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The invasive species *Ulex europaeus* (Fabaceae) shows high dynamism in a fragmented landscape of south-central Chile

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Abstract *Ulex europaeus* (gorse) is an invasive shrub deemed as one of the most invasive species in the world. *U. europaeus* is widely distributed in the south-central area of Chile, which is considered a world hotspot for biodiversity conservation. In addition to its negative effects on the biodiversity of natural ecosystems, *U. europaeus* is one of the most severe pests for agriculture and forestry. Despite its importance as an invasive species, *U. europaeus* has been little studied. Although information exists on the potential distribution of the species, the interaction of the invasion process with the spatial dynamic of the landscape and the landscape-scale factors that control the presence or absence of the species is still lacking. We studied the spatial and temporal dynamics of the landscape and how these relate to *U. europaeus* invasion in south-central Chile. We used supervised classification of satellite images to determine the spatial distribution of the species and other land covers for the years 1986 and 2003, analysing the transitions between the different land covers. We used logistic regression for modelling the increase, decrease and permanence of *U. europaeus* invasion considering landscape variables. Results showed that the species covers only around 1 % of the study area and showed a 42 % reduction in area for the studied period. However, *U. europaeus* was the cover type which presented the greatest dynamism in the landscape. We found a strong relationship between changes in land cover and the invasion process, especially connected with forest plantations of exotic species, which promotes the displacement of *U. europaeus*. The model of gorse cover increase presented the best performance, and the most important predictors were distance to seed source and landscape complexity index. Our model predicted high spread potential of *U. europaeus* in areas of high conservation value. We conclude that proper management for this invasive species must take into account the spatial dynamics of the landscape within the invaded area in order to address containment, control or mitigation of the invasion.

Keywords Land use/cover change · Biological invasions · Logistic regression · Remote sensing · Forest plantations · Biodiversity conservation

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Introduction

Biological invasions are a main cause of global change (Vitousek et al. 1997; Sala et al. 2000; McGeoch et al. 2010; Blackburn et al. 2014) and also one of the principal threats to biodiversity conservation and the functioning of natural ecosystems on a global scale (Wilcove and Chen 1998; Mack et al. 2000; Mooney and Cleland 2001). Furthermore, biological invasions have resulted in large economic and social impacts (Vilá et al. 2010).

The disturbances generated by human transformations (e.g. changes in land use, forest fires, urban expansion) can increase the likelihood of plant invasions in natural communities (Hobbs and Huenneke 1992; Von der Lippe and Kowarik 2007; Chytrý et al. 2012; Keeley and Brennan 2012). Disturbances of natural areas often tend to increase the abundance and distribution of exotic plants (D'Antonio and Vitousek 1992; Hobbs and Huenneke 1992; Lake and Lewisham 2004). Likewise, plant invasions also modify significantly disturbance regimes, such as fires or soil erosion, which implies changes in the community structure and the functioning of ecosystems (Mack and D'Antonio 1998).

In Chile, invasive plant species are present in more than half of the temperate ecosystems of the south-central area (Fuentes et al. 2013). These ecosystems are classified as a world hotspot of biodiversity (Myers et al. 2000) due to their high level of endemism and the presence of unispecific genera (Arroyo et al. 2008; Smith-Ramírez 2004). This is considered one of the most threatened regions in the world by WWF's Global 2000 Project (Dinerstein et al. 1995).

The review by Arroyo et al. (2000) indicated that Chile had approximately 690 naturalized exotic species, representing a staggering 15 % of the country's total flora. More recently, the database produced by Fuentes et al. (2013) establishes 743 exotic species recorded on mainland Chile, distributed among 631 genera and 74 families. Of these, the Fabaceae (Leguminosae) family, to which belongs *U. europaeus*, is one of the most important in the world in terms of the number of invasive species.

Fabaceae is a family of the order Fabales that includes both woody and herbaceous plants. Its geographical distribution is cosmopolitan, with more than 700 genera and 19,000 species (Zuloaga et al. 2008). Of the invasive Fabaceae that grow in Chile, the impact on the biodiversity or species richness of natural communities has been reported for *Acacia dealbata* (Fuentes-Ramirez et al. 2010; Fuentes-Ramirez et al. 2011) and *Teline monspessulana* (Pauchard et al. 2008; García et al. 2014). However, other invasive species (e.g. *Ulex europaeus*) have not been evaluated so far. An outstanding characteristic of this group of plants is their ability to fix nitrogen from the atmosphere through symbiotic associations with bacteria which colonize the roots (García et al. 2012). This allows them to grow in nutrient-poor environments and compete successfully in degraded sites.

U. europaeus L. (gorse) is considered invasive in various parts of the world, growing across a wide range of latitudes and at altitudes ranging from sea level to 4000 m (Hornoy et al. 2011). *U. europaeus* is a spiny, nitrogen-fixing shrub, native of Western Europe, with high reproductive capacity, prolonged seed latency, long vegetative period, rapid growth and high combustibility (Holm et al. 1977; Clements et al. 2001; Rees and Hill 2001). This species has been classified by IUCN as one of the hundred most invasive species in the world (Lowe et al. 2004) and a pest with deleterious effects on agriculture and forestry (Holm et al. 1977). In Chile, it is considered as one of the most severe pests for agriculture and forestry in the area between 37° and 43° S (Mathei 1995). The species was introduced in Chile at the beginning of the nineteenth century for use as a hedge plant to contain livestock and as a fodder source (Norambuena et al. 2000).

Despite its worldwide importance as an invasive species, to date, there are few studies of *U. europaeus*. Nevertheless, research on this species has explored genetic aspects (Atlan et al. 2015), evolutionary changes in its characteristics (Hornoy et al. 2011), effects on natural regeneration (Baret et al. 2006), modelling of spatial distribution (Mgidi et al. 2007) and effects on soil characteristics (Leary et al. 2006). Only a very recent research has studied the landscape characteristics that promote the establishment and spread of *U. europaeus* (Cordero et al. 2016). In Chile, studies of *U. europaeus* have concentrated principally on control methods (Norambuena and Piper 2000; Norambuena et al. 2007; Muñoz 2009), with no studies looking at the landscape dynamics and its relationship with the presence/absence of gorse.

Models of invasive species usually predict the potential distribution of the species (Chiou et al. 2013) but not the invasion dynamics. Land use/cover is among the most important factors in determining the distribution of exotic species (Hobbs 2000). This is because land use/cover has a direct effect on invasion processes, modifying the disturbance regime and environmental conditions. The interaction between *U. europaeus* invasion and landscape heterogeneity and dynamism still remains unclear in south-central Chile. To address these issues, the aims of this study were (a) to describe and analyse the spatial and temporal dynamics of the landscape, as well as its relationship with the invasion of *U. europaeus*; (b) to fit spatially explicit models to identify landscape and human factors that explain invasion; and (c) to predict areas most likely related with processes of increase, decrease and permanence of *U. europaeus*. To our knowledge, this is the very first study in addressing the spatial modelling of *U. europaeus* and how it relates to landscape characteristics.

Methods

Study area

The study was carried out in the south-western area of Los Ríos Region, Chile in the districts of Valdivia, Corral, Paillaco and La Unión (Fig. 1). The area covers approximately 320,000 ha and includes coastal plains, the Coastal Range and the Central Valley at elevations ranging from 0 to 1000 m a.s.l. This area was selected because it represents a gradient of human disturbance ranging from landscapes dominated by commercial exotic forest plantations to pristine landscapes dominated by natural forest (mainly old growth forests) within protected areas. Therefore, the study area has protected areas dominated by large fragments of natural forest, and small- and medium-sized native forest fragments within a mosaic of intensive exotic forest plantations, arable land, pastures for livestock grazing and urban settlements. The prevailing climate is wet temperate with Mediterranean influence. The mean annual temperature is 11.5 °C, with high precipitations of ca. 2600 mm, which are concentrated in the winter months (i.e. June–September; Dirección Meteorológica de Chile 2015). Two different types of soil occur in the area: metamorphic and derived from ancient volcanic ash deposits, which mostly have medium/low fertility, less than 50 cm depth and the topography influence strongly as its variability (Thiers et al. 2014).

