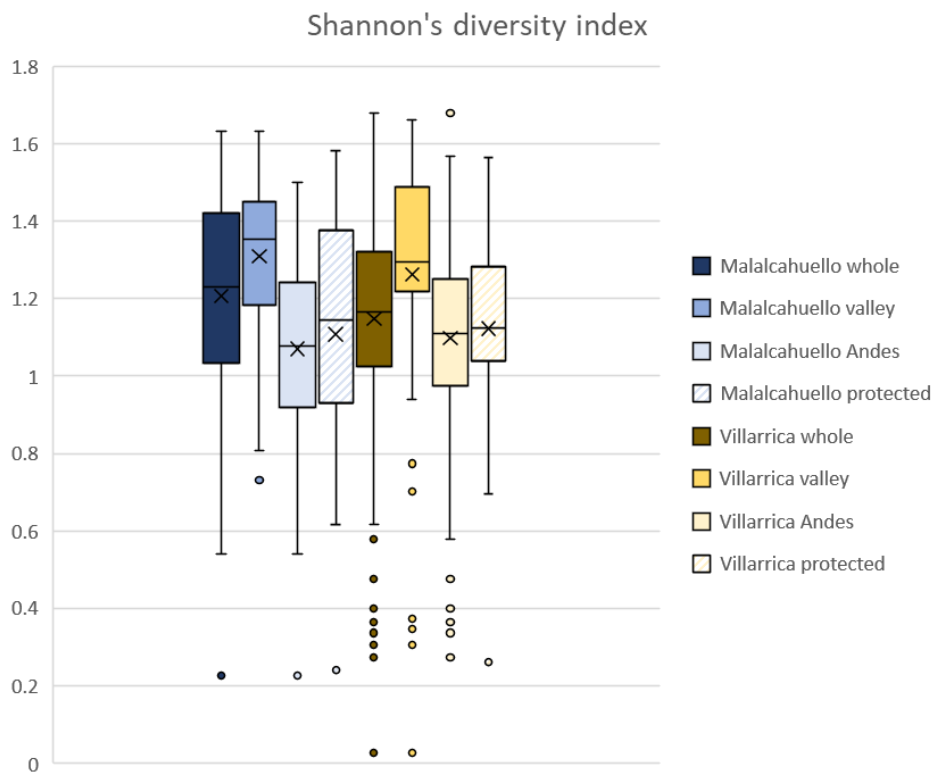
image, whole is the complete study area, valley is the low elevation subset, Andes is the high elevation subset and protected corresponds to the protected areas. The land cover classes are abbreviated as follows: CP = coniferous plantations, BP = broadleaved plantations, NNF = *Nothofagus* spp. native forests, ANF = native *A. araucana* forest.



**Figure C5** Clumpiness index for each study area and subset. In the image, whole is the complete study area, valley is the low elevation subset, Andes is the high elevation subset and protected corresponds to the protected areas. The land cover classes are abbreviated as follows: CP = coniferous plantations, BP = broadleaved plantations, NNF = *Nothofagus* spp. native forests, ANF = native *A. araucana* forest.



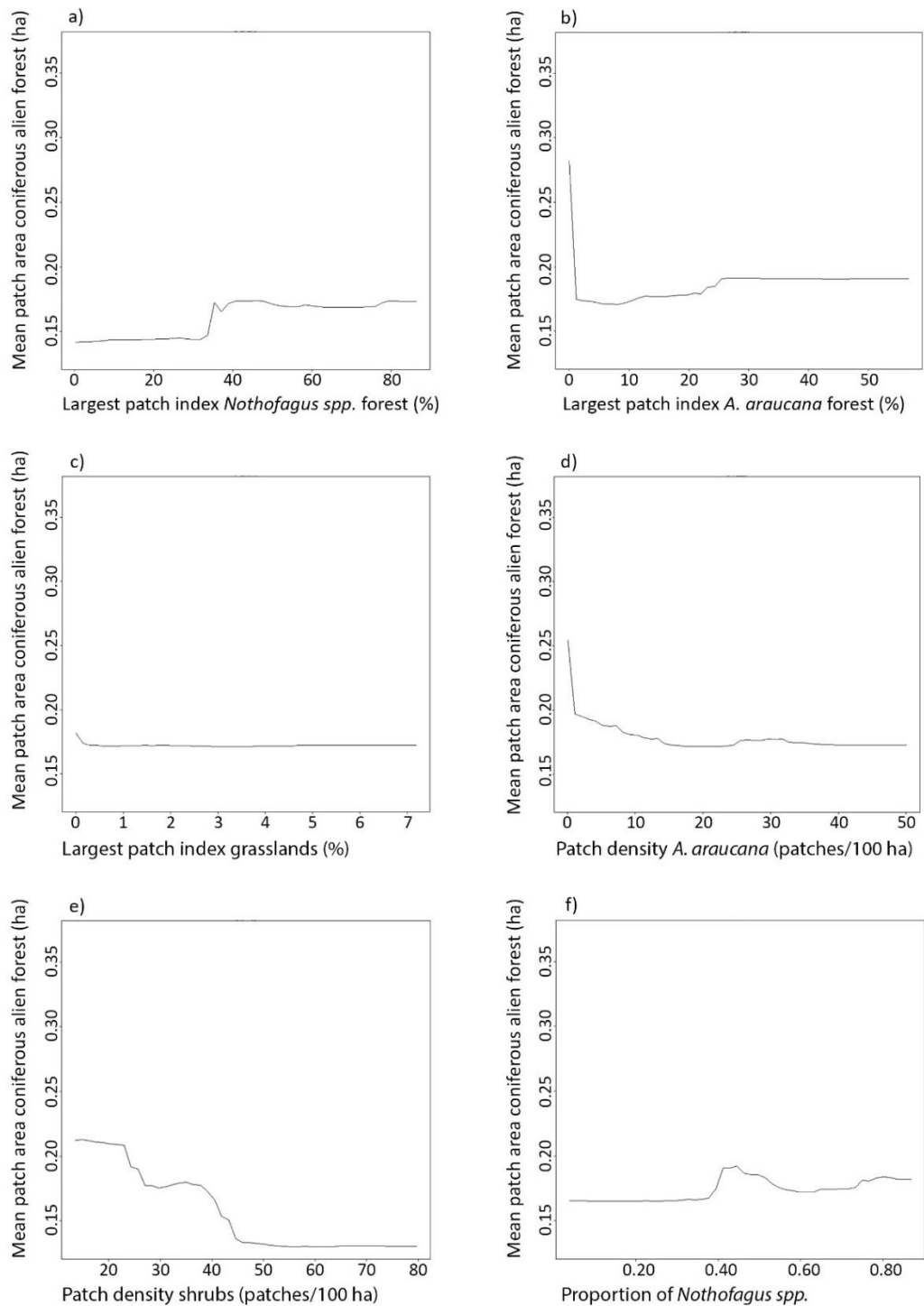
**Figure C6** Aggregation index for each study area and subset.



