Aníbal Pauchard, Rafael García, Sergio Zalba, Mauro Sarasola, Rafael Zenni, Silvia Ziller, Martín A. Nuñez

14 Pine Invasions in South America: Reducing Their Ecological Impacts Through Active Management

14.1 Introduction

Conifers, and specifically the Pinaceae family, have been one of the most visible and studied plant taxa in invasion biology (e.g., Richardson *et al.*, 1994; Rejmánek & Richardson, 1996; Ledgard, 2001; Essl *et al.*, 2011; Buckley *et al.*, 2005; Gundale *et al.*, 2014). Conifer invasions have several characteristics that make them appealing for ecologists, biogeographers, conservationists, and invasion biologists (Gundale *et al.*, 2014). First, conifers have been widely introduced and extensively planted as a forestry crop and ornamental species all around the world (Richardson, 2006; Simberloff *et al.*, 2010; Essl *et al.*, 2010). Second, most conifer species have attributes associated with high invasiveness such as large propagule production, small seeds, anemochoric dispersal, and broad ranges of climatic and edaphic tolerance (Rejmánek & Richardson, 1996; Essl *et al.*, 2010). Third, conifer invasions are conspicuous in the landscape and can be studied in the field with simple observational techniques (Richardson, 2006; Visser *et al.*, 2014). Fourth, conifer invasions can have severe impacts on the local biota and ecosystem processes such as changes in water and fire regimes (Simberloff *et al.*, 2010). Although conifers have become invasive across the world, the Southern Hemisphere has been especially affected by their establishment and expansion, particularly in the case of *Pinus* spp. (Simberloff *et al.*, 2010), which, with the exception of one species, did not occur naturally south of the Equator (Lusk, 2008).

Pine invasions are an important ecological and economic problem in different countries of the Southern Hemisphere (Simberloff *et al.*, 2010; Pauchard *et al.*, 2010). Many species of the Pinaceae family have been introduced throughout the world, mainly for forestry use. There are many positive outcomes of these large-scale introductions of non-native species, including timber production, pulp for paper, and job creation. These "positive" aspects of afforestation have been a matter of discussion when contrasted to changes in water regimes and soil characteristics (Jobbagy & Jackson, 2007; Little *et al.*, 2009), and also in light of their effects on human well-being and health (Guadagnin *et al.*, 2009). Additionally, many of these large-scale plantations have spread into adjacent plant communities, frequently colonizing areas of high conservation value. The characteristics that make pines a good choice for forestry, like fast growth and maturation, have also facilitated their expansion outside the planted areas. South Africa, Australia and New Zealand have historically been leading the research on tree invasions. In South America, concerns about the risk of tree invasions started much later, and systematic research has only been pursued in the last decade. The later establishment of large-scale plantations ca. 1970s, compared to the Anglo-Saxon countries, and the incipient environmental movement may have slowed the attention to tree invasions as a potential threat to biodiversity and ecosystem services (Fonseca *et al.*, 2013).

Pine invasions have been studied extensively since the 1980s, when the SCOPE report on biological invasions was published (Drake *et al.*, 1989). Myriad research projects on pine invasions were carried out in temperate and Mediterranean regions of the Southern Hemisphere, especially in South Africa and New Zealand, and largely in temperate grasslands and shrublands, as well as in the Mediterranean Fynbos (Richardson *et al.*, 1994; Higgins & Richardson, 1998; Richardson & Higgins, 1998; Simberloff *et al.*, 2010). Only recently have researchers started to pay close attention to pine invasions in lower latitudes and forest ecosystems, with most of these studies being carried out in South America (Simberloff *et al.*, 2010; Falleiros *et al.*, 2011; Zenni & Ziller, 2011; Zenni & Simberloff, 2013; Zalba & Villamil, 2002). In the Southern Hemisphere, pines were used in forestry plantations in sub-tropical and tropical regions more recently than in temperate regions (Richardson *et al.*, 2008), which probably explains the temporal and regional biases in pine invasion studies. Moreover, the paradigm that pines do not invade forests persisted until recently (Emer & Fonseca, 2010; Zenni & Simberloff, 2013).

In 2007, a group of scientists studying conifer invasions met at Bariloche, Argentina to discuss the present and future of conifer invasions in South America (Richardson *et al.*, 2008). They concluded that the relatively shorter period since massive introduction of conifers in South America was the main cause of the apparent resistance to invasion that was observed in several South American ecosystems, but that many species are showing invasive behavior on the continent (Table 14.1, Figure 14.1, Simberloff *et al.*, 2010). In 2008 and 2009 an international (Argentina, Brazil and Uruguay) project focusing on alien tree invasions in the Pampas Biome of South America highlighted the effects of large-scale plantations as focuses for pine invasion (Guadagnin *et al.*, 2009; Fonseca *et al.*, 2013). Research on conifer invasions can greatly benefit from studying processes at early stages of invasion, such as those in South America, compared to most evidence reported in the literature for areas where conifer invasions are consolidated. Furthermore, in terms of conservation and invasion control, this region still has the opportunity for prevention and adequate management actions to avoid large-scale invasions, saving resources to restore native ecosystems after invasions have occurred. In recent years, new studies have been conducted and evidence is mounting about the impacts that invasive pines are having in South American ecosystems, from microsite effects on plant diversity to large-scale changes in fire regimes. Thus, there is an urgent need for clear guidelines on how to deal with the undesired effects of invasive pines in South America.

Tab. 14.1: Invasive Pinaceae species registered as naturalized and/or invasive in different biomes of South America (modified from Simberloff *et al.*, 2010 based on current literature and records). "?" indicates there is no sufficient information to assess the invasive status.

Biome	Country	Species	Naturalized	Invasive
Tropical humid forests	Brazil	*Pinus caribaea*	no	no
		Pinus chiapensis	no	no
		Pinus elliottii	yes	yes
		Pinus maximinoi	yes	yes
		Pinus oocarpa	no	no
		Pinus patula	no	no
		Pinus taeda	yes	yes
		Pinus tecunumanii	no	no
Subtropical	Brazil	*Pinus caribaea*	no	no
	Argentina	*Pinus caribaea*	?	?
	Brazil	*Pinus chiapensis*	no	no
	Argentina	*Pinus elliottii*	yes	?
	Brazil	*Pinus elliotii*	yes	yes
		Pinus maximinoi	yes	yes
		Pinus oocarpa	yes	yes
		Pinus patula	no	no
		Pinus radiata	no	no
		Pinus serotina	?	?
	Argentina	*Pinus taeda*	yes	?
	Brazil	*Pinus taeda*	yes	yes
		Pinus tecunumanii	yes	yes
Tropical grasslands	Brazil	*Pinus caribaea*	no	no
		Pinus chiapensis	no	no
		Pinus elliottii	yes	yes
		Pinus kesiya	no	no
		Pinus maximinoi	no	no
		Pinus oocarpa	yes	yes
		Pinus patula	no	no
		Pinus pseudo-strobus	no	no
		Pinus taeda	yes	yes
		Pinus tecunumanii	yes	yes

continued **Tab. 14.1:** Invasive Pinacea species registered as naturalized and/or invasive in different biomes of South America (modified from Simberloff *et al.*, 2010 based on current literature and records). "?" indicates there is no sufficient information to assess the invasive status.