The main economic activities in the area are tourism, agriculture and forestry (Gobierno Regional de Los Ríos 2010). Forestry is characterized by commercial timber plantations of exotic species such as *Pinus radiata* and *Eucalyptus* spp. In the last 30 years, the forestry industry has increased sharply through the establishment of commercial plantations of exotic species to supply raw material to the pulp and timber industry, facilitated through afforestation incentives (Lara et al. 2006). The main types of land tenure are properties owned by forest companies, private protected areas, small properties, large properties (i.e. >200 ha) and public protected areas (Zamorano-Elgueta et al. 2015).

Data of land cover and landscape variables

Land cover maps

We used two sets of satellite images of 30 m spatial resolution: one Landsat 5 TM (Thematic Mapper) of 29 September 1986 and a Landsat 7 Enhance Thematic Mapper Plus (ETM+) of 3 October 2003, Path-Row 233–088. The dates were selected to correspond to the peak of spring flowering, giving an intense pale yellow pattern (Fig. 2), which allowed the distinction of the species from other types of scrub in the area (Shepherd and Lee 2002).

The images were subject to standard geometric, radiometric and atmospheric correction procedures (Lillesand et al. 2008). In order to eliminate the noise effects of relief, we excluded areas with very abrupt topography, which represented a minimal section of the study area. Due to the presence of clouds and banding resulting from damage to the ETM+ sensor in the 2003 image, these features where masked out from both images. Currently there are several methods of satellite imagery classification (e.g. object based, segmentation) (Lillesand et al. 2008). We used a supervised classification based on the maximum likelihood algorithm (Lillesand et al. 2008). This method has been mostly used in land use/land cover studies in south-central Chile (e.g. Altamirano et al. 2013; Miranda et al. 2015). Seven classes were defined: native forest, forest plantations of exotic species, scrub, agricultural land, bare land, water and *U. europaeus* scrub. To identify the classes, we used expert classification based on data from the National Vegetation Survey (CONAF et al. 1999; CONAF-CONAMA 2008) and field data. To smooth out the classification, a 3 × 3 pixel mode filter was applied and a minimum area of 9 pixels. The accuracy of classification for each image was assessed

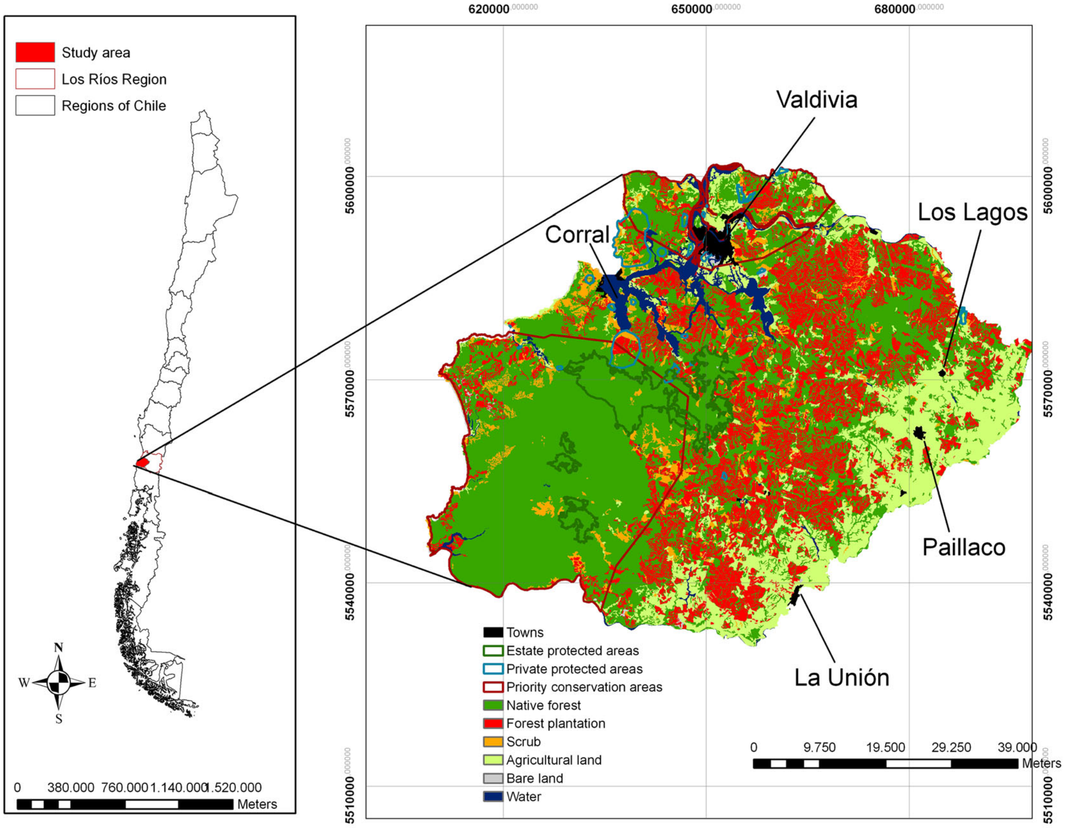

Fig. 1 Location map of the study area in Los Ríos Region, Chile. Source: Map obtained from data of National Vegetation Survey (CONAF-CONAMA, 2008)

by constructing a confusion matrix (Congalton and Green 1999). We used a stratified sample which was validated using photo-interpretation of images from Google Earth (Google Earth Image R 2005 Digital Globe) and field information from the study area. The overall accuracy of satellite images classification was 91 and 93 % for the 1986 and 2003 images, respectively (see matrix details in Appendix 1).

We produced two land cover maps one for 1986 and one for 2003. From these, we also generated binary presence-absence maps of *U. europaeus* for both dates to construct the dependent variable layers. Using the binary *U. europaeus* data, we constructed three maps to be used as dependent variables for the statistical modelling phase:

1. *Increase*: change from any non-*U. europaeus* land cover to *U. europaeus*.
2. *Decrease*: change from *U. europaeus* to any other land cover.
3. *Permanence*: where *U. europaeus* cover was maintained in both dates.

Environmental and socioeconomic data

In order to model the presence and changes of *U. europaeus* shrublands in the study period, we constructed a set of explanatory variables from spatially explicit data on land use/cover, topographical conditions and human activity collected from various sources (Table 1). All these data layers were fitted to the grid of maps land cover maps generated above.

Land use/cover variables Data were derived from the 1986 land cover map. We calculated the distance from sources of *U. europaeus* presence (seed source) to the main land covers (native forest, forest plantations, scrub, agricultural land, bare land) using the Euclidian distance method. Furthermore, we measure landscape heterogeneity