**Figure C7** Shannon's diversity index for each study area and subset.

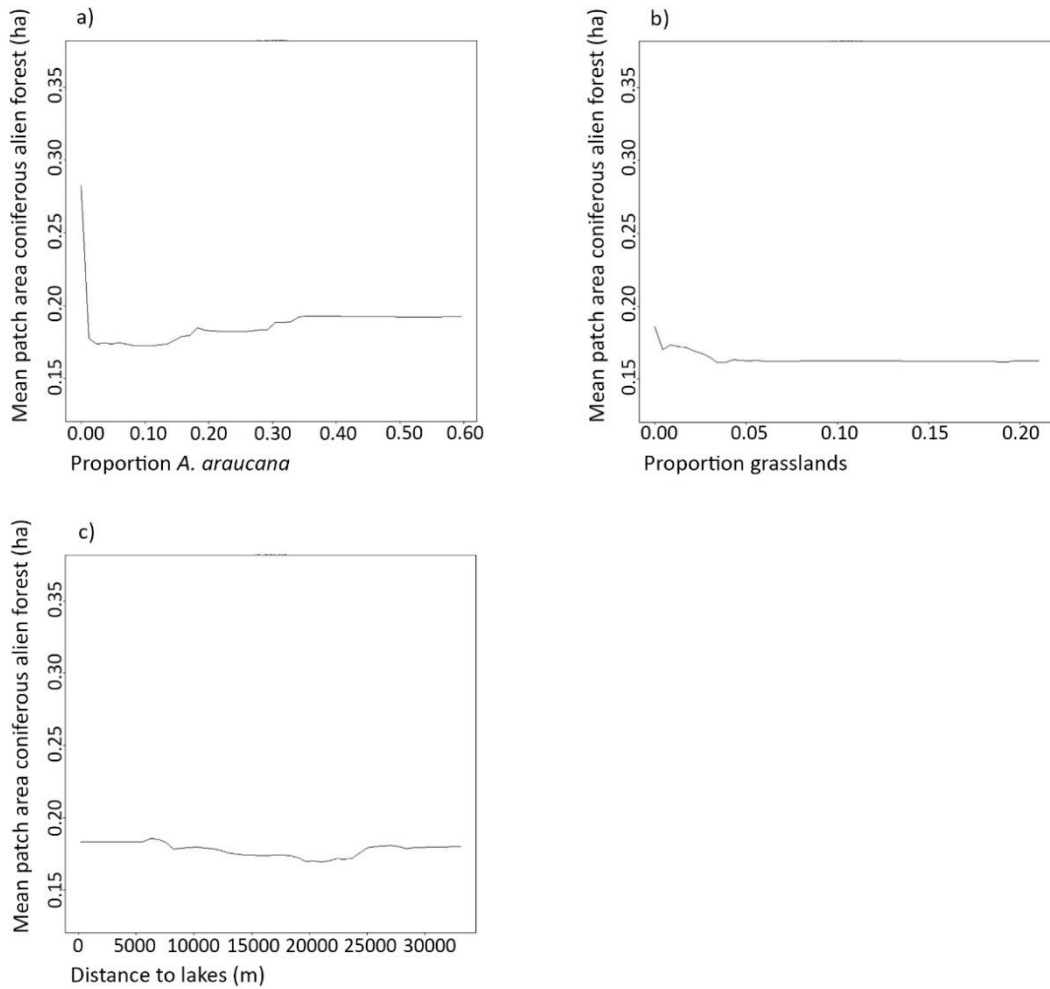
## Appendix D

Random Forest partial dependence plots for the Malalcahuello study area



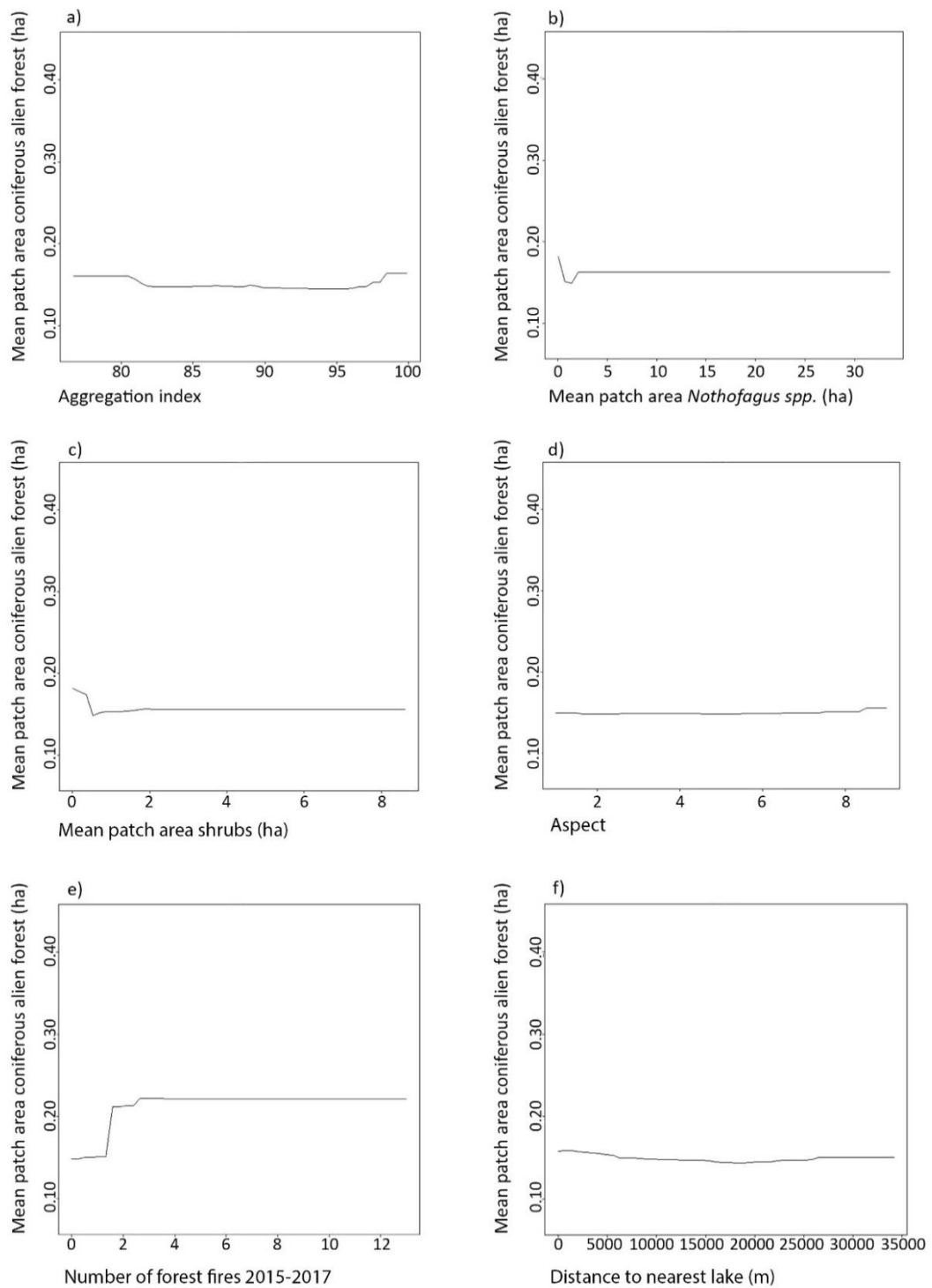
**Figure D1** Partial dependence plots for the following explanatory variables of the Malalcahuello model: a) Largest patch index of *Nothofagus spp.* forest, b) largest patch index of *A. araucana* forest, c) largest patch index of grasslands, d) patch

density of *A. araucana*, e) patch density of shrubs, and f) proportion of *Nothofagus* spp.