Biome	Country	Species	Naturalized	Invasive
Tropical dry forest	Argentina	*Pinus elliottii*	yes	yes
		Pinus halepensis	?	no
		Pinus taeda	yes	?
		Pinus radiata	?	no
Temperate grasslands	Argentina	*Pinus elliottii*	yes	yes
	Brazil	*Pinus elliottii*	yes	yes
	Uruguay	*Pinus elliottii*	?	?
	Argentina	*Pinus halepensis*	yes	yes
		Pinus patula	?	yes
		Pinus pinaster	yes	yes
	Uruguay	*Pinus pinaster*	yes	yes
	Argentina	*Pinus pinea*	no	no
		Pinus radiata	yes	yes
		Pinus roxburghii	?	?
		Pinus taeda	yes	?
	Brazil	*Pinus taeda*	yes	yes
Warm desert	Argentina	*Pinus ponderosa*	yes	no
Evergreen Sclerophyllous Forest	Chile	*Pinus radiata*	yes	yes
		Pseudotsuga menziesii	?	?
Temperate rain forest	Argentina	*Pinus contorta*	yes	no
	Chile	*Pinus radiata*	yes	yes
	Argentina	*Pinus radiata*	yes	?
		Pseudotsuga menziesii	yes	yes
	Chile	*Pseudotsuga menziesii*	yes	yes
Mixed mountain systems	Chile	*Pinus contorta*	yes	yes
		Pinus ponderosa	yes	no
		Pinus radiata	no	no
		Pinus sylvestris	yes	yes
		Pseudotsuga menziesii	yes	yes
Cold winter desert	Argentina	*Pinus contorta*	yes	yes
	Chile	*Pinus contorta*	yes	yes
	Argentina	*Pinus ponderosa*	yes	no
		Pseudotsuga menziesii	yes	no

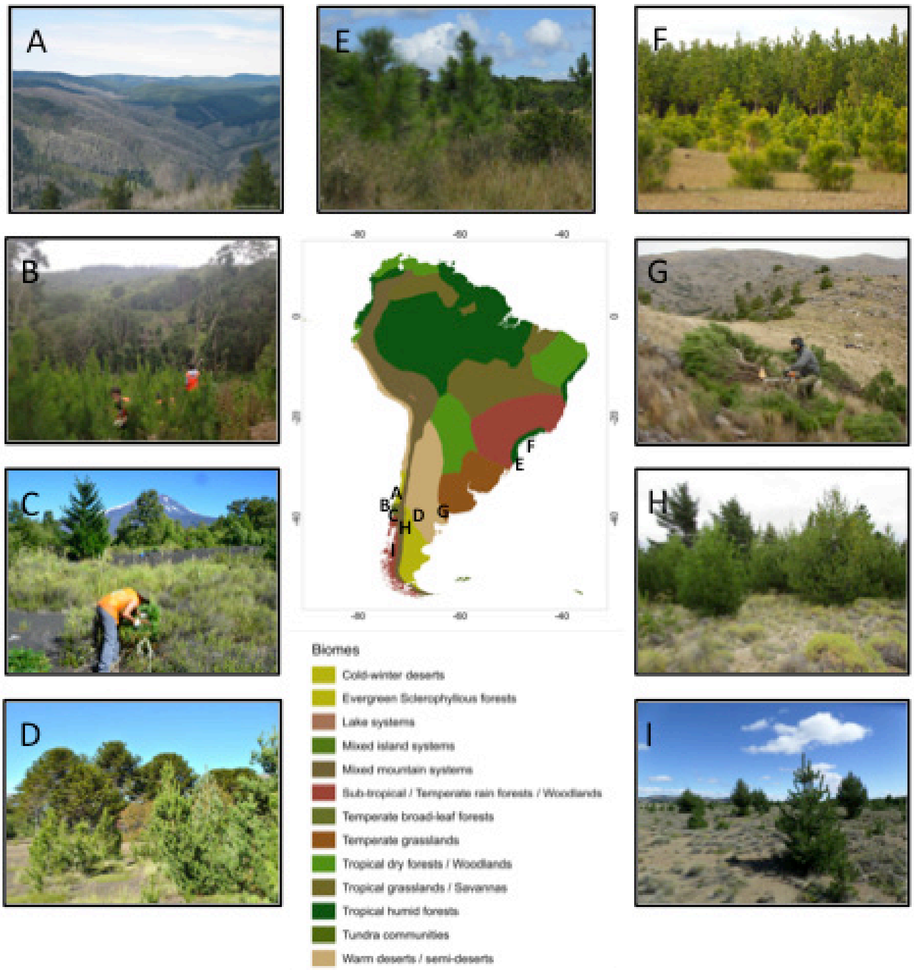

Fig. 14.1: Examples of pine invasions in South America. A) *Pinus radiata* in Maulino Forest. Cauquenes, Chile; B) University volunteers controlling *Pinus radiata* in Costal Forest. Hualpen, Chile; C) University volunteers controlling *Pseudotsuga menziesii* in Conguillio National Park, Chile; D) *Pinus contorta* in Malalcahuello National Reserve, Chile; E) and F) *Pinus taeda* in the Restinga ecosystem, Florianópolis, Brasil; G) Control of *Pinus halepensis* in Ernesto Tornquist Provincial Park. Buenos Aires, Argentina; H) *Pinus contorta* in Bariloche, Argentina; I) *Pinus contorta* in patagonian steppe. Coyhaique, Chile.

Here, we review the evidence of impacts, management, and the policy context of pine invasions in South America in order to advance in the reduction of this threat to biodiversity and ecosystem services. We will tackle these issues across biomes of South America: tropical and subtropical forests; tropical and subtropical grasslands, savannas and shrublands; Mediterranean forests and shrublands; temperate grasslands, savannas and shrublands; and temperate and sub-Antarctic forests. We advocate a more comprehensive approach to control pine invasions using prevention, early detection, containment and population management, restoration, and the inclusion of society in all steps of this process. We hope that drawing upon the experience of other countries with more advanced invasion scenarios may help to better manage the problem in South America before higher environmental, social and economical costs arise.

14.2 Pine Invasions and Their Impacts in South America

14.2.1 Tropical and Subtropical Forests

Pines were introduced to tropical and subtropical South America in the 19th century, but mostly at small scales and for horticulture. During the second half of the 20th century, governments started to import numerous pine species for silvicultural experimentation, and by the 1960s large-scale commercial pine plantations became common in many areas (Zenni & Ziller, 2011; Zenni & Simberloff, 2013). Because commercial plantations benefited from provenance trials, foresters tended to plant provenances and species in climates and soils to which they were well adapted (Zenni *et al.*, 2014). This resulted in positive genotype-by-environment interactions that promoted invasions. Currently, latitudinal patterns of pine invasion seem to match the latitudinal variation in pine native distributions. Not surprisingly, tropical pines are invasive in tropical regions, sub-tropical pines are invasive in sub-tropical regions, and temperate pines are invasive in temperate regions. Climate is a major driver limiting pine range distributions in both native and introduced ranges (e.g., Boulant *et al.*, 2009; Nuñez & Medley, 2011; McGregor *et al.*, 2012a). In the tropical Central Savannas of South America we currently see invasions by *Pinuso ocarpa* and *Pinus elliottii*, whereas in sub-tropical forests and grasslands we see invasions by *Pinus taeda* and also by *P. elliottii*. In sub-tropical forests *Pinus glabra* is also invasive, though only in one location as far we know, probably owing to the limited planting of the species (Zenni & Simberloff, 2013). We lack records of pine invasions in tropical forests (i.e., Amazon or Atlantic rainforests).