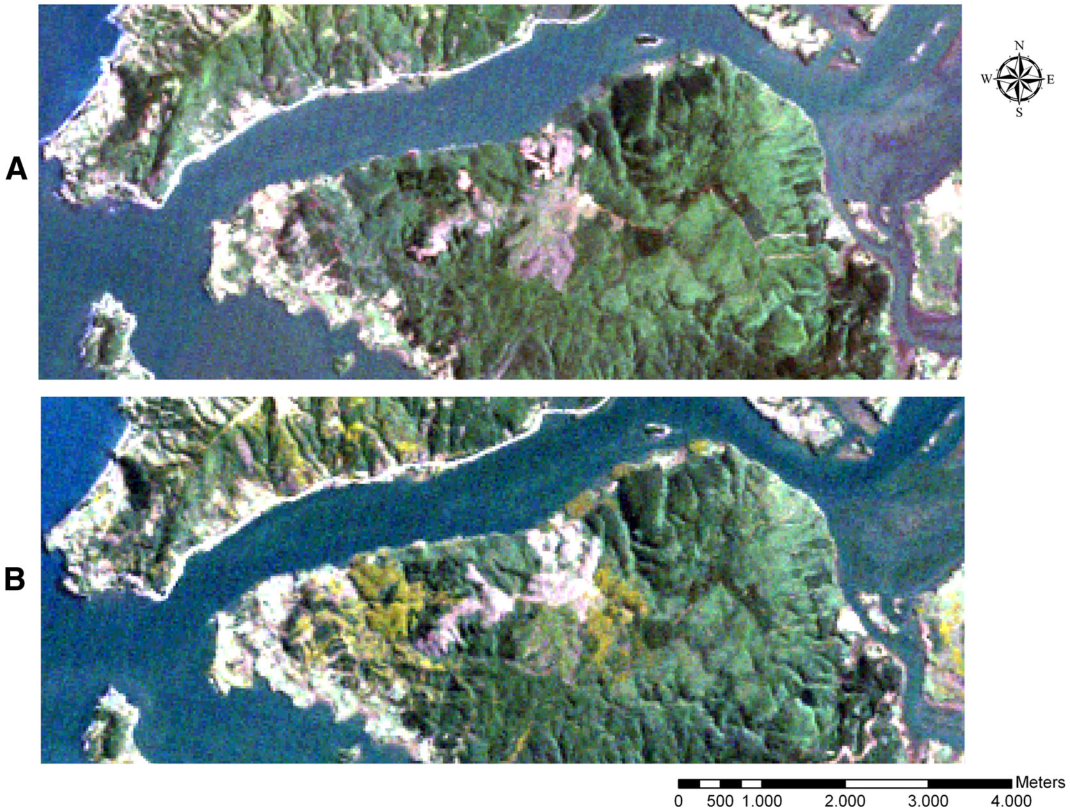

Fig. 2 Differences in the spectral response of *U. europaeus* in winter (June) (**a**) and spring (September) (**b**) 1986. The yellow colour corresponds to flowering of the species. Source: Satellite imagery Landsat 5 TM (Thematic Mapper)

Table 1 Landscape variables used to fit the increase, decrease and permanence models for *U. europaeus*

Variable	Min	Max	Source
Land use/soil cover variables			
Distance to native forest (m)	0	29,056	Map of soil covers based on Landsat image 1986
Distance to forestry plantations (m)	0	29,445	Map of soil covers based on Landsat image 1986
Distance to scrub (m)	0	29,075	Map of soil covers based on Landsat image 1986
Distance to farmland (m)	0	28,841	Map of soil covers based on Landsat image 1986
Distance to bare soil (m)	0	28,565	Map of soil covers based on Landsat image 1986
Distance to seed source (m)	0	32,207	Map of soil covers based on Landsat image 1986
Landscape complexity index (n.a.)	1	10	Map of soil covers based on Landsat image 1986
Topographical variables			
Elevation (m)	0	1042	Shuttle radar topography mission—SRTM
Slope (degrees)	0	60	Shuttle radar topography mission—SRTM
Aspect(n.a.)	−1	1	Shuttle radar topography mission—SRTM
Human activity variables			
Frequency of fires (number)	0	30	Corporación Nacional Forestal (CONAF)
Distance to towns (m)	0	58,789	Catastro de recursos vegetacionales (CONAF)
Distance to rivers (m)	0	32,742	Dirección General de Aguas (DGA)
Distance to roads (m)	0	35,192	Ministerio de Obras Públicas (MOP)

through a landscape complexity index. Some indexes have been proposed to represent landscape heterogeneity (McGarigal et al. 2012). Here we devised a landscape complexity (LC) index to characterize the landscape heterogeneity. This index is the quotient between the number of classes and the number of pixels of the dominant class in a moving window of 450 × 450 m.

$$\mathrm{LC} = \text{number classes} \Big/ \text{number of pixels of dominant class}$$

Topographical variables We used a digital elevation model of 90 m spatial resolution derived from the Shuttle Radar Topography Mission (SRTM), from which we calculated the variables of elevation, slope and aspect for inclusion in the model as predictor variables. The SRTM data were obtained from the Global Land Cover Facility (GLCF) website (http://www.landcover.org).

Human activity variables To characterize human activity and impacts, we used the variables of distance to towns, rivers and roads, using the National Vegetation Survey dataset (CONAF et al. 1999; CONAF-CONAMA 2008), just as with the land use/cover variables, calculated as Euclidian distance.

Change dynamic of land use/cover

Transitions between the different classes of land use/cover were calculated using the Land Change Modeller module in the IDRISI-Selva software (Eastman 2012). These transitions were compared between classes by a change matrix, obtaining the specific flows affecting losses and gains during the 17-year time period.

Statistical analysis

In order to avoid spatial autocorrelation, the study area was divided into 1-km^2 quadrants. Through this, we obtained the available data for each of the three types of processes being studied, i.e. increase (n = 1062), decrease (n = 1066) or permanence (n = 1062) of *U. europaeus*. We fit a logistic regression model for each of the processes. Because this type of model is affected by unbalanced data (e.g. larger proportion of one type of data; Maddala 1992), each database was set to have the same proportion of zeros (absence) and ones (presence) by randomly selecting the same amount of 0 and 1 observations in each database.

We reduced the number of potential predictor variables by excluding those that were highly correlated ($r > 0.6$). This was the case with distance to agricultural land that was highly correlated with elevation, distance to forest plantations, distance to bare land and distance to towns. The remaining variables (Table 1) were assessed as predictors by fitting logistic regression models for modelling each of the processes (increase, decrease or permanence of *U. europaeus*). We fit the models by maximum likelihood using a 70 % of each database, and the remaining observations were reserved for validation purposes.

The best model for each process was obtained by comparing the Akaike Information Criterion (AIC) index among several model alternatives. Furthermore, we evaluated the performance of each of the final models by computing the area under the curve (AUC) of the receiver operating characteristic (ROC), as well as calculating the sensitivity and specificity indicators representing the percentage of correctly classified presences and absences (Pearce and Ferrier 2000). The cutoff probability of occurrence of the process being modelled was 0.5 (Liu et al. 2005).

Results

Landscape change

The results of the land use/cover change analysis show that the area occupied by forests and scrub remained unchanged, while the cover of forest plantations increased and the area of bare land diminished (Table 2). These transitions show the dynamics of the forest and agricultural industries through mechanisms such as clear cutting or preparation of land for sowing crops or for grazing, which reflected in an increase in the cover of arable land and meadows between 1986 and 2003.

Figure 3 shows the transitions expressed as net losses and gains between the various land covers of the study area. Forest plantation was the most stable land cover, with transition most frequently to bare land and, to a lesser degree, to *U. europaeus* scrub. Native forest also remained stable with loss coming mainly through degradation to scrub, and, to a lesser degree, conversion to forest plantations. Strong transitions were observed in bare land and scrub, and in agricultural land.

Although *U. europaeus* scrub does not represent a significant area of the landscape (1.2 and 0.7 % in 1986

Table 2 Area and percentage of land covers for 1986 and 2003 in the study area

Land use/cover	1986		2003	
	Area (ha)	%	Area (ha)	%
Native forest	112,629	34.9	112,729	35.0
Forest plantations	55,833	17.3	78,937	24.5
Scrub	49,548	15.4	48,080	14.9
Agricultural land	18,180	5.6	44,529	13.8
Bare land	76,097	23.6	30,756	9.5
U. europaeus scrub	3741	1.2	2147	0.7

and 2003 respectively), it was a very dynamic land cover. It presented a reduction of 42.6 % from 3741 to 2147 ha of the study area (Table 2). Actually, it was the most dynamic land cover of the landscape with an unchanged area in only 9.6 % of its initial area. Large areas transitioned to forest plantations (54.9 %), scrub (22.5 %) and, to a lesser degree, bare land (9.8 %). The land covers which transitioned to *U. europaeus* were mainly scrub and bare land, and to a lesser degree, forest plantations and agricultural land.