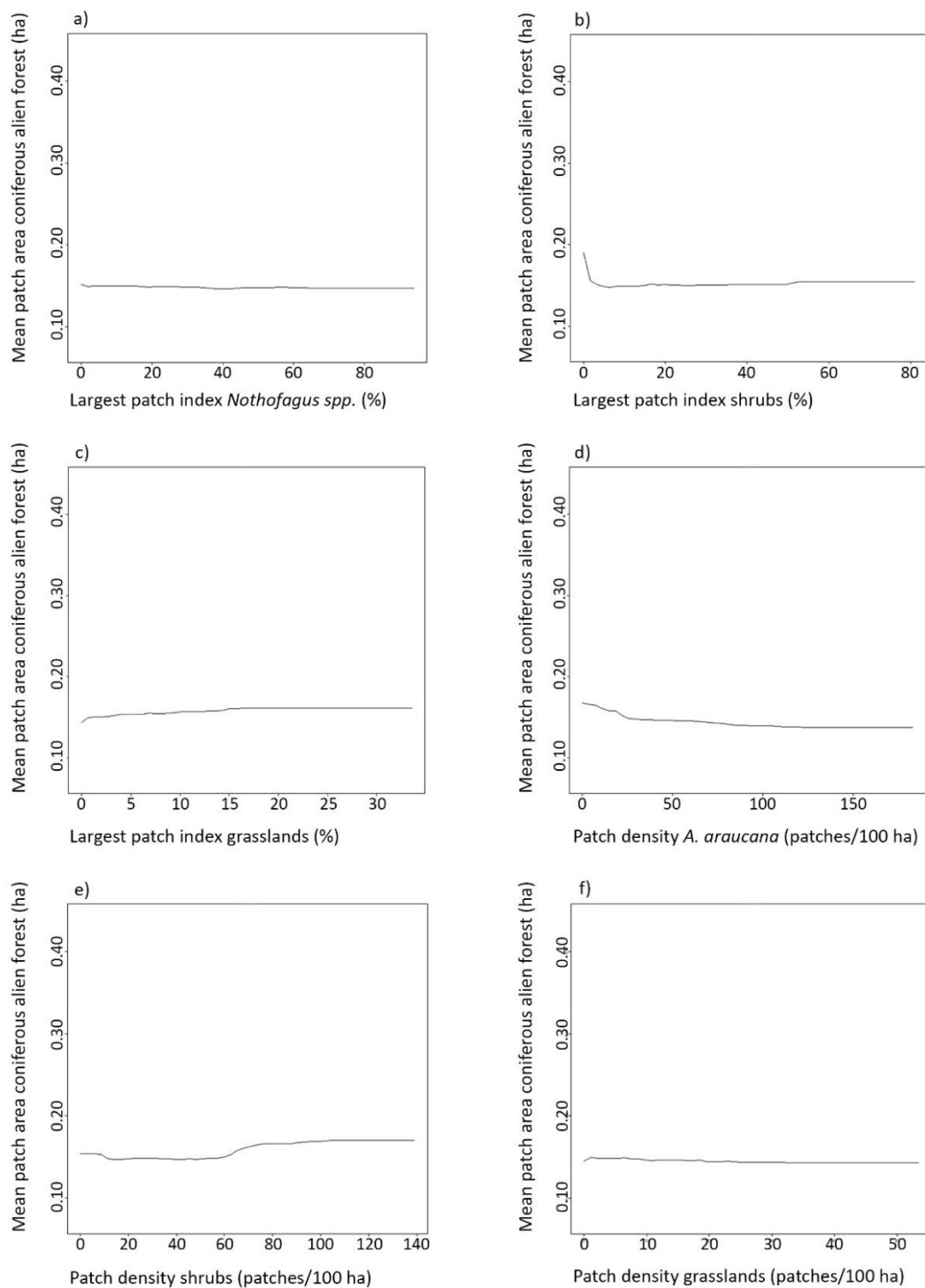


**Figure D2** Partial dependence plots for the following explanatory variables of the Malalcahuello model: a) Proportion of *A. araucana*, b) proportion of grasslands, and c) distance to lakes.

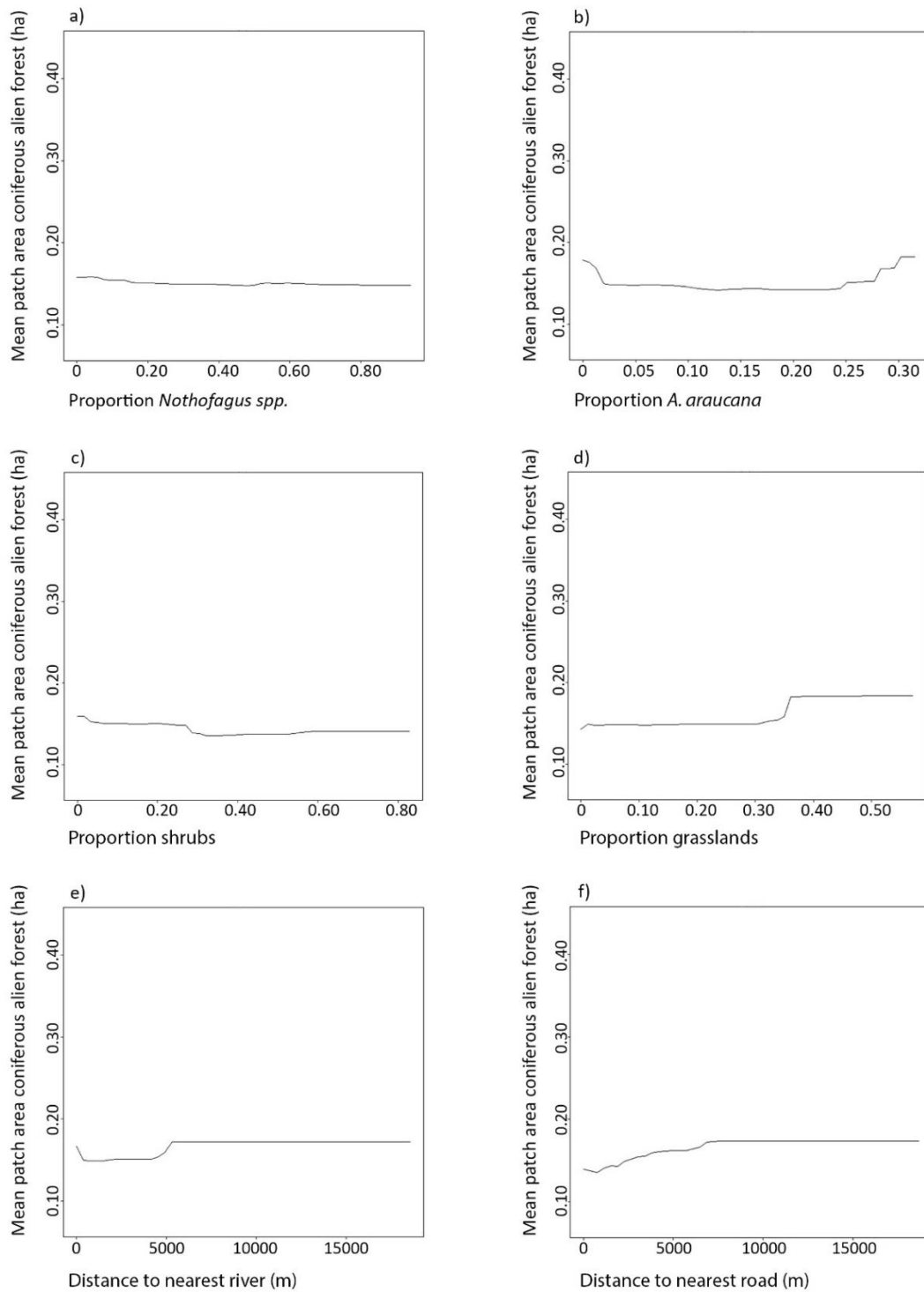
Random Forest partial dependence plots for the Villarrica study area.



**Figure D3** Partial dependence plots for the following explanatory variables of the Villarrica model: a) Aggregation index, b) mean patch area of *Nothofagus* spp., c) mean patch area of shrubs, d) aspect, e) number of forest fires 2015-2017, and f) distance to nearest lake.

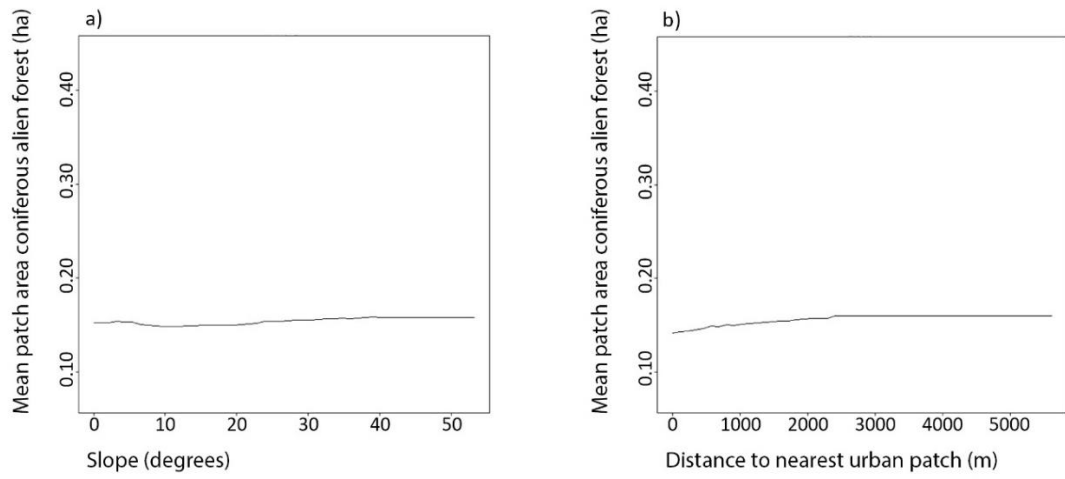


**Figure D4** Partial dependence plots for the following explanatory variables of the Villarrica model: a) Largest patch index of *Nothofagus* spp., b) largest patch index of shrubs, c) largest patch index of grasslands, d) patch density of *A. araucana*, e) patch density of shrubs, and f) patch density of grasslands.



**Figure D5** Partial dependence plots for the following explanatory variables of the Villarrica model: a) Proportion of *Nothofagus* spp., b) proportion of *A. araucana*, c) proportion of shrubs, d) proportion of grasslands, e) distance to nearest river, and f) distance to nearest road.