14.2.2 Tropical and Subtropical Grasslands, Savannas and Shrublands

The impacts of invasive pines in tropical and sub-tropical savannas have been studied more extensively than in other tropical and sub-tropical biomes in South America. In these savannas, *P. elliottii* densities can reach more than 12,000 plants per hectare in a period of 20 years and exclude non-woody plants from the native community. The result is a novel plant community highly dissimilar to non-invaded areas (Abreu & Durigan, 2011). However, in older pine stands, the re-establishment of native species previously excluded may occur, owing to the natural trimming process of the invasive population. Sub-tropical pines are shade-intolerant species and self-recruitment underneath dense pine invasions is limited, which results in an open canopy after pines reach full size, lowering competition for light and allowing native species to re-colonize the invaded area (Abreu *et al.*, 2011). In these cases, even though pine density in the invasive population decreases and native richness and abundance increases, the plant community remains highly dissimilar to non-invaded communities (Abreu *et al.*, 2011; Dostál *et al.*, 2013).

In tropical ecosystems, the impact caused by pine invasions on native plant communities is probably similar to the impacts observed in other biomes. The impacts are visible faster in these ecosystems, due to higher pine growth rates in warmer regions. In grassland relicts in the Atlantic rainforest, pine invasions decrease the overall richness and abundance of native grasses, forbs, and shrubs (Falleiros *et al.*, 2011). Researchers have characterized the spread of pines in South American sub-tropical forests (e.g., Emer & Fonseca, 2010, Zenni & Simberloff, 2013), but little is known about its impacts. The few studies that attempted to measure impact of pine invasions in tropical forests found that invasions tend to increase the depth of the litter layer and reduce recruitment of native species (Voltolini & Zanco, 2010).

Sub-tropical pines (e.g., *P. taeda* and *P. elliottii*) are also invasive in coastal dunes along the Atlantic coast of Brazil (Portz *et al.*, 2011; Zenni & Ziller, 2011). The invasions in dune habitats frequently form monocultures thick with pines that exclude native species and cause soil erosion. Also, most invasion reports are from coastal dunes in southern Brazil. Only preliminary assessments have been made on the status of pine invasions and their impacts on coastal dunes in lower latitudes, such as with the spontaneous spread of *P. pinaster* in southern Buenos Aires (Argentina) (Yezzi *et al.*, 2011, 2013; Cuevas & Zalba, 2011).

14.2.3 Mediterranean Forests and Shrublands

In the Mediterranean region of Chile, Spaniards and other Europeans first introduced non-native conifers in the 17th century, initially as ornamentals and later to control erosion and stabilize dunes (Peña *et al.*, 2008; Simberloff *et al.*, 2010). In the late 19th century, monospecific plantations were established where native forests had been destroyed (e.g. by mining, fires and grazing). *Pinus radiata* was introduced unintentionally in 1885 and planted with other North American conifers as an ornamental species (Lara & Veblen, 1993). Starting in the late 1960s, *P. radiata* was planted in large scale afforestation projects, heavily subsidized by the Chilean government since 1974, as the species was found to be especially suitable for the timber and pulp industries (Espinosa *et al.*, 1990; Le Maitre, 1998). At present, commercial plantations of *P. radiata* reach c.a. 1.5 million ha and are the basis for forest industry in Chile (Infor, 2013).

Pinus radiata from plantations are invading natural forests and xeric open shrublands, especially when disturbances such as harvesting and fire open the forest (Bustamante & Simonetti, 2004). There is evidence of pine invasion in open areas; however, the mechanism is not yet clear. The effect of the shading of native shrub *Lithrea caustica* on the seedling establishment of *P. radiata* is positive for recruitment and negative for seedling survival in semiarid ecosystems. This suggests that a common mechanism proposed to resist invasion in forest ecosystems such as shading is probably not sufficient to inhibit invasion in a semiarid region (Becerra & Bustamante, 2011). In coastal maulino forests, *P. radiata* is invading following deforestation and fragmentation (Bustamante & Castor, 1998; Bustamante *et al.*, 2003). In small isolated *Nothofagus* forests, surrounded by *P. radiata* plantations, the species invades protection zones, especially after disturbance that creates openings in the vegetation (Bustamante & Castor, 1998).

This invasion is relatively recent and has occurred rapidly. Bustamante and Simonetti (2005) indicated that the seeds of *P. radiata* are dispersed into the native forests, however seedling establishment occurs only at the edges and, therefore, they conclude that this exotic species is not invading less disturbed native forests. In less than a decade, it is possible to find a high proportion of young individuals and reproductive individuals growing inside fragments of native forest (Gomez *et al.*, 2011). Moreover, the pine density is negatively correlated with fragment size, while the proportion of reproductive pines is not similarly correlated (Gomez *et al.*, 2011). Due to the small size of the forest fragments present in this area along with current evidence, an increase is expected in the invasion of pine in the Mediterranean area. This situation could be reversed if active management of the invasion in native forest fragments and surrounding plantations is applied.

Pines, besides their negative impacts on biodiversity, may hold some potential for restoration of wildlife within plantations. *Pinus radiata* plantations might be a suitable habitat where the native tree *Cryptocaria alba* can regenerate (Guerrero & Bustamante 2007). Similarly, in degraded open sites where nurse plants are not available, *P. radiata* trees invading may facilitate the regeneration of native species, although studies show that facilitation produced by native trees is stronger than that produced by *P. radiata* (Becerra & Montenegro, 2013). Moreover, a well-developed understory in forestry plantations might serve as a surrogate habitat for native species and mitigate the negative effect of plantations on species richness (Simonetti *et al.*, 2013)

14.2.4 Temperate Grasslands and Savannas

Patagonia temperate grasslands or steppes in Chile and Argentina cover a much more extensive area compared with forests in Southern South America. They have also been the target of large afforestation plans, especially in Argentina. From 1970, pines have been planted mostly in these areas and nowadays, due to legal restrictions on plantations with non-native species in native forest areas, grasslands are the only ecosystem where companies and local owners can establish new pine plantations. Overall, two species were used in monospecific plantations: *Pinus ponderosa* and *Pinus contorta*. *Pinus contorta* has shown more aggressive invasive behavior, generating saplings up to 4 km from parent trees (Langdon *et al.*, 2010). When close to the source of seeds, *Pinus contorta* has invaded under different densities that depend on vegetation cover and land use, among other variables. In steppe areas where grazing has been removed, the invasion is more dense (Sarasola *et al.*, 2006). On the other hand, ponderosa pine, the species most planted, has not shown invasive potential, with saplings reaching only 50 meters from parent trees.