Landscape variables related with invasion by *U. europaeus*

The "increase model" showed that there is a significant negative relationship between the expansion of *U. europaeus* and the distance from the seed source (invasion focus), as well as a significant positive relationship between expansion of *U. europaeus* and landscape heterogeneity, measured by the landscape complexity index. The "decrease model", in turn, showed the disappearance of *U. europaeus* to be positively related with distance to towns and negatively with the frequency of fires and the landscape complexity index. The invasion of *U. europaeus* was lower in sites which are more homogeneous in terms of land cover, less disturbed by fire and further away from urban centres. On the other hand, permanence of *U. europaeus* scrub, as described by the "permanence model", was negatively affected by distance from towns and positively by the frequency of fires and the landscape complexity index. The possibility of *U. eropaeus* scrub remaining stable was higher in places frequently disturbed by fire, with highly heterogeneous land cover and close to urban centres.

This must be added to the fact that the variables of distance to forest plantations, distance to bare land and distance to towns were more directly related to the invasion by *U. europaeus*.

Predicting increase, decrease and permanence of *U. europaeus*

The process of expansion showed the best prediction as expressed by the increase model, with the best goodness of fit and greater precision (Table 3). The values for sensitivity (fraction of true positives) and specificity (fraction of true negatives) were also higher in the increase model. The permanence model had the highest number of omission errors and the decrease model the highest number of commission errors (Table 3). However, all three models (increase, decrease and permanence) presented AUC values greater than 0.7 that allow for an acceptable distinction between the presences and absences. The overall accuracy for the increase model was substantially higher than for the decrease and permanence models at 89.6 % compared to 70.0 and 72.4 % respectively. Decrease and permanence showed larger errors of omission and commission offering lower reliability.

The prediction map for the increase model of *U. europaeus* (Fig. 4a) shows that the species has already started to become established (foci of invasive scrub have appeared during the study period) in parts of the Valdivia National Reserve, belonging to the National System of State-Protected Wilderness Areas (SNASPE), the Coastal Range and Curiñanco priority areas and several private protected areas. The prediction map for decrease model (Fig. 4b) shows areas at the south-west of the study area, with populations of *U. europaeus* showing a lower potential of invasion. This area corresponds to a large part of the Coastal Range priority site and the Coastal Alerce National Park, both included in SNASPE. Finally, the prediction map for "permanence model" of *U. europaeus* (Fig. 4c) shows two sectors, namely the larger one consisting of areas adjoining the city of Valdivia and the smaller around the towns Niebla and Corral.

Discussion

Our research indicates that the invasion of *U. europaeus* during the study period has been strongly influenced by

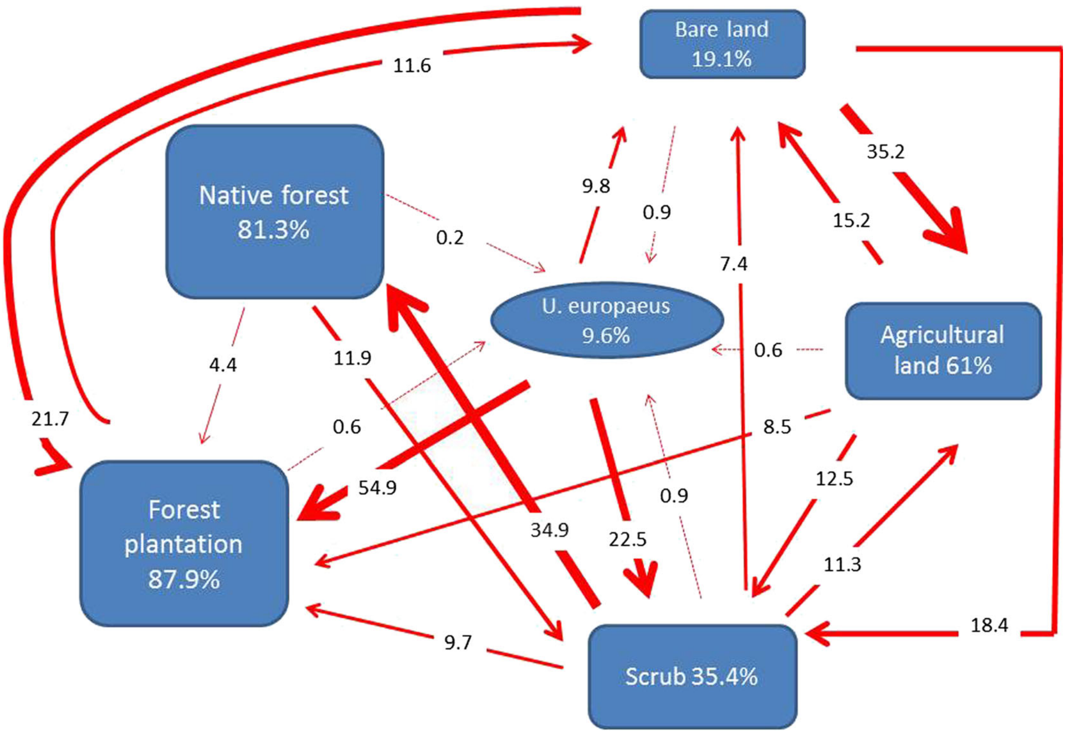

Fig. 3 Land cover transitions in the study area for the period 1986–2003. The size of the *arrows* indicates the magnitude of the change. The *figures* in each *box* correspond to the percentage of each land cover maintained over the period

landscape context and dynamics, particularly by land covers, such as exotic forest plantations. This reinforces findings from studies in Mediterranean landscapes showing that landscape composition (i.e. land use/cover) represents by far the most important group of variables associated with invasions of exotic plant species (González-Moreno et al. 2013).

Despite decreasing by 43 % in area during the study period, *U. europaeus* scrub was still the most dynamic cover in the landscape. This indicates that invasion foci are taking advantage of niches available in more open and degraded land covers such as bare land, agricultural land and scrub. This is to be expected and confirms that alterations and change dynamics constitute ideal scenarios for

Table 3 Statistical models for increase, decrease and permanence of *U. europaeus*

Model	Variable	Coefficient	SE	Pr(>|z|)	AIC	AUC	Sensitivity	Specificity
Increase	Intercept	0.3475	0.2425	0.1520	573.0	0.93	0.95	0.83
	Distance to seed source	−0.0011	0.0001	<0.0001				
	Landscape complexity index	0.3323	0.0617	<0.0001				
Decrease	Intercept	0.1654	0.2728	0.5442	948.1	0.73	0.80	0.59
	Distance to towns	0.0001	0.0001	<0.0001				
	Frequency of fires	−0.2458	0.0497	<0.0001				
	Landscape complexity index	−0.0883	0.0383	0.0211				
Permanence	Intercept	−0.2831	0.2774	0.3076	987.8	0.74	0.65	0.79
	Distance to towns	−0.0001	0.0001	0.0004				
	Frequency of fires	0.1553	0.0429	0.0003				
	Landscape complexity index	0.1078	0.0389	0.0056				