**Figure D6** Partial dependence plots for the following explanatory variables of the Villarrica model: a) Slope, and b) distance to nearest urban patch.

## Appendix E

**Table E1** Arbitrary values used for the change detection analysis. For each pair of images, the values of the newer image were subtracted from the values of the older image to assess change.

Land cover class	Values older image	Values newer image
Alien plantations	100	100
Native woody species	1200	200
Grasslands	2410	410
Water	3600	600
Bare	4700	700
Snow	5800	800
Urban	6900	900

## Appendix F

### Confusion matrices Landsat time series study

**Table F1** Confusion matrix for the 1980s composite. 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, PA = producer's accuracy and UA = user's accuracy.

	CP	BP	NNF	ANF	Shrubs	Grass	Water	Bare	Snow	Urban	TOTAL	UA
CP	24	0	1	0	0	2	0	0	0	3	30	0.80
BP	6	8	0	0	0	0	0	0	0	3	17	0.47
NNF	0	0	50	0	4	0	0	0	0	0	54	0.93
ANF	0	0	0	42	3	0	0	9	0	0	54	0.78
Shrubs	0	0	0	2	36	0	0	0	0	0	38	0.95
Grass	1	1	0	0	1	28	0	0	0	0	31	0.90
Water	0	0	0	0	0	0	16	2	0	0	18	0.89
Bare	0	0	0	0	0	0	1	23	0	0	24	0.96
Snow	0	0	0	1	0	0	0	1	22	0	24	0.92
Urban	0	0	0	0	0	0	0	0	0	10	10	1.00
TOTAL	31	9	51	45	44	30	17	35	22	16	300	
PA	0.77	0.89	0.98	0.93	0.82	0.93	0.94	0.66	1.00	0.63		
Overall	0.86											

**Table F2** Confusion matrix for the 1990s composite. 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, PA = producer's accuracy and UA = user's accuracy.

	CP	BP	NNF	ANF	Shrubs	Grass	Water	Bare	Snow	Urban	TOTAL	UA
CP	58	11	0	0	0	1	0	0	0	0	70	0.83
BP	6	24	1	1	0	0	0	0	0	0	32	0.75
NNF	0	2	42	8	9	0	0	0	0	0	61	0.69
ANF	0	0	1	23	3	0	0	2	0	0	29	0.79
Shrubs	0	0	0	5	31	5	0	2	0	0	43	0.72
Grass	2	2	0	0	3	23	0	0	0	1	31	0.74
Water	0	0	0	0	0	0	26	1	0	0	27	0.96
Bare	0	0	0	0	0	0	0	27	0	0	27	1.00
Snow	0	0	0	0	0	0	0	0	14	0	14	1.00
Urban	1	0	0	0	0	4	0	0	0	10	15	0.67
TOTAL	67	39	44	37	46	33	26	32	14	11	349	
PA	0.87	0.62	0.95	0.62	0.67	0.70	1.00	0.84	1.00	0.91		
Overall	0.80											

**Table F3** Confusion matrix for the 2000s composite. 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, PA = producer's accuracy and UA = user's accuracy.

	CP	BP	NNF	ANF	Shrubs	Grass	Water	Bare	Snow	Urban	TOTAL	UA
CP	42	4	1	0	0	0	0	0	0	0	47	0.89
BP	10	41	0	0	0	0	0	0	0	0	51	0.80
NNF	2	1	40	4	9	8	0	0	0	0	64	0.63
ANF	0	0	1	27	1	0	0	2	0	0	31	0.87
Shrubs	0	0	0	2	35	5	0	0	0	1	43	0.81
Grass	0	0	0	0	0	29	0	0	0	0	29	1.00
Water	0	0	0	0	0	0	24	0	0	0	24	1.00
Bare	0	0	0	1	0	0	0	25	0	0	26	0.96
Snow	0	0	0	0	0	0	0	0	20	0	20	1.00
Urban	0	0	0	0	0	1	0	0	0	19	20	0.95
TOTAL	54	46	42	34	45	43	24	27	20	20	355	
PA	0.78	0.89	0.95	0.79	0.78	0.67	1.00	0.93	1.00	0.95		
Overall	0.85											

**Table F4** Confusion matrix for the 2010s composite. 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, PA = producer's accuracy and UA = user's accuracy.

	CP	BP	NNF	ANF	Shrubs	Grass	Water	Bare	Snow	Urban	TOTAL	UA
CP	40	5	0	0	0	0	1	0	0	5	51	0.78
BP	4	35	0	0	0	0	0	0	0	0	39	0.90
NNF	3	5	63	26	9	4	0	0	0	0	110	0.57
ANF	0	0	1	49	0	0	0	0	0	0	50	0.98
Shrubs	1	1	0	5	37	6	0	2	0	0	52	0.71
Grass	0	0	0	0	0	53	0	0	0	0	53	1.00
Water	0	0	0	0	0	0	45	0	0	0	45	1.00
Bare	0	0	0	0	0	0	1	26	0	0	27	0.96
Snow	0	0	0	0	0	0	0	0	46	0	46	1.00
Urban	0	0	0	0	0	0	0	0	0	37	37	1.00
TOTAL	48	46	64	80	46	63	47	28	46	42	510	
PA	0.83	0.76	0.98	0.61	0.80	0.84	0.96	0.93	1.00	0.88		
Overall	0.85											

## Appendix G

### Land cover change matrices

**Table G1** Land cover change matrix 1980s-2010s. The following abbreviations are used: AP = alien plantations, NWS = native woody species.

		2010s land cover class (km <sup>2</sup> )						
1980s land cover class (km <sup>2</sup> )		AP	NWS	Grass	Water	Bare	Snow	Urban
Whole study area	Alien plantations	536.59	1479.22	194.36	4.87	1.13	2.93	40.00
	Native woody species	100.54	8534.07	93.01	20.89	80.17	207.05	75.21
	Grass	938.83	1754.15	2327.35	3.77	0.39	17.41	246.82
	Water	1.40	41.04	0.05	76.57	11.90	2.54	0.45
	Bare	10.72	67.12	14.79	6.27	101.34	19.32	1.71
	Snow	0.17	39.73	0.09	1.19	4.63	308.33	96.50
	Urban	11.21	33.43	11.59	3.03	0.79	0.05	6.77
Valley	Alien plantations	427.26	1034.25	180.20	0.94	0.68	2.93	39.03
	Native woody species	9.38	97.39	3.60	0.00	0.01	0.00	0.07
	Grass	864.86	1458.95	2201.16	0.36	0.28	17.40	244.55
	Water	0.36	0.61	0.02	0.16	0.05	0.02	0.03
	Bare	8.03	27.33	14.30	0.07	1.15	0.19	1.44
	Snow	0.14	0.04	0.02	0.00	0.01	0.00	0.02
	Urban	10.13	24.59	9.77	0.00	0.61	0.05	6.22
Andes	Alien plantations	109.43	445.04	14.16	3.94	0.45	0.00	0.97
	Native woody species	91.16	8436.68	89.41	20.89	80.15	207.05	75.14
	Grass	74.00	295.24	126.18	3.41	0.10	0.01	2.26
	Water	1.04	40.42	0.03	76.41	11.84	2.52	0.42
	Bare	2.70	39.79	0.49	6.19	100.19	19.13	0.27

	Snow	0.03	39.69	0.07	1.19	4.62	308.33	96.47
	Urban	1.09	8.84	1.83	3.03	0.18	0.00	0.56
Protecte d areas	Alien plantations	2.08	18.76	0.15	0.05	0.12	0.00	0.00
	Native woody species	5.21	1203.81	1.51	0.84	62.41	70.77	6.40
	Grass	0.55	9.28	2.06	0.00	0.00	0.00	0.03
	Water	0.13	16.75	0.02	9.81	10.78	2.18	0.15
	Bare	0.10	10.75	0.00	0.55	84.58	14.96	0.06
	Snow	0.00	5.16	0.02	0.35	3.93	117.59	5.90
	Urban	0.01	0.18	0.01	0.00	0.00	0.00	0.00

**Table G2** Land cover change matrix 1980s-1990s. The following abbreviations are used: AP = alien plantations, NWS = native woody species.