Extensive pine plantations have been established in areas that were formerly steppe or open woodland, where lack of fuel continuity was a major limitation to fire spread (Nuñez & Raffaele, 2007). Today, however, areas of these non-native conifers have burned and others create the potential for extensive crown fires in habitats previously characterized by surface fires and lower fuel volumes (Veblen *et al.*, 2003). Burned plantations interfere with post-fire succession to the original matorral vegetation (Nuñez & Raffaele, 2007) creating a positive feedback between fire and invasive pines. This effect of fires promoting pine expansion has also been documented for warmer pampas grasslands, where invasions by *Pinus halepensis* and *P. radiata* have been associated with fires (Zalba *et al.*, 2008). Introduced conifer plantations in Argentinean Patagonia have increased fire severity. For example, plantations of *Pinus ponderosa* and *P. contorta* have produced major conflagrations, initiated by lightning and human activity.

Replacement of native treeless vegetation with dense even-aged forests is by far the most striking impact of pine invasions in these ecosystems (Richardson *et al.*, 1994; Richardson & Higgins, 1998; Zalba & Villamil, 2002). Pine species may threaten to convert entire shrubland and grassland communities into conifer forests, with several native species in danger of at least local extirpation (Harding, 2001) even in high elevation environments (Pauchard *et al.*, 2009). In northwest Patagonia in Argentina, Lantschner *et al.*, (2008) found that the impact of pine plantations on bird communities depends on the landscape context and stand management practices. When plantations replaced steppes, the bird community was partially replaced by a new one, similar to that of ecotonal forests.

Pine invasion in Argentinean pampas reduced the diversity of native plants, displacing endemic species and promoting invasion by other non-natives (Zalba & Villamil, 2002; Cuevas & Zalba, 2009), and are also associated with changes in bird communities, including decline of obligate grassland birds and colonization by species that are less habitat-specific and colonize from forested regions (Zalba, 2000). Grazing has been found to promote plant invasion in this region (de Villalobos & Zalba, 2010, Loidy *et al.*, 2010; 2012), where cattle and feral horses reduce the biological resistance of native plant communities, creating windows of opportunity for the establishment of pine seedlings (de Villalobos *et al.*, 2011).

The light environment underneath pine plantations and invasions is critical to determine vegetation diversity. A reduction in plant diversity has been observed under young *Pinus contorta* trees (5-15 years) in the steppe of Coyhaique in the Chilean Patagonia (Pauchard *et al.*, unpublished data). Similar results were found in the steppes of Patagonia in Argentina, where dense and closed plantations with cover close to 90% see reductions of vegetation species richness from 31.7 to 10.4, and a cover reduction from 45.5% to less than 0.5% (Gyenge *et al.*, 2010). The reduction in diversity is positively correlated with pine size, and it can therefore be expected that older invasions have similar effects to those seen in plantations where almost no understory species are able to coexist with pines (Nuñez & Raffaele; 2007). Corley *et al.*, (2006; 2012) found that in dense plantation assemblages, richness of ants and beetles were modified and impoverished, with fewer ants in introduced pine plantations than in native steppe vegetation and fewer ant species in dense plantations. The impacts of pine invasions may be more intense that those associated with plantations, given the significantly higher density of stems in invaded areas (> 10 000 ind/ha) compared with the less dense plantations (ca. 1 000-1 600 ind/ha). Therefore, studies should address both the effects of plantations and invasion in these open environments.

14.2.5 Temperate and Sub-antarctic Forests

The temperate ecosystems of Chile and Argentina have been extensively planted with introduced Pinaceae species in recent decades, both for production purposes and to restore eroded and degraded land (Pauchard *et al.*, 2014). In southern Chile, the Chilean Forestry Institute (Instituto Forestal) developed a species introduction program in the early 1960s, establishing trial plots in more than 100 areas located between the semi-arid and temperate regions, including protected areas such as parks and reserves (Loewe & Murillo 2001; Pauchard *et al.*, 2014). Peña and Pauchard (2001) first warned that the introduced conifers of these trial plots were becoming invasive. *Pinus radiata* is not suitable for these harsher environmental conditions; therefore other species (*P. contorta*, *P. ponderosa*, *P. sylvestris*, and *Pseudotsuga menziesii*) are becoming widespread for plantation in more extreme, colder environments, accounting for ca. 50,000 ha. All species have been shown to be able to establish outside plantations, and *P. contorta* and *P. menziesii* are considered invasive in particular types of ecosystems (Peña *et al.*, 2007; Simberloff *et al.*, 2003). *Pinus contorta* and *P. menziesii* have become invasive in open and disturbed areas, as well as in natural vegetation, in the southernmost, colder environments (Peña *et al.*, 2008; Pauchard *et al.*, 2008). In Argentina, conifers were first introduced for forestry purposes to Isla Victoria, in the Patagonian Andes, in 1910, as an experimental project (Simberloff *et al.*, 2010). A *Pinus ponderosa* C. Lawson plantation initiated there in 1927 is one of the first conifer plantations in Latin America (Cozzo, 1987). Plantations on this island served as a government nursery for the rest of Argentina through the 1960s (references in Simberloff *et al.*, 2002), and plantation forestry in Patagonia increased rapidly in the 1970s (Schlichter & Laclau, 1998). Today, plantings of Pinaceae species are unevenly distributed across the Patagonian forest as a product of specific and localized efforts of afforestation. Those plantations include species such as *P. radiata, P. ponderosa, P. contorta, P. sylvestris, Larix decidua* and *Pseudotsuga menziesii*. Although several species are able to establish in native forests openings, *P. menziesii* is by far the most invasive conifer in forest dominated by *Austrocedrus chilensis and Nothofagus dombeyi* (Sarasola *et al.*, 2006).

Some factors seem to be slowing down the invasion of Pinaceae into forests, but these factors clearly do not provide a full barrier to the invasion (Simberloff *et al* 2010; Nuñez *et al.*, 2013). Among these factors, seed predation is playing an important role since pine seeds appear to be highly consumed and preferred by the native seed predator fauna (mainly birds and rodents) (Nuñez *et al.*, 2008; Caccia & Ballaré, 1998). Also, lack of compatible mycorrhizal fungi may be delaying the invasion in forests (Nuñez *et al.*, 2009). Although these factors may prevent invasion, other factors such as the presence of exotic mammals (deer, wild boar) can be promoting the invasion by preferring to consume their competitors — native tree species — instead of pines (Nuñez *et al.*, 2008; Relva *et al.*, 2010). In addition, exotic mammals have been shown to disperse mycorrhizal fungi needed for pine invasion, so they may be accelerating their invasion (Nuñez *et al.*, 2013). Therefore, even though some factors may be acting as a barriers to invasion in forests, there is ample evidence that pinaceae can invade in temperate and sub-Antarctic forests, producing large impacts.

Few studies have shown impacts of introduced conifers in temperate forests in South America, in either plantations or invaded areas. A decrease in plant diversity was recorded in *Araucaria araucana* forests invaded by *Pinus contorta* in the Andes of South-central Chile (Urrutia *et al.*, 2013). In forested ecosystems, invasive pines displace native species, but their effect seems to be lower than in the open temperate steppe (Pauchard *et al.*, unpublished data). Changes in fuel conditions and therefore in potential increase in fire frequency and intensity has been reported for these Araucaria forests. The main effect in young invasive stands is related to higher flammability and fuel continuity, although it is expected than in older stands the amount of fine fuel could also increase (Cóbar-Carranza *et al.*, 2014). Due to the lack of studies of the impacts of pine invasions in the biota of temperate forests, changes in animal and plant communities documented in conifer plantations can be extrapolated to invaded ecosystems. In small mixed plantations of *Pseudotsuga menziesii*, *Pinus radiata*, and *P. sylvestris* embedded in forests dominated by *Nothofagus dombeyi* in Northwest Patagonia, Paritsis and Aizen (2008) found decreasing species richness of understory vascular plants, epigeal beetles, and birds.