SE standard error, *AIC* Akaike Information Criterion, *AUC* area under curve

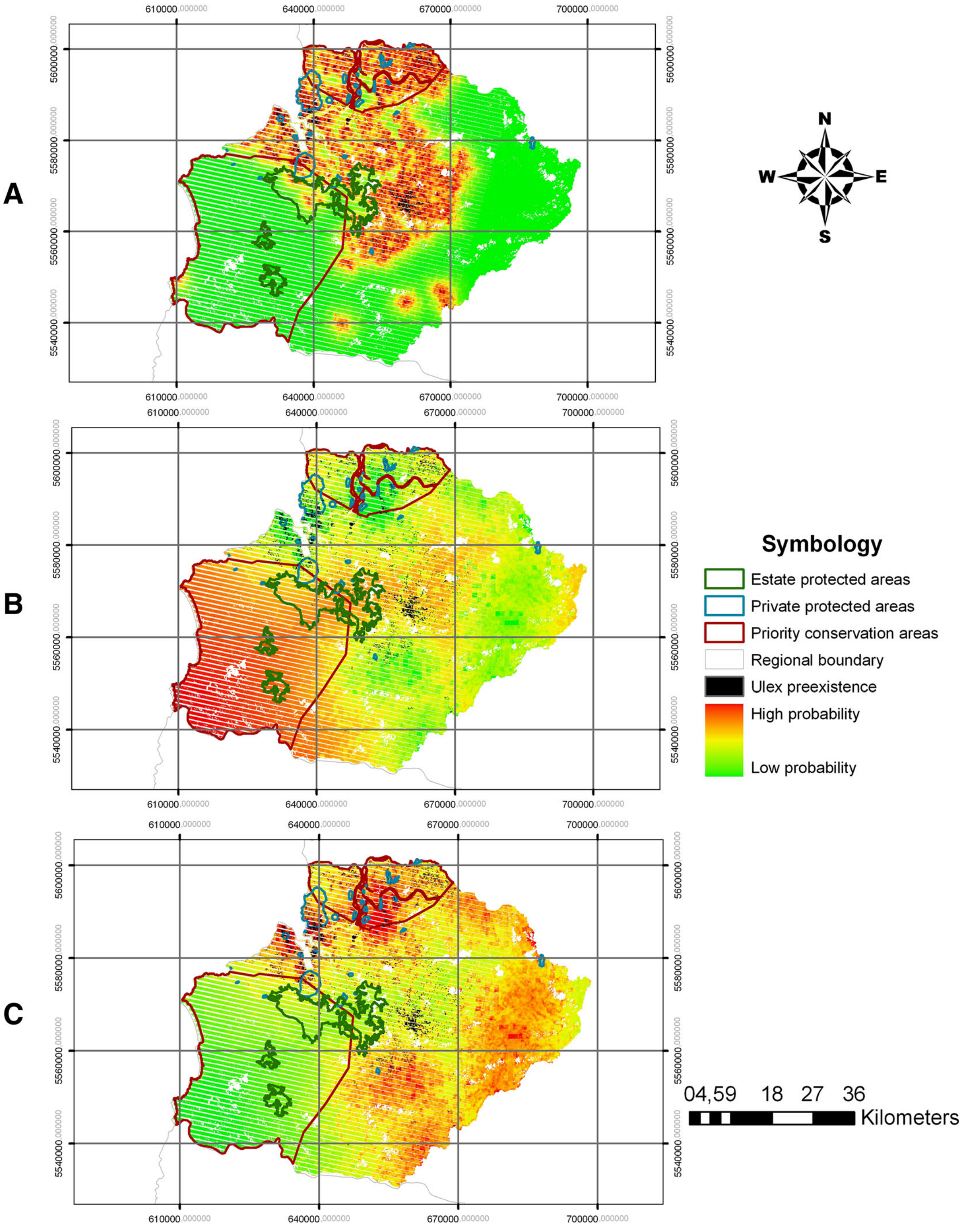

Fig. 4 Prediction maps for the increase (**a**), decrease (**b**) and permanence models for *U. europaeus* (**c**). The areas outlined in *black* are the invasion foci identified in both 1986 and 2003

the establishment of *U. europaeus*, which takes advantage of disturbed or deforested areas and over-grazed meadows (Clements et al. 2001; Prasad 2003).

An interesting finding is the relation of forest plantations and *U. europaeus* establishment. In this land use, more than 50 % of the land, previously invaded, transitioned into exotic forest plantations. An additional considerable proportion (10 %) of *U. europaeus* was transformed into bare land, which mostly represents land under preparation for the establishment of exotic forest plantations or agricultural crops in the near future. Therefore, the interaction observed for *U. europaeus* with exotic forest plantations seems to play an important role in controlling the expansion of *U. europaeus*, although this control may be only temporary. From field observations, we found that forest plantations in their earlier stages (first 5 years) are invaded by *U. europaeus* scrub because the spacing between the *P. radiata* and the *Eucalyptus* sp. seedlings offers suitable light conditions for gorse to become established. This persistent invasion requires constant control, causing management problems and costs during the early stages of plantations establishment. When the plantations are harvested after 15 to 20 years, the soil disturbance caused by clear cutting promotes the germination of buried seeds, even though *U. europaeus* has been absent from the plantation throughout the intervening period (Hill et al. 2001). The availability of suitable land for rapid propagation after the harvest generates new costs associated with soil preparation prior to replanting for the next crop. This dynamic allows new foci to become established which remain latent for a few years and then develop a seed bank which is sufficient to maintain the invasion viability.

Arable and livestock farming activities also interact strongly with the dynamic of *U. europaeus*. Land abandoned after use for these activities presents ideal conditions for the establishment of gorse. In addition, the practice of using the species to build hedges around fields provides a permanent source of seeds, which allows the invasion to spread to other cultivated areas. To mitigate the invasion process, many farmers burn the land invaded by *U. europaeus*. However, some studies have reported a significant increase in the germination of the species from the seed bank in areas recently affected by fire (Pauchard et al. 2011). In this way, fire perversely might rather help strengthen the invasion process, enhancing germination and the appearance of new shoots after burning (García et al. 2010). This is confirmed by our results that show a significant negative relation between frequency of fires with decrease and a positive relation with permanence of *U. europaeus* in the predictive models. On the other hand, *U. europaeus* scrub areas may promote the occurrence of forest fires because of modifications to the regime of ecosystem disturbances. Hill et al. (2001) and Zouhar (2005) indicate how fire helps to strengthen mono-specific areas of *U. europaeus* scrub as it is a pyrophile species (i.e. it is fire-resistant and spreads more readily than other species after burning). Because gorse scrub accumulates large quantities of necromass (dead spines) under the canopy and contains little humidity (the species has no leaves as they have all evolved into spines), it favours the occurrence and spread of fire. Land subject to reiterated fires appears to maintain more viable populations of *U. europaeus* than land where no fires occur. Similar pattern has been observed in *T. monspessulana* in south-central Chile (García et al. 2010).

Landscape dynamic involves the existence of landscape spatial and temporal heterogeneity (Turner et al. 2001), which is also very important for invasion dynamic. Melbourne et al. (2007) point out that environmental heterogeneity increases susceptibility to invasion because it promotes the invasion and the co-existence of mechanisms which are not possible in more homogeneous landscapes. We found that areas with higher heterogeneity as expressed by the landscape complexity index are more likely to be invaded and present also higher levels of permanence. These results are also consistent with the study of Vilá et al. (2010), who argue that the primary cause of a lower level of invasion by exotic plants may be lower heterogeneity of the landscape and less pressure from propagules.

Factors related with human activity (i.e. distance to towns and frequency of fires) are good predictors of both decrease and permanence of *U. europaeus*, giving support to the close relationship between this species and anthropogenic disturbance (Norambuena et al. 2007; Muñoz 2009). Indeed, the permanence model shows two sectors with high probability of permanence, which have suffered greater transformation due to the larger concentrations of human population. The reason is that proximity of population centres has more open and disturbed spaces with which offers empty niches for the propagation of the species, as well as effective seed dispersal mechanisms through human mediation. This pattern also is confirmed in the recent research of Cordero et al. (2016), where they concluded that disturbances (e.g. urban development) associated to roadsides and adjacent habitat patches were the most important factors explaining gorse occurrence. However, contrary to expectation, in our research, no influence of distance from roads was observed for the invasion increase model, as shown in the previous study. There are two possible causes for this: first, the road map used in the analysis only included main and secondary roads but not smaller roads or forest tracks. Second, the resolution of the satellite images was probably too coarse for the identification of *U. europaeus* scrub used for small hedgerows.