		2010s land cover class (km <sup>2</sup> )						
	1980s land cover class (km <sup>2</sup> )	AP	NWS	Grass	Water	Bare	Snow	Urban
Whole study area	Alien plantations	888.40	686.41	542.17	5.21	0.68	0.01	42.53
	Native woody species	66.06	8719.57	84.92	5.56	88.30	137.80	7.79
	Grass	737.35	419.08	3407.81	7.28	0.07	0.08	105.27
	Water	2.30	35.10	0.54	84.72	9.02	2.03	0.18
	Bare	16.00	38.81	26.40	3.87	116.89	8.47	2.55
	Snow	0.11	140.12	0.19	1.49	12.39	289.54	6.77
	Urban	11.43	9.10	30.63	0.28	0.12	0.00	12.43
Valley	Alien plantations	742.72	307.05	499.93	2.87	0.06	0.00	38.95
	Native woody species	19.34	80.92	8.64	0.15	0.00	0.01	0.47
	Grass	699.68	176.99	3193.97	6.24	0.02	0.01	98.90
	Water	0.59	0.04	0.31	0.17	0.00	0.00	0.07
	Bare	12.58	4.05	24.80	0.09	0.65	0.00	2.07
	Snow	0.04	0.00	0.13	0.00	0.00	0.00	0.04



	Urban	10.20	1.90	25.56	0.02	0.06	0.00	10.74
Andes	Alien plantations	145.75	379.46	42.24	2.34	0.61	0.01	3.58
	Native woody species	46.72	8638.65	76.29	5.41	88.30	137.80	7.32
	Grass	37.71	242.12	213.84	1.04	0.05	0.07	6.37
	Water	1.71	35.06	0.23	84.55	9.02	2.03	0.11
	Bare	3.42	34.76	1.60	3.78	116.24	8.47	0.48
	Snow	0.07	140.12	0.06	1.49	12.39	289.54	6.73
	Urban	1.23	7.20	5.07	0.26	0.06	0.00	1.69
Protected areas	Alien plantations	2.01	18.42	0.40	0.21	0.10	0.00	0.01
	Native woody species	3.39	1251.56	0.98	2.84	69.69	21.83	0.68
	Grass	0.11	9.13	2.60	0.00	0.00	0.04	0.03
	Water	0.16	15.35	0.01	15.87	6.66	1.77	0.00
	Bare	0.04	9.48	0.00	2.40	91.68	7.38	0.00
	Snow	0.00	23.77	0.02	1.39	11.01	95.96	0.80
	Urban	0.00	0.16	0.03	0.00	0.00	0.00	0.00

**Table G3** Land cover change matrix 1990s-2000s. The following abbreviations are used: AP = alien plantations, NWS = native woody species.

		2010s land cover class (km <sup>2</sup> )						
1980s land cover class (km <sup>2</sup> )		AP	NWS	Grass	Water	Bare	Snow	Urban
Whole study area	Alien plantations	836.15	709.70	94.87	4.86	10.74	0.00	65.34
	Native woody species	162.36	9514.85	159.94	14.51	48.61	92.21	55.72
	Grass	501.30	1018.05	2484.18	12.18	5.39	0.20	71.38
	Water	1.16	12.58	5.27	78.75	9.45	1.13	0.07
	Bare	0.14	21.99	0.07	6.42	187.66	11.11	0.09
	Snow	0.00	142.73	0.01	0.05	21.58	269.69	3.86

	Urban	40.41	39.63	65.59	2.61	2.27	2.38	24.64
Valley	Alien plantations	743.00	597.72	79.85	1.22	7.54	0.00	55.83
	Native woody species	64.39	468.35	36.77	0.02	0.03	0.00	1.40
	Grass	479.84	924.47	2279.48	0.20	1.94	0.00	67.42
	Water	0.71	3.44	5.09	0.22	0.04	0.00	0.04
	Bare	0.02	0.12	0.00	0.00	0.62	0.00	0.04
	Snow	0.00	0.01	0.00	0.00	0.00	0.00	0.00
	Urban	37.57	31.61	61.82	0.08	1.66	0.00	18.50
Andes	Alien plantations	93.17	112.04	15.04	3.64	3.20	0.00	9.50
	Native woody species	97.97	9046.59	123.22	14.49	48.58	92.21	54.32
	Grass	21.46	93.58	204.70	11.98	3.46	0.19	3.96
	Water	0.45	9.15	0.18	78.53	9.41	1.13	0.03
	Bare	0.12	21.88	0.07	6.42	187.04	11.11	0.05
	Snow	0.00	142.72	0.01	0.05	21.58	269.69	3.86
	Urban	2.84	8.02	3.77	2.53	0.61	2.38	6.14
Protected areas	Alien plantations	0.54	5.02	0.13	0.03	0.01	0.00	0.00
	Native woody species	1.04	1355.82	3.18	1.61	28.58	32.38	2.95
	Grass	0.11	1.62	2.21	0.00	0.00	0.02	0.13
	Water	0.04	3.57	0.00	9.56	8.74	1.00	0.00
	Bare	0.01	14.19	0.00	2.92	153.32	9.93	0.02
	Snow	0.00	29.44	0.00	0.03	17.84	137.06	0.28
	Urban	0.00	0.79	0.02	0.00	0.00	0.31	0.64

**Table G4** Land cover change matrix 2000s-2010s. The following abbreviations are used: AP = alien plantations, NWS = native woody species.

		2010s land cover class (km <sup>2</sup> )						
1980s land cover class (km <sup>2</sup> )		AP	NWS	Grass	Water	Bare	Snow	Urban

Whole study area	Alien plantations	871.23	557.76	175.50	0.63	0.24	1.20	38.44
	Native woody species	411.83	10374.07	511.31	2.37	20.20	193.08	159.66
	Grass	228.15	860.65	1897.68	0.07	0.04	12.12	156.06
	Water	3.85	3.79	0.40	107.62	3.61	0.12	0.28
	Bare	9.63	37.40	1.32	4.73	169.39	63.51	0.97
	Snow	0.02	60.89	0.26	1.10	6.38	284.08	23.98
	Urban	74.76	54.20	54.77	0.07	0.47	3.52	88.06
Valley	Alien plantations	763.95	457.05	168.59	0.27	0.12	1.20	37.85
	Native woody species	285.58	1451.81	442.54	0.12	0.32	6.14	52.19
	Grass	201.61	695.35	1743.87	0.07	0.04	12.10	154.80
	Water	0.73	0.11	0.13	0.73	0.14	0.06	0.15
	Bare	4.15	4.67	0.90	0.28	1.82	0.62	0.63
	Snow	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Urban	64.13	34.17	53.05	0.07	0.36	0.48	45.73
Andes	Alien plantations	107.30	100.73	6.91	0.36	0.12	0.00	0.58
	Native woody species	126.33	8922.33	68.77	2.25	19.88	186.95	107.47
	Grass	26.58	165.33	153.81	0.00	0.00	0.02	1.26
	Water	3.12	3.68	0.27	106.89	3.47	0.07	0.13
	Bare	5.48	32.73	0.42	4.45	167.58	62.89	0.34
	Snow	0.02	60.89	0.26	1.10	6.38	284.08	23.98
	Urban	10.62	20.03	1.72	0.00	0.11	3.04	42.33
Protected areas	Alien plantations	0.64	1.00	0.05	0.04	0.00	0.00	0.00
	Native woody species	7.44	1320.52	2.05	0.65	14.20	53.46	12.13
	Grass	0.20	3.50	1.83	0.00	0.00	0.00	0.01
	Water	0.04	1.74	0.00	10.08	2.17	0.04	0.07
	Bare	0.11	13.67	0.00	0.72	140.83	53.08	0.07
	Snow	0.00	12.82	0.05	0.93	5.48	155.92	5.50

	Urban	0.04	0.86	0.12	0.00	0.00	0.22	2.79
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## Appendix H

Confusion matrices very high resolution land cover maps

**Table H1** Accuracy assessment Worldview object-based analysis Malalcahuello North. The following abbreviations are used: PA = producer's accuracy and UA = user's accuracy.

	Pinaceae	<i>Nothofagus</i> spp.	<i>Araucaria araucana</i>	Shrubs	Grasslands	Bare	Urban	Total	UA
Pinaceae	46	0	19	1	0	0	0	66	0.70
<i>Nothofagus</i> spp.	5	66	7	8	1	0	0	87	0.76
<i>Araucaria araucana</i>	10	0	50	4	0	0	0	64	0.78
Shrubs	1	2	1	22	0	0	0	26	0.85
Grasslands	0	0	0	1	30	0	0	31	0.97
Bare	0	0	0	0	0	29	4	33	0.88
Urban	0	0	0	0	0	0	15	15	1.00
Total	62	68	77	36	31	29	19	322	
PA	0.74	0.97	0.65	0.61	0.97	1.00	0.79		
Overall accuracy	0.80								

**Table H2** Accuracy assessment Worldview object-based analysis Malalcahuello South. The following abbreviations are used: PA = producer's accuracy and UA = user's accuracy.