14.3 Current Policies and Management Actions on Pine Invasions

Biological invasions have only recently become part of the conservation agenda in South America. Unfortunately, many invasive species have been, and are still promoted as, important cultivars in forestry, farming and farm-fisheries (Nuñez & Pauchard, 2010). Pine plantations have been widely promoted across South America as an efficient and fast forestry cultivar, creating a strong industrial complex that uses pines as primary material for pulp, boards, and wooden furniture (Espinosa *et al.*, 1990). Government subsidies for pine plantations have been put in place in many countries, including Chile, Argentina, and Brazil. Native ecosystems and degraded areas have been widely afforested and no prevention actions have been implemented to control spread (Simberloff *et al.*, 2010).

Deficient forestry policies have increased the risk of pine invasions as much as the lack of data on invasive species has hindered control efforts. The first databases on invasive alien species in South America were online in 2003 in Argentina and in 2004 in Brazil (Box 14.1). The database structure was then disseminated to Chile, Uruguay, Paraguay, Bolivia, Peru, Ecuador, Colombia, Venezuela, and Surinam. This database was established between 2005 and 2011 by joint efforts of the Inter-American Biodiversity Information Network (IABIN) thematic network on invasive alien species (I3N), The Horus Institute for Environmental Conservation and Development in Brazil, the Universidad Nacional del Sur in Argentina, The Nature Conservancy South America Invasive Alien Species Program, and the Global Invasive Species Programme (GISP). The database led to the increased visibility of biological invasion issues at a continental scale, and to governmental concerns aligned with the guidelines and requests of the Convention on Biological Diversity.

Box 14.1. Alien Species Database and Conifer Invasion Management.

The I3N Network databases have been a substantial aid for research and a starting point for broader consultation upon building the species lists at the state or national levels. Once lists are in place, regulations are needed to define restrictions for the use of the species, and more development in legal terms, leading to changes in land use and general awareness. In Rio Grande do Sul, Brazil, where what is probably the most extensive pine (*P. taeda* and *P. elliottii*) invasion in Brazil can be seen along sand dunes and coastal grasslands, specific regulations for the use of pines have been proposed. The intention lies in reorganizing the landscape, maintaining plantations that are clearly limited and not allowed to expand over the surrounding landscape. The ornamental use of pines will be interrupted, as well as their use as shade trees or windbreaks. Production shall continue unharmed, but more responsibly while applying proper management practices to prevent and control invasions. These regulations are derived from Portaria SEMA RS 79/2013, the official list of invasive alien species published by the State Secretary of Environment.

Governments have taken considerable time to realize the importance of the threat posed by invasive species, while economic and political criteria have overridden environmental concerns. In the last few years, efforts to limit invasions have finally percolated into government and policy. Brazil, Colombia, and Uruguay developed and published official National Strategies for managing invasive alien species between 2009 and 2011, and Argentina is expected to complete one by the end of 2016. Although a rather large number of articles have been published in Latin America on biological invasions, the issue remains relatively under-studied in academic circles (Quiroz *et al.*, 2009). Pines are no exception, with most of the work already done focused on describing and understanding invasions, and less on how to control them.

Private efforts have, in some cases, moved faster than government initiatives when dealing with tree invasions. In the last decade, controlling biological invasions became a principle of the Forest Stewardship Council, first applied to pines that escape plantations, then gradually extended to other invasive species present in forest company properties. Although the principle is in place, complementary regulations are needed to make it functional. Official lists of invasive species constitute important references along with regulations that limit or even prohibit their use. The Colombian Ministry of Environment and Sustainable Development published a preliminary list of invasive species in 2011 (Ministerio de Ambiente y Desarrollo Sostenible, 2011), as well as a publication on risk analysis of introduced species in 2010 (Baptiste *et al.*, 2010). In Brazil, due to the lack of a national list, three official lists have been published at the state level in Paraná, Santa Catarina, and Rio Grande do Sul, while Rio de Janeiro and São Paulo are working on official lists at present. In Chile, FSC criteria and social awareness have pushed companies to recognize the invasive status of *Pinus radiata*. Companies are now required to control invasive pines outside plantations under these new forest certification schemes. Interestingly, forest companies are leading ambitious plans to control *P. radiata* in protection zones and forests of high conservation value.

The concern and awareness of the problem of biological invasions have grown in parallel with worldwide awareness and have been incorporated into the agenda of different agencies of national and provincial governments. This growth is the result of a maturation process over the last ten years, probably associated with pressure from environmental NGOs, as well as the increased availability of local and regional scientific information that gives certainty to the real dimension and relevance of the issue. Thus, in May 2013, a workshop funded by the national government of Argentina was held in Bariloche with the participation of international and local researchers, provincial and national decision makers, and forestry producers. Consensus was reached on the need to take action. Among the actions outlined is the inclusion of national regulations to subsidize new requirements for forestry plantations, such as measures to prevent invasions, monitoring, and control plans in future afforestation efforts, and exclusion of *Pinus contorta* among the species to be subsidized for plantations due to its high invasive potential. These regulations became effective in 2014 (MAGPNA, 2014).

In temperate and sub-antarctic forests, although substitution of native forest is no longer allowed, governments of both Chile and Argentina are still pushing for further pine plantations in eroded or deforested areas. Unfortunately, native forests take a relatively long time to recover after fire or other disturbances, especially if these are repeated over time. Thus, in an effort to reduce deforestation and increase provision of forest goods, governments have subsidized plantations in marginal conditions with limited economic results and negative environmental impacts. Forest companies, which in the past were also responsible for extensive plantations of invasive pines in these areas, are now recognizing that they have to reduce these negative impacts. Lately, these companies have not been planting some species, such as *Pinus contorta*, and have plans for the eradication of plantations of this species in Chilean Patagonia.

Efforts to control pines in South America are very limited, and pine control is only recently becoming a concern for forest companies and government agencies. In Argentinean Patagonia, the national parks administration is reducing the areas with plantations inside their domain and is controlling the spread of pines into natural areas (APN, 2000). Recently, several projects including pine invasion control and monitoring protocols in provincial forests are being conducted, funded by the national government. Their goal is to evaluate the impact of invasion and strategies to control their spread, with the ultimate goal of incorporating pine invasion control as a regular forestry practice. One incentive for these projects is the new national forest protection law that aims, among other goals, to control the presence of exotic species in national forests. South American temperate grasslands have been the setting for some of the first attempts at controlling invasive pines and restoring native communities. Mechanical control of *Pinus halepensis* and *P. radiata* has been conducted in Southern Argentinean Pampas since the early nineties, combined with studies on pine reproductive biology, seed longevity, and invasion spread, following an adaptive management strategy (Cuevas & Zalba, 2009; 2010; 2013). Considering the effects of fires on seed release from serotinous cones, and the interval between recruitment and seed set, fire has been identified both as a promoter of pine invasion but also as a potential tool for their control, providing that time between fires is less than that required for tree maturation (Cuevas & Zalba, 2010; Zalba *et al.*, 2009).