The increase model of *U. europaeus* was also the most parsimonious, with two predictor variables. The high sensitivity of this model shows few errors of omission in the model fit, predicting occurrences in places where they do in fact occur. The variable distance from seed source is one of the most important variables, indicating that the vicinity of infestation foci is crucial for the process of invasion. This is explained by the dispersal strategy of autochory, characteristic of the species, resulting in seed dispersal over very short distances of 2 to 6 m (Ríos, 2005). Furthermore, the seeds present high establishment potential and survival rates based on a persistent seed bank with seeds that remain viable in the soil for long periods of time (Hill et al. 2001).

Remote sensing was very useful for detecting, mapping and monitoring invasions, especially in this species due to its morphological characteristics (Rew et al. 2005). Shepherd and Lee (2002) used ETM+ landsat images to distinguish *U. europaeus* scrub during flowering in New Zealand and report that the method is a precise, practical approach for determining the extent of the species on a regional scale. In this study, we used images taken at the peak of the flowering season of *U. europaeus* in the region, allowing us to distinguish the zones which had suffered heavy invasion (scrub in flower) from the surrounding vegetation. However, this study only identified foci covering at least 9 pixels. Smaller patches cannot always be reliably identified using this spatial resolution. Small invasion foci, for example on roadsides, seem to play an important role in the spread of the species (Pauchard and Alaback 2004).

The results of invasion models show the potential threat to protected zones in the study area. Although the moderate degree of homogeneity in areas dominated by native forest may contain the advance of this invasive species, the greater susceptibility to invasion of low density native scrub species, cleared areas, forest edges, river banks and anthropized zones could affect conservation efforts. This process appears to be related with forest fragmentation, deforestation or degradation around conservation areas (Zamorano-Elgueta et al. 2015) and represent drivers for the establishment of *U. europaeus* scrub foci, being a threat to the preservation of areas in their natural state.

Although information exists on the displacement of native flora by *U. europaeus* (Baret et al. 2006), its other effects and interactions with other groups of native species are still unknown. For example the arrival of this species in natural areas may signify the start of links with insects, mammals and birds and thus affect ecological processes such as seed dispersal, ecological succession and trophic chains. Research done in places where gorse has been catalogued as invasive has found that some species of native fauna may benefit from the invasion. For example endemic lizards and some invertebrates in Sri Lanka find food and refuge (Somaweera et al. 2012). Amaya-Villarreal and Renjifo (2010) report greater abundance of the native bird species *Diglossa humeralis* and *Basileuterus nigrocristatus* along forest edges invaded by *U. europaeus* in the high Andes area of Colombia. Harris et al. (2004), in a study done in New Zealand, found that the *U. europaeus* habitat was richer in native invertebrate groups than local ecosystems. Other studies also point to the important successional role of *U. europaeus* as a precursor of native forest in revegetation projects (Williams and Karl 2002).

Finally, the findings of the present research, like those of other studies (González-Moreno et al. 2013), indicate that proper management of this invasive species must consider the spatial landscape dynamic of the area invaded. This will contribute to the formulation of containment, control or mitigation plans.

Conclusion

Although we found a decrease in the invaded area of *U. europaeus* within the study period, these results are indeed novel because of the interaction of forest plantations with gorse at landscape scale. Gorse was the most dynamic of the analysed land use covers, being positively influenced by the landscape complexity and particularly by forest plantations. Expansion of forest plantations in south-central Chile seems to diminish the spread of *U. europaeus*, but at the same time, they might present some negative impacts for the conservation of natural areas (Simonetti et al. 2007). Proper monitoring of *U. eropaeus* invasion dynamics at landscape level may result in useful insights for management decision making; for example gorse permanence resulted more likely in areas affected by reiterated fires, and attention should be put to the observed expansion of this invasive species inside public and private protected areas.

Since *U. europaeus* is one the most invasive species in the world, this research constitutes the first attempt to characterize the invasion of gorse at landscape scale, and as such, land managers and conservation agencies should benefit from these findings.

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Appendix

Table 4 Confusion matrices of satellite imagery classification: (a) 1986, (b) 2003

Class	NF	FP	SC	AL	BL	W	UE	Total	User's accuracy (%)
(a)									
Native forest (NF)	44	2	4	0	0	0	0	50	88
Forest plantations (FP)	0	42	6	2	0	0	0	50	84
Scrub (SC)	2	1	47	0	0	0	0	50	94
Agricultural land (AL)	0	0	1	48	1	0	0	50	96
Bare land (BL)	0	0	0	1	49	0	0	50	98
Water (W)	0	0	0	0	0	50	0	50	100
U. europaeus scrub (UE)	2	0	9	0	0	0	39	50	78
Total	48	45	67	51	50	50	39	350	
Producer's accuracy (%)	92	93	70	94	98	100	100		
Overall accuracy (%)	91								
(b)									
Native forest	44	1	5	0	0	0	0	50	88
Forest plantations	1	47	2	0	0	0	0	50	94
Scrub	1	5	43	0	0	0	1	50	86
Agricultural land	0	0	1	48	1	0	0	50	96
Bare land	0	0	0	1	49	0	0	50	98
Water	0	0	0	0	0	50	0	50	100
U. europaeus scrub	0	0	7	0	0	0	43	50	86
Total	46	53	58	49	50	50	44	350	
Producer's accuracy (%)	96	89	74	98	98	100	98		
Overall accuracy (%)	93								

References

Altamirano, A., Aplin, P., Miranda, A., Cayuela, L., Algar, A., & Field, R. (2013). High rates of forest loss and turnover obscured by classical landscape measures. *Applied Geography, 40*, 199–211.

Amaya-Villarreal, A. M., & Renjifo, L. M. (2010). Effects of gorse (*Ulex europaeus*) on the birds of a high Andean forest edge. *Ornitologia Colombiana, 10*, 11–25.

Arroyo, M. T., Marquet, P., Marticorena, C., Simonetti, J., Caviares, L., Squeo, F., Roíz, R., & Massardo, F. (2008). El hotspot chileno, prioridad mundial para la conservación. In J. Rovira, J. Ugalde, & M. Stutzin (Eds.), *Biodiversidad de Chile, patrimonio y desafíos* (pp. 90–93). Santiago: CONAMA, Ocho Libros Editores.

Arroyo, M. T., Marticorena, C., Matthei, O., & Cavieres, L. (2000). Plant invasions in Chile: present patterns and future predictions. In H. A. Mooney & R. Hobbs (Eds.), *Invasive* *species in a changing world* (pp. 385–421). Washington: Island Press.

Atlan, A., Schermann-Legionnet, A., Udo, N., & Tarayre, M. (2015). Self-incompatibility in *Ulex europaeus*: variations in native and invaded regions. *International Journal of Plant Sciences, 176*, 515–524.

Baret, S., Rouget, M., Richardson, D. M., Lavergne, C., Egoh, B., Dupont, J., & Strasberg, D. (2006). Current distribution and potential extent of the most invasive alien plant species on La Réunion (Indian Ocean, Mascarene islands). *Austral Ecology, 31*, 747–758.

Blackburn, T. M., Essl, F., Evans, T., Hulme, P. E., Jeschke, J. M., Kuhn, I., et al. (2014). A unified classification of alien species based on the magnitude of their environmental impacts. *Plos Biology, 12*(5), e1001850.

Chiou, C., Wang, H., Chen, Y., Grant, W., & Lu, M. (2013). Modeling potential range expansion of the invasive shrub *Leucaena leucocephala* in the Hengchun Peninsula, Taiwan. *Invasive Plant Science and Management, 6*, 492–501.