	Pinaceae	<i>Nothofagus</i> spp.	<i>Araucaria araucana</i>	Shrubs	Grasslands	Total	UA
Pinaceae	84	7	5	0	0	96	0.88

<i>Nothofagus</i> spp.	10	75	17	8	0	110	0.68
<i>Araucaria araucana</i>	1	6	42	11	0	60	0.70
Shrubs	0	0	1	20	3	24	0.83
Grasslands	0	0	0	3	23	26	0.88
Total	95	88	65	42	26	316	
PA	0.88	0.85	0.65	0.48	0.88		
Overall accuracy	0.77						

**Table H3** Accuracy assessment Worldview object-based analysis Villarrica North. The following abbreviations are used: PA = producer's accuracy and UA = user's accuracy.

	Pinaceae	<i>Nothofagus</i> spp.	Shrubs	Grasslands	Bare	Urban	Total	UA
Pinaceae	44	0	8	0	2	0	54	0.81
<i>Nothofagus</i> spp.	0	58	3	0	0	0	61	0.95
Shrubs	3	11	43	1	0	0	58	0.74
Grasslands	0	0	0	25	0	0	25	1.00
Bare	0	0	4	0	17	0	21	0.81
Urban	0	0	0	0	0	5	5	1.00
Total	47	69	58	26	19	5	224	
PA	0.94	0.84	0.74	0.96	0.89	1.00		
Overall accuracy	0.86							

**Table H4** Accuracy assessment Worldview object-based analysis Villarrica South. The following abbreviations are used: PA = producer's accuracy and UA = user's accuracy.

	Pinaceae	<i>Nothofagus</i> spp.	Shrubs	Grasslands	Bare	Total	UA
Pinaceae	69	11	0	0	0	80	0.86
<i>Nothofagus</i> spp.	3	66	7	0	0	76	0.87
Shrubs	0	2	37	0	0	39	0.95
Grasslands	0	0	0	48	0	48	1.00
Bare	0	0	0	0	33	33	1.00
Total	72	79	44	48	33	276	
PA	0.96	0.84	0.84	1.00	1.00		
Overall accuracy	0.92						

## Appendix I

Landscape metrics results very high resolution imagery assessment

**Table II** Class level metrics Malcalcahuello North.

Distance to plantation	Land cover class	Class level metrics						
		PD (patches/ha)	AREA_MN (ha)	LPI (%)	PAFRAC	CLUMPY	PLAND (%)	cAI (%)
Plantation	Coniferous alien trees	79.52	0.92	53.34	N/A	0.82	73.43	95.12
	<i>Nothofagus</i> spp. trees	1049.60	0.00	0.81	1.25	0.77	4.38	77.87
	<i>Araucaria araucana</i> trees	1765.24	0.01	2.60	1.29	0.78	18.71	81.76
	Shrubs	333.96	0.01	0.87	1.14	0.83	3.01	83.39
	Grasslands	270.35	0.00	0.08	1.41	0.55	0.38	55.50
100 m	Coniferous alien trees	566.58	0.04	6.76	1.34	0.84	21.96	87.19
	<i>Nothofagus</i> spp. trees	748.69	0.02	8.07	1.31	0.87	18.46	89.38
	<i>Araucaria araucana</i> trees	870.10	0.04	6.57	1.34	0.83	31.40	88.44
	Shrubs	531.16	0.01	0.30	1.27	0.81	6.59	82.54
	Grasslands	217.52	0.02	1.69	1.27	0.88	4.32	88.42
200 m	Coniferous alien trees	522.43	0.04	5.14	1.36	0.84	20.49	87.64
	<i>Nothofagus</i> spp. trees	666.09	0.03	9.01	1.30	0.88	20.87	90.25
	<i>Araucaria araucana</i> trees	831.53	0.04	5.18	1.33	0.83	29.90	88.05
	Shrubs	539.84	0.02	1.78	1.30	0.84	9.66	85.20
	Grasslands	230.74	0.01	0.22	1.26	0.82	2.30	82.60
300 m	Coniferous alien trees	584.59	0.03	2.26	1.34	0.83	15.25	85.68
	<i>Nothofagus</i> spp. trees	658.69	0.03	4.43	1.30	0.87	21.69	90.04
	<i>Araucaria araucana</i> trees	679.28	0.05	15.22	1.37	0.83	34.75	89.18



	Shrubs	617.52	0.02	2.89	1.30	0.84	11.28	85.76
	Grasslands	288.18	0.02	0.94	1.25	0.86	4.46	86.16
400 m	Coniferous alien trees	508.24	0.03	4.39	1.34	0.84	12.68	86.41
	<i>Nothofagus</i> spp. trees	713.95	0.03	5.09	1.31	0.87	24.31	90.09
	<i>Araucaria araucana</i> trees	746.22	0.04	7.31	1.33	0.85	29.04	89.62
	Shrubs	645.38	0.02	0.93	1.31	0.82	10.88	84.39
	Grasslands	250.08	0.02	2.18	1.30	0.87	5.46	87.42
500 m	Coniferous alien trees	482.05	0.02	1.92	1.30	0.84	8.40	84.94
	<i>Nothofagus</i> spp. trees	634.70	0.04	4.13	1.32	0.87	25.55	90.21
	<i>Araucaria araucana</i> trees	911.88	0.02	1.55	1.34	0.82	19.30	85.39
	Shrubs	478.04	0.04	5.69	1.31	0.87	17.38	88.93
	Grasslands	164.70	0.05	5.62	1.30	0.91	8.87	92.06
600 m	Coniferous alien trees	287.54	0.02	0.92	1.35	0.84	5.87	84.54
	<i>Nothofagus</i> spp. trees	544.28	0.02	4.33	1.31	0.86	13.26	87.71
	<i>Araucaria araucana</i> trees	795.87	0.02	1.72	1.32	0.83	18.85	86.34
	Shrubs	467.26	0.03	5.01	1.34	0.86	14.41	87.81
	Grasslands	256.73	0.06	10.70	1.32	0.91	16.53	92.44
700 m	Coniferous alien trees	361.45	0.01	0.74	1.30	0.82	4.76	82.78
	<i>Nothofagus</i> spp. trees	456.25	0.01	2.74	1.23	0.86	6.60	87.20
	<i>Araucaria araucana</i> trees	616.24	0.04	8.58	1.35	0.86	25.88	89.34
	Shrubs	397.00	0.03	2.73	1.30	0.86	11.54	87.63
	Grasslands	462.18	0.04	2.96	1.30	0.88	17.98	89.85
800 m	Coniferous alien trees	221.04	0.01	0.46	1.30	0.82	2.43	82.46
	<i>Nothofagus</i> spp. trees	412.60	0.03	4.93	1.31	0.90	11.43	91.02
	<i>Araucaria araucana</i> trees	648.37	0.03	7.13	1.32	0.86	21.72	88.92
	Shrubs	493.65	0.02	3.25	1.30	0.84	7.72	85.60