In Chile, no formal studies have been conducted to develop the best eradication techniques, but forest companies are under pressure to eradicate invasive pines in protection zones due to forest certification regulations (Pauchard *et al.*, 2014). The impact of pine invasions is slowly being considered both by private and public stakeholders, and there are currently plans to develop techniques to control them at local scales. However, it is not yet clear if large-scale management will be implemented or if specific regulations will enforce the control of pine invasions.

14.4 The Future: How to Reduce Impacts?

In technical terms, at the stand scale, pine invasions may be among the least complicated to control, especially as adult trees do not re-sprout once cut down. However, in landscapes dominated by pine plantations, propagule pressure is so high that re-invasion may occur and ecosystem processes that trigger invasion may already be changed (e.g. fire regimes). Also, because they are dispersed by wind, pines can reach places with limited accessibility in mountains and protection zones, where they are not detected until their canopies have surpassed the native vegetation.

Given the relative predictability in terms of the species and the areas than can be invaded, and how easy it is to detect and eliminate them, people managing invasive pines may have an advantage compared to other more subtle or resistant invasions (e.g. *Acacia* spp.). Nonetheless, there are several shortcomings, especially as pines are seen as an important economic resource, and therefore, cultural and economic limitations usually complicate the implementation of prevention and control strategies. Conflicts abound in governmental agencies with contrasting visions and objectives, as well as on private properties where owners wish to take advantage of invasions without taking responsibility for them, and thus do not undertake any proper forest management. The main challenges are in communication and coordination, and clear regulations are strongly needed.

New regulations and research in some key areas (e.g. restoration) are crucial in helping to prevent the impacts of these invasions throughout South America. Invasions are widespread on the continent, but appropriate management can prevent future invasions and control current ones, at least in properties associated with forestry production. A clear understanding of how biotic, abiotic, and anthropogenic factors affect pine invasion is needed for effective control (Figure 14.2). There is no universal "recipe" for pine invasion management, as the factors determining the invasion outcome vary with the species, the site, and the silvicultural and control strategies. Thus, only general guidelines of the best management practices can be addressed in this text.

Controlling pines is not only good for the environment but also for the forestry industry, given that invasions hamper future silvicultural practices, are sources of diseases, increase fire frequency, and affect the sustainability of the forestry business with negative marketing repercussions. Here, we address some of the key stages of pine invasion management:

1) Prevention and risk analysis: Despite the fact that the biological characteristics of species have not been enough to precisely predict invasion behavior by alien species, risk analysis can provide valuable inferences on invasion probability and its consequences. This information can be used in decision-making and to define management strategies and public policies (National Research Council, 2002). Risk does not only involve species biology, but also environmental damage, social and economic impacts, and public health issues, as well as the feasibility of control or eradication. Risk is the product of the likelihood of an event or process and its consequences (National Research Council, 2002). Risk analysis techniques are now an important tool to prevent plant invasions and may be used for screening new pine introductions (see Box 14.2).

Fig. 14.2: Site-specific factors determining the outcome of a pine invasion modified from Pauchard *et al*. (2014). The interaction between species traits and the invaded community determines the expected invasion risk. However, silvicultural techniques and landscape planning may reduce or increase this risk, affecting propagule pressure. For example, the use of other species wind buffers to avoid seed dispersal or the establishment of plantations in safe-sites of the landscape may reduce the risk of invasion. Control actions should target all stages of the invasion and should be coordinated with the silvicultural schemes at the stand and lanscape scales. Policies and regulations set the stage for both silvicultural and control practices and therefore are crucial to reduce pine invasions.

Box 14.2. Assessing the Risk of New Pine Introductions.

Risk assessment protocols can be used for different and complementary purposes. Originally conceived to assess the risk of introducing new species into a country, they can be applied to help define management priorities for several species in a given area according to the level of risk, select species in a commercial genus that will pose less problems in terms of invasion, provide scientific evidence of invasive capacity in the absence of field data or research, and provide impartial assessments of species to be included in official invasive species lists. Pines may also be screened using risk assessment tools (McGregor *et al.*, 2012b).

One of the most relevant variables of weed risk assessment is climate. Climate is a good predictor of conifer naturalization and invasion both at the genotype and at the species levels (Nuñez & Medley, 2011; Zenni *et al.*, unpublished). In general, climatic information from native and other invasive ranges of pines can be used to assess the risk of naturalization in another region. Climate, combined with knowledge from forestry trials (Zenni *et al.*, 2014) and intense propagule pressure, explains why commercial forestry tends to be the most frequent source of pine invasions (Essl *et al.*, 2010).

The Universidad Nacional del Sur in Argentina and the Horus Institute in Brazil designed new risk analysis protocols and adjusted existing ones to fit South American conditions through the I3N - IABIN Invasive Species Network. Results of assessments are available for 149 plants at http://www.institutohorus.org.br/index.php?modulo=inf_analise_risco. This platform can be used to evaluate new pine introductions in South America, including analyses of species pathways within the South American continent.

It is unlikely that many new conifer species will be introduced in South America in the coming years, given that a large proportion of species have already been introduced to the continent. However, introduction programs are still in place for forestry and ornamental uses. Therefore, pre-introduction screening tools (i.e., Pheloung *et al.*, 1999) should be used before allowing new introductions into South America. Predictive models that consider species mean trait values (e.g., Rejmánek & Richardson, 1996; Grotkopp *et al.*, 2002) tend to work only at larger spatial scales (Zenni & Simberloff, 2013). Instead, models of current and potential spread at the genotype level that explicitly incorporate climate, soil, belowground interactions, and commercial plantations can be highly informative. For instance, recent work has shown that genotype-environment interactions are a major driver of *Pinus taeda* invasions (Zenni *et al.*, in review). In this study, the authors found that genetic constraints limit the ability of provenances to invade in unfavorable introduced habitats. The invasive potential of provenances was largely predictable by isoclines of temperature and precipitation. The adaptive mechanism was strong enough to overcome important differences in propagule pressure.

2) Early detection: Plans to start forestry practices need to be associated with plans to control the spread of trees. Decades of pine invasion research have shed light on the process of invasion. There is strong scientific evidence available in a number of areas that can be readily used for prevention and control of invasions. There is information on the key characteristics of invaded ecosystems that make them more prone to invasion, and the characteristics of species that make them more capable of invading. For example, there is ample evidence that some areas are more invasible than others (e.g. open disturbed areas are easily invaded, while close mature forests are less so), and some species are more invasive than others (species with smaller seeds invade more than species with large seeds (Rejmanek & Richardson, 1996)). The identification of the ecological events that open opportunities for pine establishment and invasion, like fires, grazing by large ungulates, and topsoil removal, can help to prevent or lead to early detection and eradication of new foci of invasion. This information can be relatively easy to use in legislation to guide tree plantations into areas with low risk of invasion, as well as to direct the introduction of species with low probability of invasion. This may not solve the invasion problems completely, but would make it easier or cheaper to manage. Species that are notably problematic (e.g. *Pinus contorta*, *P. radiata*, *P. elliottii*) and areas more prone to invasion or sensitive zones, such as natural reserves, need to be excluded or carefully monitored.