Chytrý, M., Wild, J., Pyšek, P., Jarošík, V., Dendoncker, N., Reginster, I., et al. (2012). Projecting trends in plant invasions in Europe under different scenarios of future land-use change. *Global Ecology and Biogeography, 21*, 75–87.

Clements, D. R., Peterson, D. J., & Prasad, R. (2001). The biology of Canadian weeds. *Ulex europaeus* L. *Canadian Journal of Plant Science, 81*, 325–337.

CONAF (Corporación Nacional Forestal, CL), CONAMA (Comisión Nacional del Medio Ambiente, CL), BIRF (Banco Internacional de Reconstrucción y Fomento, USA), Universidad Austral de Chile, Pontificia Universidad Católica de Chile, Universidad Católica de Temuco. (1999). *Catastro y evaluación de los recursos vegetacionales nativos de Chile*. Santiago: Corporación Nacional Forestal

CONAF & CONAMA (Corporación Nacional Forestal & Comisión Nacional del Medioambiente). (2008). Catastro de uso de suelo y vegetación. Monitoreo y actualización. Región de Los Ríos, periodo 1998–2006. Santiago: Corporación Nacional Forestal.

Congalton, R.G. & Green, K. (1999). Assessing the accuracy of remotely sensed data: principles and practices. Florida: Lewis Publishers, Boca Raton.

Cordero, R. L., Torchelsen, F. P., Overbeck, G. E., & Anand, M. (2016). Analyzing the landscape characteristics promoting the establishment and spread of gorse (*Ulex europaeus*) along roadsides. *Ecosphere, 73*, e01201.

D'Antonio, C. M., & Vitousek, P. M. (1992). Biological invasions by exotic grasses, the grass/fire cycle, and global change. *Annual Review of Ecology and Systematics, 23*, 63–87.

Dinerstein, E., Olson, D.M., Graham, D.J., Webster, A.L., Primm, S.A. & Bookbinder, M.P. (1995). *Una evaluación del estado de conservación de las ecoregiones terrestres de América Latina y el Caribe*. Washington: Banco Mundial/WWF.

Dirección Meteorológica de Chile. (2015). Climas de Chile. Dirección Meteorológica de Chile. http://www.meteochile.gob.cl/climas/climas_novena_region.html#b. Accessed 10 Oct 2015.

Eastman, J. R. (2012). *IDRISI selva*. Worcester: Clark University.

Fuentes, N., Pauchard, A., Sánchez, P., Esquivel, P., & Marticorena, A. (2013). A new comprehensive database of alien plant species in Chile based on herbarium records. *Biological Invasions, 15*, 847–858.

Fuentes-Ramirez, A., Pauchard, A., Marticorena, A., & Sánchez, P. (2010). Relación entre la invasión de *Acacia dealbata* Link (Fabaceae: Mimosoideae) y la riqueza de especies vegetales en el centro-sur de Chile. *Gayana Botanica, 67*, 176–185.

Fuentes-Ramirez, A., Pauchard, A., Cavieres, L. A., & García, R. A. (2011). Survival and growth of *Acacia dealbata* vs. native trees across an invasion front in south-central Chile. *Forest Ecology and Management, 261*, 1003–1009.

García, R. A., Pauchard, A., Cavieres, L. A., Peña, E., & Rodriguez, M. F. (2010). El fuego favorece la invasión de Teline monspessulana (Fabaceae) al aumentar su germinación. *Revista Chilena de Historia Natural, 83*, 443–452.

García, R., Pauchard, A., & Escudero, A. (2014). French broom (Teline monspessulana) invasion in south-central Chile depends on factors operating at different spatial scales. *Biological Invasions, 16*, 113–124.

García, R. A., Fuentes-Ramírez, A., & Pauchard, A. (2012). Effects of two nitrogenfixing invasive species on soil chemical properties in south-central Chile. *Gayana Botánica, 69*(1), 189–192.

González-Moreno, P., Pino, J., Carreras, D., Basnou, C., Fernández-Rebollar, I., & Vila, M. (2013). Quantifying the landscape influence on plant invasions in Mediterranean coastal hábitats. *Landscape Ecology, 28*, 891–903.

Harris, R. J., Toft, R., Dugdale, J. S., Williams, P. A., & Rees, J. (2004). Insect assemblages in a native (kanuka-*Kunzea ericoides*) and an invasive (gorse-*Ulex europaeus*) shrubland. *New Zealand Journal of Ecology, 28*(1), 35–47.

Hill, R. L., Gourlay, A. H., & Barker, R. J. (2001). Survival of *Ulex europaeus* seeds in the soil at three sites in New Zealand. *New Zealand Journal of Botany, 39*(2), 235–244.

Hobbs, R. F., & Huenneke, L. F. (1992). Disturbance, diversity, and invasion: implications for conservation. *Conservation Biology, 6*(3), 324–337.

Hobbs, R. J. (2000). Land use changes and invasions. In H. A. Mooney & R. J. Hobbs (Eds.), *Invasive species in a changing world* (pp. 55–64). Washington DC: Island Press.

Holm, LG., Plucknett, D.L., Pancho, J.V. & Herberger, T.P. (1977). *The world's worst weeds. Distribution and biology.* Honolulu, USA: University Press, [Reprinted 1991 by Krieger Publishing Company, Malabar, Florida]

Hornoy, B., Tarayre, M., Hervé, M., Gigord, L., & Atlan, A. (2011). Invasive plants and enemy release: evolution of trait means and trait correlations in *Ulex europaeus*. *PloS One, 6*(10), e26275.

Keeley, J. E., & Brennan, T. J. (2012). Fire-driven alien invasion in a fire-adapted ecosystem. *Oecologia, 169*, 1043–1052.

Lake, J. C., & Lewisham, M. R. (2004). Invasion success of exotic plants in natural ecosystems: the role of disturbance, plant attributes and freedom from herbivores. *Biological Conservation, 117*, 215–226.

Lara, A., Reyes, R., & Urrutia, R. (2006). Bosques Nativos. In Instituto de Asuntos Públicos & Universidad de Chile (Eds.), *Informe País: Estado del Medio Ambiente en Chile* (pp. 107–139). Santiago: Universidad de Chile, Centro de Análisis de Políticas Públicas.

Leary, J. K., Hue, N. V., Singleton, P. W., & Borthakur, D. (2006). The major features of an infestation by the invasive weed legume gorse (*Ulex europaeus*) on volcanic soils in Hawaii. *Biology and Fertility of Soils, 42*, 215–223.

Lillesand, T., Kiefer, R., & Chipman, J. (2008). *Remote sensing and image interpretation*. New York: John Wiley.

Liu, C., Berry, P. M., Dawson, T. P., & Pearson, R. G. (2005). Selecting thresholds of occurrence in the prediction of species distributions. *Ecography, 28*(3), 385–393.

Lowe, S., Browne, M., Boudjelas, S. & De Poorter, M. (2004). 100 of the world's worst invasive alien species: a selection from the Global Invasive Species Database. Auckland: The Invasive Species Specialist Group (ISSG) a specialist group of the Species Survival Commission (SSC) of the World Conservation Union (IUCN).

Mack, M. C., & D'Antonio, C. M. (1998). Impacts of biological invasions on disturbance regimes. *Trends in Ecology and Evolution, 13*(5), 195–198.

Mack, R. N., Simberloff, D., Lonsdale, W. M., Evans, H., Clout, M., & Bazzaz, F. A. (2000). Biotic invasions: causes, epidemiology, global consequences, and control. *Ecological Applications, 10*(3), 689–710.

Maddala, G. S. (1992). *Introduction to econometrics*. New York: Macmillan Publishing Company.

Mathei, O. (1995). *Manual de las malezas que crecen en Chile*. Santiago: Alfabeta Impresores.