	Grasslands	456.81	0.06	9.66	1.30	0.90	29.28	92.61
900 m	Coniferous alien trees	267.53	0.01	0.43	1.23	0.83	3.80	84.06
	<i>Nothofagus</i> spp. trees	450.97	0.05	10.46	1.26	0.92	24.24	93.88
	<i>Araucaria araucana</i> trees	649.71	0.03	2.28	1.30	0.85	18.47	87.36
	Shrubs	588.56	0.02	1.71	1.28	0.85	11.34	86.64
	Grasslands	321.03	0.05	7.64	1.28	0.91	14.81	92.29
1000 m	Coniferous alien trees	272.01	0.02	1.13	1.25	0.86	5.64	87.18
	<i>Nothofagus</i> spp. trees	592.60	0.05	12.57	1.33	0.87	26.99	90.33
	<i>Araucaria araucana</i> trees	806.32	0.02	5.52	1.35	0.83	16.42	85.85
	Shrubs	854.90	0.02	2.41	1.32	0.82	16.67	84.79
	Grasslands	330.30	0.03	3.87	1.27	0.89	10.75	90.16
1100 m	Coniferous alien trees	239.53	0.01	0.57	1.23	0.82	2.98	82.21
	<i>Nothofagus</i> spp. trees	416.58	0.10	25.06	1.36	0.87	41.40	92.63
	<i>Araucaria araucana</i> trees	718.59	0.02	2.46	1.34	0.83	15.93	85.61
	Shrubs	812.32	0.02	5.67	1.31	0.84	18.18	86.95
	Grasslands	208.29	0.04	6.90	1.25	0.95	7.75	95.38
1200 m	Coniferous alien trees	448.77	0.01	1.07	1.24	0.83	5.75	83.93
	<i>Nothofagus</i> spp. trees	473.70	0.08	18.29	1.33	0.88	39.03	92.58
	<i>Araucaria araucana</i> trees	847.67	0.01	0.90	1.26	0.82	11.78	83.84
	Shrubs	772.87	0.02	2.22	1.33	0.82	15.30	84.67
	Grasslands	236.85	0.02	1.29	1.32	0.84	3.95	84.77
1300 m	Coniferous alien trees	273.22	0.01	0.57	1.26	0.82	2.63	82.50
	<i>Nothofagus</i> spp. trees	375.68	0.08	11.25	1.32	0.90	28.99	92.94
	<i>Araucaria araucana</i> trees	478.13	0.01	1.07	1.23	0.84	5.84	84.67
	Shrubs	648.89	0.02	2.31	1.31	0.82	10.89	83.99
	Grasslands	239.07	0.02	1.43	1.39	0.82	3.98	82.79

1400 m	Coniferous alien trees	270.18	0.02	3.49	1.31	0.84	5.97	85.08
	<i>Nothofagus</i> spp. trees	457.23	0.07	16.98	1.27	0.90	33.05	93.35
	<i>Araucaria araucana</i> trees	602.71	0.02	1.14	1.25	0.83	9.16	84.52
	Shrubs	602.71	0.02	1.47	1.27	0.84	10.08	85.86
	Grasslands	394.88	0.02	2.31	1.29	0.85	7.26	86.38
1500 m	Coniferous alien trees	127.05	0.05	2.43	N/A	0.93	5.78	93.22
	<i>Nothofagus</i> spp. trees	296.44	0.01	1.28	1.16	0.88	4.09	88.15
	<i>Araucaria araucana</i> trees	423.49	0.02	1.91	1.30	0.84	6.56	85.02
	Shrubs	232.92	0.02	1.44	1.38	0.85	3.77	85.22
	Grasslands	232.92	0.06	4.46	1.55	0.88	12.90	89.52

**Table I2** Landscape level metrics Malalcahuello North.

Distance to plantation	Landscape level metrics	
	AI (%)	SHDI
Plantation	91.31	0.81
100 m	89.15	1.72
200 m	89.53	1.66
300 m	89.25	1.68
400 m	90.02	1.72
500 m	90.43	1.74
600 m	91.31	1.72
700 m	91.35	1.67
800 m	91.37	1.63
900 m	92.27	1.70
1000 m	90.04	1.73

1100 m	90.97	1.54
1200 m	90.83	1.54
1300 m	93.31	1.42
1400 m	92.36	1.60
1500 m	94.00	1.28

**Table B3** Class level metrics Malalcahuello South.

Distance to plantation	Land cover class	Class level metrics						
		PD (patches/ha)	AREA_MN (ha)	LPI (%)	PAFRAC	CLUMPY	PLAND (%)	cAI (%)
Plantation	Coniferous alien trees	257.10	0.29	35.82	1.38	0.79	75.65	94.84
	<i>Nothofagus</i> spp. trees	1698.40	0.01	3.60	1.32	0.78	22.20	82.53
	<i>Araucaria araucana</i> trees	276.57	0.01	0.22	1.29	0.78	2.13	78.70
	Shrubs							
	Grasslands	19.48	0.00	0.01	N/A	0.53	0.02	52.78
100 m	Coniferous alien trees	1029.86	0.03	2.70	1.35	0.81	32.74	87.36
	<i>Nothofagus</i> spp. trees	662.78	0.09	10.33	1.38	0.81	59.01	92.07
	<i>Araucaria araucana</i> trees	505.75	0.01	0.21	1.31	0.80	7.26	81.85
	Shrubs	34.67	0.00	0.06	1.24	0.77	0.17	77.16
	Grasslands	75.46	0.01	0.59	1.26	0.88	0.82	88.58
200 m	Coniferous alien trees	981.60	0.02	2.71	1.34	0.82	23.80	86.56
	<i>Nothofagus</i> spp. trees	572.42	0.11	16.25	1.38	0.81	63.12	92.82
	<i>Araucaria araucana</i> trees	602.11	0.02	1.11	1.33	0.81	10.43	83.27
	Shrubs	97.52	0.01	0.16	1.31	0.82	0.99	81.89
	Grasslands	184.45	0.01	0.22	1.23	0.83	1.66	83.61
300 m	Coniferous alien trees	1065.89	0.02	4.60	1.31	0.83	17.23	85.58

	<i>Nothofagus</i> spp. trees	476.98	0.14	17.60	1.36	0.81	67.58	93.68
	<i>Araucaria araucana</i> trees	596.90	0.02	1.22	1.38	0.81	13.77	83.57
	Shrubs	90.60	0.01	0.10	1.32	0.79	0.65	78.95
	Grasslands	157.22	0.00	0.10	1.20	0.81	0.78	80.69
400 m	Coniferous alien trees	1239.97	0.01	0.59	1.31	0.79	14.74	82.15
	<i>Nothofagus</i> spp. trees	483.70	0.14	18.64	1.36	0.79	68.35	93.31
	<i>Araucaria araucana</i> trees	744.75	0.02	0.97	1.36	0.80	14.85	83.23
	Shrubs	107.49	0.01	0.27	1.35	0.82	1.26	81.78
	Grasslands	222.66	0.00	0.21	1.24	0.77	0.80	77.55
500 m	Coniferous alien trees	1281.70	0.01	0.62	1.26	0.80	12.80	82.34
	<i>Nothofagus</i> spp. trees	678.96	0.08	10.37	1.36	0.80	57.41	91.43
	<i>Araucaria araucana</i> trees	886.80	0.03	3.19	1.36	0.80	25.72	85.33
	Shrubs	76.21	0.04	1.61	1.38	0.89	2.83	88.97
	Grasslands	103.92	0.01	0.80	1.28	0.86	1.24	85.71
600 m	Coniferous alien trees	730.91	0.01	0.89	1.24	0.84	10.74	86.05
	<i>Nothofagus</i> spp. trees	1389.76	0.02	4.25	1.32	0.80	29.76	86.09
	<i>Araucaria araucana</i> trees	638.26	0.07	19.95	1.41	0.80	46.78	89.39
	Shrubs	319.13	0.03	3.32	1.41	0.85	10.83	86.75
	Grasslands	298.54	0.01	0.48	1.30	0.79	1.88	79.68
700 m	Coniferous alien trees	949.22	0.01	0.50	1.18	0.82	9.55	83.72
	<i>Nothofagus</i> spp. trees	1443.34	0.02	5.80	1.32	0.79	34.28	86.42
	<i>Araucaria araucana</i> trees	494.11	0.09	15.72	1.44	0.79	46.39	88.82
	Shrubs	338.08	0.03	3.51	1.38	0.85	8.95	86.70
	Grasslands	260.06	0.00	0.50	1.20	0.82	0.84	82.50
800 m	Coniferous alien trees	1044.55	0.01	1.16	1.32	0.79	10.10	81.25
	<i>Nothofagus</i> spp. trees	1438.40	0.02	7.28	1.34	0.79	33.90	85.92