3) Containment and population control: Regarding post-invasion management, plans have to be in place to control the spread even when all precautions selecting sites and proper species have been taken (Figure 14.2). An adaptive management strategy is usually the best way to reduce uncertainty about the best control options, and also for understanding the causes behind the invasion (Zalba, 2010; Zalba & Ziller, 2007). Pines tend to colonize new areas, even in their native range, so expecting them not to do so is unrealistic. Information on the age of cone production (i.e. the start of seed production) is key for controlling the invasion. It is important to implement control measures soon after the trees reach reproductive age and not wait long enough for invasive trees to start producing cones themselves. Knowing the dynamics of the seeds, both in the cones and in the soil bank, is key information for effective pine control. Visiting the sites every few years (depending on the area and the age of cone production) to remove all the trees colonizing plantation surroundings is critical.

When invasions reach large scales and dominate the landscape, restoration becomes essential. Otherwise, if no active restoration is conducted, new invasions by pines or by other invasive plants (with similar or higher impacts) are highly likely. In South America, we can potentially avoid reaching the levels of invasion found in other countries such as New Zealand or South Africa, but if such levels are reached, active restoration plans must be implemented. Given the cost of restoration efforts, it is highly beneficial to avoid the stage were restoration becomes a requirement.

In order to avoid invasion of areas outside plantations, along roads, and throughout the landscape, regulations need to be in place restricting the use of pines to forestry, wood, pulp and paper, and other forest products. Other common uses, such as shade, wind-breaks, or ornamental purposes, need to be banned. Plantation owners have to be made responsible for control work and forest company associations need to engage in cleaning up current invasions, taking responsibility for the wide spread of the species.

Roadsides must be prioritized and kept clean of pines. Forestry stands must be clearly visible and pines must be contained within them. Those interested in producing pines must have projects approved by environmental agencies, even if there is a simplified process for small farmers, who also have to commit to controlling spread beyond plantations. Pine seedlings must no longer be distributed by public institutions or events, except for forestry production programs with clear rules established to avoid invasions. Pine species that has shown invasive behavior, in South America or elsewhere, should be listed as invasive species so that all broader legislation referring to environmental impacts can also refer to side the effects of badly managed plantations.

4) Restoration: Pines may be relatively easy to control locally, but international experience shows that restoration is needed for achieving a true recovery of the native vegetation after prolonged invasions. New Zealand has conducted massive efforts to control pine invasions in temperate grasslands using herbicide and mechanical control, but such efforts do not necessarily imply a recovery of the natural ecosystem. Other invasive species such as shrubs and herbs may become dominant as the pine is controlled. Active restoration may be required in cases where pines have decimated the original native plant populations and where few or no propagules of the native species are present. Changes in soil chemistry, litter depth, and soil biota may hamper the recovery of native vegetation. Ad hoc restoration plans should be developed in conjunction with control measures. In some cases, passive restoration may be sufficient to restore natural vegetation, for example around isolated pine trees in protection zones. In other cases, where massive stands of pines have displaced native vegetation, active restoration needs to be planned before controlling the pine. It may even be recommended that restoration and control are gradually implemented to facilitate the establishment of the native vegetation under the protection posed by the invasive pines. Control plans should always be aimed at restoring ecosystem function, structure, and the composition of native plant communities. On the other hand, the use of pine species for restoration purposes needs to be carefully reviewed and regulated (Zalba, 2013).

5) Public awareness and regulatory frameworks: Great effort should be aimed at increasing awareness of the importance of preserving native biodiversity and the threats posed by the expansion of invasive alien species, including pines. Education actions have to be directed not just to the public in general, but also to specific actors like governmental agencies, international aid and development programmes, and professionals in the field of natural resources. For pines, the positive services that they provide when growing in plantations should be recognized, and a clear distinction should be made with those pines growing outside plantations and invading natural communities. This distinction may be difficult to comprehend for the general public, but it is crucial to avoid useless generalizations and confrontations between private and public stakeholders. Basic regulations should be approved in all countries of South America indicating that pine plantations owners should take all necessary actions to avoid the negative externality of pine invasions. Forest certification schemes have already advanced towards that goal, thus it should not be difficult to adopt those regulations into national laws.

14.5 Concluding Remarks

Pine invasions are occurring in South America across multiple biomes and landscapes. However, there is a delay to these invasions when compared to similar regions in Africa, Australia and New Zealand, because of the more recent expansion of forest afforestation. While the impacts of pine invasions vary according to the ecosystem being invaded, it is clear that they pose a risk to local and regional biodiversity by affecting ecosystem composition, structure, and function. Changes in fire regimes may be one of the most striking elements of pine invasions. Policy and legal frameworks in South America have favored pine plantations with no consideration of the negative externality caused by their invasions. Still, governments subsidize the planting of extremely invasive pines. Actions in the future should consider: 1) prevention of new introductions; 2) early detection of invasions; 3) containment and control; 4) restoration; and 5) strengthening of regulations and public awareness. South America is still on time to take the necessary actions to reduce pine invasions, and it should learn from the experience of other regions where pine invasions are now causing major ecological and economic costs.

14.6 Acknowledgements

Aníbal Pauchard and Rafael García were funded by Fondecyt 1140485, ICM P05-002 and PFB-23. Martín Nuñez was funded by NSF 0948930 and BIO Nº5. Sergio Zalba received support from the Universidad Nacional del Sur and CONICET (Consejo Nacional de Investigaciones Científicas y Técnicas).

In a nutshell

- Pines are among the best-studied groups of invasive species. In South America pines have been planted for forestry, and in many areas are becoming highly invasive.
- The impacts of pine invasions can be high and vary according to region, posing a risk to local and regional biodiversity by affecting ecosystem composition, structure, and function. Still, governments subsidize the planting of extremely invasive pines.
- It is clear that governments and stakeholders should take immediate action to prevent this problem, which is currently at a stage where its control is doable.
- The experience of other regions such Africa, Australia, and New Zealand should help South America to prevent and manage pine invasions to avoid further environmental and economic costs.