McGarigal, K., Cushman, S.A. & Ene, E. (2012). FRAGSTATS v4: Spatial Pattern Analysis Program for categorical and continuous maps. Computer software program produced by the authors at the University of Massachusetts, Amherst. http://www.umass.edu/landeco/research/fragstats/fragstats.html.

McGeoch, M. A., Butchart, S. H. M., Spear, D., Marais, E., Kleynhans, E. J., Symes, A., et al. (2010). Global indicators of biological invasion: species numbers, biodiversity impact and policy responses. *Diversity and Distributions, 16*, 95–108.

Melbourne, B. A., Cornell, H. V., Davies, K. F., Dugaw, C. J., Elmendorf, S., Freestone, A. L., et al. (2007). Invasion in a heterogeneous world: resistance, coexistence or hostile takeover? *Ecology Letters, 10*(1), 77–94.

Mgidi, T. N., Le Maitre, D. C., Schonegevel, L., Nel, J. L., Rouget, M., & Richardson, D. M. (2007). Alien plant invasions—incorporating emerging invaders in regional prioritization: a pragmatic approach for southern Africa. *Journal of Environmental Management, 84*(2), 173–187.

Miranda, A., Altamirano, A., Cayuela, L., Pincheira, F., & Lara, A. (2015). Different times, same story: native forest loss and landscape homogenization in three physiographical areas of south-central Chile. *Applied Geography, 60*, 20–28.

Mooney, H. A., & Cleland, E. E. (2001). The evolutionary impact of invasive species. *Proceedings of the National Academy of Sciences of United States of America, 98*(10), 5446–5451.

Muñoz, E. (2009). El espinillo (*Ulex europaeus*) un invasor biológico en el sur de Chile: estado de su conocimiento y alternativas de control. *Gestión Ambiental, 17*, 23–44.

Myers, N., Mitterneier, R., Mitterneier, C., Da Fonseca, G., & Kents, J. (2000). Biodiversity hotspots for conservation priorities. *Nature, 403*, 853–858.

Norambuena, H., Martínez, G., Carrillo, R., & Neira, M. (2007). Host specifity and establishment of Tetranychus lintearius (Acari: Tetranychidae) for biological control of gorse, *Ulex europaeus*(Fabaceae) in Chile. *Biological Control, 40*, 204–212.

Norambuena, H., & Piper, G. L. (2000). Impact of Apion ulicis on *Ulex europaeus* seed dispersal. *Biological Control, 17*, 267–271.

Norambuena, H., Escobar, S. & Rodríguez, F. (2000). The Biocontrol of gorse, *Ulex europaeus*, in Chile: a progress report. Montana: Proceedings of the X International Symposium on Biological Control of Weeds.

Pauchard, A., & Alaback, P. B. (2004). Influence of elevation, land use, and landscape context on patterns of alien plant invasions along roadsides in protected areas of south-central Chile. *Conservation Biology, 18*(1), 238–248.

Pauchard, A., García, R., Peña, E., González, C., Cavieres, L., & Bustamante, R. (2008). Positive feedbacks between plant invasions and fire regimes: Teline monspesulana (L) K. Kock (Fabaceae) in central Chile. *Biological Invasions, 10*, 547–553.

Pauchard, A., García, R., Langdon, B., & Fuentes, N. (2011). The invasion of non-native plants in Chile and their impacts on biodiversity: history, current status, and challenges for management. In E. Figueroa (Ed.), *Biodiversity conservation in the Americas: lessons and policy recommendations* (pp. 133–165). Santiago: Editorial FEN-Universidad de Chile.

Pearce, J., & Ferrier, S. (2000). Evaluating the predictive performance of habitat models developed using logistic regression. *Ecological Modelling, 133*, 225–245.

Gobierno Regional de Los Ríos. (2010). Plan Los Ríos 2010–2014. http://www.goredelosrios.cl/directory/files/ctapub/PlanregionalLosRios.pdf. Accessed 20 Apr 2012.

Prasad, R.P. (2003). *Managament and control of gorse and scotch broom in British Columbia*. Victoria, BC: Natural Resources Canada, Canadian Forest Service, Pacific Forestry Centre.

Rees, M., & Hill, R. L. (2001). Large-scale disturbances and biological control and the dynamics of gorse populations. *Journal of Applied Ecology, 38*, 364–378.

Rew, L. J., Maxwell, B. D., & Aspinall, R. (2005). Predicting the occurrence of nonindigenous species using environmental and remotely sensed data. *Weed Science, 53*, 236–241.

Ríos, H.F. (2005). Guía técnica para la restauración ecológica de áreas afectadas por especies vegetales invasoras en el Distrito Capital. Bogotá: Jardín Botánico de Bogotá José Celestino Mutis.

Sala, O. E., Chapin, I. F. S., Armesto, J. J., Berlow, E., Bloomfield, J., Dirzo, R., et al. (2000). Global biodiversity scenarios for the year 2100. *Science, 287*, 1770–1774.

Shepherd, J. D., & Lee, W. G. (2002). Satellite mapping of gorse at regional scales. *New Zealand Plant Protection, 55*, 95–98.

Simonetti, J., Grez, A., Celis-Diez, J., & Bustamante, R. (2007). Herbivory and seedling performance in a fragmented temperate forest of Chile. *Acta Oecologica, 32*, 312–318.

Smith-Ramírez, C. (2004). The Chilean coastal range: a vanishing center of biodiversity and endemism in South American temperate rain forests. *Biodiversity and Conservation, 13*, 373–393.

Somaweera, R., Wijayathilaka, N., & Bowatte, G. (2012). Does the invasive shrub *Ulex europaeus* benefit an endemic Sri Lankan lizard? *Herpetological Conservation and Biology, 7*(2), 219–226.

Thiers, O., Reyes, J., Gerding, V., & Schlatter, J. E. (2014). Suelos en ecosistemas forestales. In C. Donoso, M. E. González, & A. Lara (Eds.), *Ecología forestal. Bases para el manejo* *sustentable y conservación de los bosques nativos de Chile* (pp. 133–178). Valdivia: Ediciones UACh.

Turner, M., Gardner, R. & O'Neill, R. (2001). Landscape ecology in theory and practice. Patterns and process. New York: Springer.

Vilá, M., Basnou, C., Pysek, P., et al. (2010). How well do we understand the impacts of alien species on ecosystem services? A pan-European, cross-taxa assessment. *Frontiers in Ecology and the Environment, 8*, 135–144.

Vitousek, P. M., D'Antonio, C. M., Loope, L. L., Rejmánek, M., & Westbrooks, R. (1997). Introduced species: a significant component of human caused global change. *New Zealand Journal of Ecology, 21*(1), 1–16.

Von Der Lippe, M., & Kowarik, I. (2007). Long-distance dispersal of plants by vehicles as a driver of plant invasions. *Conservation Biology, 21*, 986–996.

Wilcove, D. S., & Chen, L. Y. (1998). Management costs for endangered species. *Conservation Biology, 12*(6), 1405–1407.

Williams, P. A., & Karl, B. J. (2002). Birds and small mammals in kanuka (*Kunzea ericoides*) and gorse (*Ulex europaeus*) scrub and the resulting seed rain and seedling dynamics. *New Zealand Journal of Ecology, 26*(1), 31–41.

Zamorano-Elgueta, C., Rey Benayas, J. M., Cayuela, L., Hantson, S., & Armenteras, D. (2015). Native forest replacement by exotic plantations in southern Chile (1985–2011) and partial compensation by natural regeneration. *Forest Ecology and Management, 345*, 10–20.

Zouhar, K. (2005). *Ulex europaeus*. Fire Information System (FEIS). http://www.fs.fed.us/database/feis/. Accessed 20 July 2012.

Zuloaga, F.O., Morrone, O. & Belgrano, M. (2008). Catálogo de las Plantas Vasculares del Cono Sur. Monographs in Systematic Botany 107. St. Louis, Missouri: Missouri Botanical Garden Press.