	<i>Araucaria araucana</i> trees	616.46	0.08	18.79	1.39	0.80	48.88	90.01
	Shrubs	222.61	0.02	2.16	1.30	0.89	5.50	89.16
	Grasslands	171.24	0.01	0.62	1.23	0.86	1.62	86.13
900 m	Coniferous alien trees	1205.12	0.01	0.56	1.20	0.78	6.82	79.82
	<i>Nothofagus</i> spp. trees	652.78	0.07	21.82	1.38	0.81	46.00	89.75
	<i>Araucaria araucana</i> trees	778.31	0.05	9.79	1.40	0.80	42.00	88.55
	Shrubs	225.96	0.01	1.46	N/A	0.84	3.34	84.09
	Grasslands	301.28	0.01	0.68	1.26	0.82	1.84	82.16
1000 m	Coniferous alien trees	1122.58	0.01	2.15	1.34	0.78	8.90	79.66
	<i>Nothofagus</i> spp. trees	493.94	0.13	61.18	1.44	0.79	62.03	92.11
	<i>Araucaria araucana</i> trees	1392.00	0.02	6.39	1.39	0.78	28.31	84.10
	Shrubs	89.81	0.01	0.70	N/A	0.91	0.74	91.30
	Grasslands	89.81	0.00	0.01	N/A	-1.00	0.02	0.00

**Table I4** Landscape level metrics Malalcahuello South.

Distance to plantation	Landscape level metrics	
	AI (%)	SHDI
Plantation	91.75	0.63
100 m	89.73	0.92
200 m	90.07	0.98
300 m	90.69	0.91
400 m	89.90	0.92
500 m	88.55	1.09
600 m	87.58	1.27
700 m	87.27	1.20

800 m	87.63	1.17
900 m	88.24	1.09
1000 m	88.71	0.91

**Table I5** Class level metrics Villarrica North.

Distance to plantation	Land cover class	Class level metrics						
		PD (patches/ha)	AREA_MN (ha)	LPI (%)	PAFRAC	CLUMPY	PLAND (%)	cAI (%)
Plantation	Coniferous alien trees	103.67	0.80	47.17	1.40	0.79	83.30	96.54
	<i>Nothofagus</i> spp. trees	698.23	0.01	0.48	1.27	0.81	7.16	82.59
	Shrubs	818.66	0.01	0.31	1.32	0.79	9.53	81.33
	Grasslands	3.05	0.00	0.00	N/A	0.78	0.00	77.78
100 m	Coniferous alien trees	1293.61	0.03	12.00	1.32	0.82	36.28	88.73
	<i>Nothofagus</i> spp. trees	502.84	0.04	8.97	1.30	0.88	20.48	90.22
	Shrubs	776.17	0.05	7.49	1.35	0.82	38.53	88.73
	Grasslands	77.20	0.04	1.45	1.30	0.91	3.34	91.48
200 m	Coniferous alien trees	804.60	0.01	1.27	1.28	0.81	7.25	82.79
	<i>Nothofagus</i> spp. trees	344.83	0.19	31.00	1.36	0.84	64.24	94.40
	Shrubs	932.32	0.03	4.32	1.32	0.83	27.50	87.43
	Grasslands	51.09	0.00	0.13	N/A	0.79	0.18	79.51
300 m	Coniferous alien trees	314.30	0.00	0.33	1.26	0.77	1.03	77.37
	<i>Nothofagus</i> spp. trees	362.65	0.21	47.01	1.35	0.82	74.55	95.39
	Shrubs	870.36	0.03	8.51	1.31	0.84	23.81	87.74
	Grasslands	8.06	0.00	0.02	N/A	1.00	0.02	100.00
400 m	Coniferous alien trees	59.74	0.00	0.01	N/A	0.42	0.04	41.94
	<i>Nothofagus</i> spp. trees	68.28	1.24	84.73	N/A	0.77	84.88	96.59

	Shrubs	955.85	0.02	1.68	1.31	0.81	14.82	83.86
	Grasslands	8.53	0.00	0.01	N/A	1.00	0.01	100.00
500 m	Coniferous alien trees	25.14	0.00	0.00	N/A	0.43	0.01	42.86
	<i>Nothofagus</i> spp. trees	159.20	0.47	73.43	1.37	0.84	74.88	95.87
	Shrubs	871.40	0.03	5.66	1.30	0.85	24.91	89.01
	Grasslands							
600 m	Coniferous alien trees							
	<i>Nothofagus</i> spp. trees	432.44	0.17	35.53	1.37	0.78	72.20	93.82
	Shrubs	1157.08	0.02	7.09	1.36	0.79	26.87	84.79
	Grasslands	23.38	0.01	0.26	N/A	0.91	0.26	91.19
700 m	Coniferous alien trees							
	<i>Nothofagus</i> spp. trees	1457.32	0.04	38.63	1.37	0.77	56.61	90.15
	Shrubs	1233.11	0.03	23.64	1.37	0.80	41.94	88.32
	Grasslands	56.05	0.01	0.40	N/A	0.95	0.40	94.64
800 m	Coniferous alien trees							
	<i>Nothofagus</i> spp. trees	2925.84	0.01	8.14	1.28	0.77	23.14	82.69
	Shrubs	975.28	0.07	52.99	N/A	0.79	67.68	93.07
	Grasslands							

**Table I6** Landscape level metrics Villarrica North.

Distance to plantation	Landscape level metrics	
	AI (%)	SHDI
Plantation	94.09	0.57
100 m	89.08	1.23
200 m	91.51	0.88



300 m	93.31	0.64
400 m	94.67	0.44
500 m	94.15	0.58
600 m	91.30	0.64
700 m	89.39	0.76
800 m	90.02	0.82

**Table I7** Class level metrics Villarrica South.

Distance to plantation	Land cover class	Class level metrics						
		PD (patches/ha)	AREA_MN (ha)	LPI (%)	PAFRAC	CLUMPY	PLAND (%)	cAI (%)
Plantation	Coniferous alien trees	126.54	0.67	52.62	1.32	0.82	84.87	97.31
	<i>Nothofagus</i> spp. trees	1287.80	0.01	1.39	1.32	0.80	13.62	82.51
	Shrubs	121.73	0.01	0.12	1.30	0.80	1.11	80.32
	Grasslands	17.62	0.01	0.15	1.23	0.89	0.23	89.03
100 m	Coniferous alien trees	1499.69	0.01	1.47	1.35	0.77	13.81	80.37
	<i>Nothofagus</i> spp. trees	348.60	0.19	17.70	1.36	0.83	67.62	94.35
	Shrubs	337.70	0.02	1.22	1.35	0.85	7.70	85.80
	Grasslands	78.07	0.03	1.46	1.14	0.95	2.34	94.78
200 m	Coniferous alien trees	914.05	0.01	0.46	1.33	0.78	7.96	79.63
	<i>Nothofagus</i> spp. trees	212.46	0.33	29.55	1.35	0.85	71.03	95.58
	Shrubs	377.97	0.03	2.05	1.32	0.88	10.18	88.80
	Grasslands	49.41	0.02	1.03	1.13	0.95	1.22	95.45
300 m	Coniferous alien trees	689.84	0.01	0.74	1.31	0.81	6.74	81.93
	<i>Nothofagus</i> spp. trees	328.03	0.22	29.58	1.36	0.82	70.66	94.64
	Shrubs	501.70	0.04	9.86	1.36	0.87	19.52	89.90

	Grasslands	110.95	0.00	0.07	1.25	0.75	0.36	75.38
400 m	Coniferous alien trees	485.32	0.01	1.26	1.31	0.82	5.54	83.01
	<i>Nothofagus</i> spp. trees	194.13	0.42	81.30	1.39	0.79	81.98	96.24
	Shrubs	647.10	0.02	4.79	1.33	0.84	11.91	86.17
	Grasslands	145.60	0.00	0.15	1.21	0.78	0.53	78.23
500 m	Coniferous alien trees	756.49	0.00	0.37	1.32	0.74	3.69	75.07
	<i>Nothofagus</i> spp. trees	256.23	0.32	80.55	1.34	0.79	82.76	96.34
	Shrubs	378.24	0.04	6.04	1.37	0.86	13.32	87.53
	Grasslands	73.21	0.00	0.15	N/A	0.78	0.23	78.26
600 m	Coniferous alien trees	973.89	0.01	0.76	1.34	0.74	4.95	74.93
	<i>Nothofagus</i> spp. trees	243.47	0.34	82.55	N/A	0.82	83.24	96.93
	Shrubs	378.73	0.03	8.49	1.31	0.90	10.48	90.95
	Grasslands	108.21	0.01	0.88	N/A	0.83	1.24	83.58

**Table 18** Landscape level metrics Villarrica South.

Distance to plantation	Landscape level metrics	
	AI (%)	SHDI
Plantation	95.05	0.49
100 m	91.94	1.03
200 m	93.84	0.96
300 m	92.86	0.86
400 m	94.20	0.61
500 m	94.34	0.56
600 m	95.04	0.60