14.7 Bibliography

Abreu, R. C. R., Assis, G. B., Frison, S., *et al.*, (2011). Can native vegetation recover after slash pine cultivation in the Brazilian Savanna? Forest Ecology and Management, 262, 1452-1459.
Abreu, R. C. R., Durigan, G. (2011). Changes in the plant community of a Brazilian grassland savannah after 22 years of invasion by *Pinus elliottii* Engelm. Plant Ecology and Diversity, 4, 269-278.
APN (2000). Proyecto de manejo silvicultural para la paulatina erradicacion de especies forestales exoticas de la Isla Victoria. Administración de Parques Nacionales de la República Argentina.
Baptiste, M. P., Castaño, N., Cárdenas, D., *et al.*, (Eds.) (2010). *Análisis de riesgo y propuesta de categorización de especies introducidas para Colombia*. Bogotá, D. C., Colombia: Instituto de Investigación de Recursos Biológicos Alexander von Humboldt.
Becerra, P. I., Bustamante, R. O. (2011). Effect of a native tree on seedling establishment of two exotic invasive species in a semiarid ecosystem. Biological Invasions, 13, (12) 2763-2773.
Becerra, P. I., Montenegro, G. (2013). The widely invasive tree *Pinus radiata* facilities regeneration of native woody species in a semi-arid ecosystem. Applied Vegetation Science, 16, 173-183.
Boulant, N., Garnier, A., Curt, T., *et al.*, (2009). Disentangling the effects of land use, shrub cover and climate on the invasion speed of native and introduced pines in grasslands. Diversity and Distributions, 15, 1047-1059.
Buckley, Y.M., Brockerhoff, E., Langer, L., *et al.*, (2005). Slowing down a pine invasion despite uncertainty in demography and dispersal. Journal of Applied Ecology, 42, 1020-1030.
Bustamante, R. O., Castor, C. (1998). The decline of an endangered temperate ecosystem: the ruil (*Nothofagus alessandrii*) forest in central Chile. Biodiversity and Conservation, 7, 1607–1626.
Bustamante, R. O., Simonetti, J. A. (2005). Is *Pinus radiata* invading the native vegetation in central Chile? Demographic responses in a fragmented forest. Biological Invasions, 7, 243–249.
Bustamante, R. O., Serey, I. A., Pickett, S. T. A. (2003). Forest fragmentation, plant regeneration and invasion processes across edges in central Chile. In G. A. Bradshaw & P. A. Marquet (Eds), How Landscapes Change. Ecological Studies 162 (pp.145–60). Berlin: Springer.
Caccia, F. D., Ballaré, C. L. (1998). Effects of tree cover, understory vegetation, and litter on regeneration of Douglas-fir (*Pseudotsuga menziesii*) in southwestern Argentina. Canadian Journal of Forest Research, 28, 683-692.
Cóbar-Carranza, A., García, R. A., Pauchard, A., *et al.*, (2014). Effect of *Pinus contorta* invasion on forest fuel properties and its potential implications on the fire regime of *Araucaria araucana* and *Nothofagus antarctica* forests. Biological Invasions, 16, 2273-2291.
Corley, J. C., Sackmann, P., Rusch, V. *et al.* (2006). Effects of pine tree silviculture on the ant assemblages (Hymenoptera: Formicidae) of the Patagonian steppe. Forest Ecology and Management, 122, 162-166.
Corley, J. C., Villacide, J.M., Vesterinen, M. (2012). Can forest management lessen the impact of pine plantations on beetle and ant diversity in the Patagonian steppe? Southern Forests, 74 (3), 195–202.
Cozzo, D. (1987). Evolución de la silvicultura de plantaciones forestales en Argentina. La postulación de mayores espacios de mejoramiento silvicultura. Simposio sobre Silvicultura y Mejoramiento Genético de Especies Forestales 1, 81–95. Buenos Aires: Centro de Investigaciones y Experiencias Forestales.
Cuevas, Y. A., Zalba, S. M. (2009). Control de pinos invasores en el Parque Provincial Ernesto Tornquist (Buenos Aires): áreas prioritarias y análisis de costos. BioScriba, 2, (2) 76-89.
Cuevas, Y. A., Zalba, S. M. (2010). Recovery of native grasslands after removing invasive pines. Restoration Ecology, 18, (5) 711-719.
Cuevas, Y. A., Zalba, S. M. (2011). Forestaciones de pino y ambientes de dunas costeras: rol de la depredación de semillas. Boletín de la Sociedad Argentina de Botánica, 46, (Supl.) 68.

Cuevas, Y. A., Zalba, S. M. (2013). Efecto del tipo de corte y de tratamientos en el mantillo para la restauración de pastizales naturales invadidos por Pinus halepensis. Boletín de la Sociedad Argentina de Botánica, 48 (2), 315-329.

de Villalobos, A. E., Zalba, S. M. (2010). Continuous feral horses grazing and grazing exclusion in mountain pampean grasslands in Argentina. Acta Oecologica, 36, 514-519.

de Villalobos, A. E., Zalba, S. M., Peláez, D. (2011). Pinus halepensis invasion in mountain pampean grassland: effects of feral horses grazing on seedling establishment. Environmental Research, 111, 953-959.

Dostál, P., Müllerová, J., Pyšek, P., *et al*. (2013). The impact of an invasive plant changes over time. Ecology Letters, 16 (10), 1277-1284.

Drake, J. A., Mooney, H. A., di Castri, F., *et al*. (1989). Biological invasions: a global perspective. Chichester, UK: John Wiley & Sons.

Emer, C., Fonseca, C. R. (2010). Araucaria forest conservation: mechanisms providing resistance to invasion by exotic timber trees. Biological Invasions, 13, 189-202.

Espinosa, M., Eso, R., Drake, F. (1990). Silvicultura de las plantaciones forestales en Chile: pasado, presente y futuro. Agro-Ciencia, 6, 131–44.

Essl, F., Mang, T., Dullinger, S., *et al*. (2011). Macroecological drivers of alien conifer naturalizations worldwide. Ecography, 34, 1076-1084.

Essl, F., Moser, D., Dullinger, S., *et al*. (2010). Selection for commercial forestry determines global patterns of alien conifer invasions. Diversity and Distributions, 16, 911-921.

Falleiros, R. M., Zenni, R. D., Ziller, S. R. (2011). Invasão e manejo de Pinus taeda em campos de altitude do Parque Estadual o Pico Paraná, Paraná, Brasil. Revista Floresta, 41, 123-134.

Fonseca, C., Guadagnin, D. L., Emer, C., *et al*. (2013). Invasive alien plants in the Pampas grasslands: a tri-national cooperation challenge. Biological Invasions, 15 (8), 1751-1763.

Gómez, P., Bustamante, R., San Martin, J., *et al*. (2011). Population structure of Pinus radiata D.Don in fragments of Maulino Forest in Central Chile. Gayana Botanica, 68 (1), 97-101.

Grotkopp, E., Rejmánek, M., Rost, T.L. (2002). Toward a causal explanation of plant invasiveness: seedling growth and life-history strategies of 29 pine (Pinus) species. The American Naturalist, 159, 396-419.

Guadagnin, D. L., Zalba, S.M., Costa Gorriz, B., *et al*. (2009). Árvores e arbustos exóticos invasores no Bioma Pampa - questões ecológicas, culturais e socioeconômicas de um desafio crescente. In V. De Patta Pillar, S. C. Müller, Z. M. de Souza, A. V. Ávila Jacques (Eds.), Campos Sulinos: conservação e uso sustentable da biodiversidade (pp.300-316). Brasilia D.F.: Ministerio do Meio Ambiente.

Guerrero, P. C., Bustamante, R. O. (2007). Can native tree species regenerate in Pinus radiata plantations in Chile? Evidence from field and laboratory experiments. Forest Ecology and Management, 253, 97-102.

Gundale, M. J., Pauchard, A., Langdon, B., *et al*. (2014). Can model species be used to advance the field of invasion ecology? Biological Invasions, 16 (3), 591-607.

Gyenge, J. E., Fernández, M. E., Rusch, V., *et al*. (2010). Towards Sustainable Forestry Development in Patagonia: Truths and Myths of Environmental Impacts of Plantations with Fast-Growing Conifers. The Americas Journal of Plant Science and Biotechnology, 3, 9-22.

Harding, M. (2001). South Island Wilding Conifer Strategy. Christchurch, NZ: Department of Conservation.

Higgins, S. I., Richardson, D. M. (1998). Pine invasions in the southern hemisphere: modelling interactions between organism, environment and disturbance. Plant Ecology, 135, 79-93.

INFOR (2013). El Sector Forestal Chileno 2013. Instituto Forestal (INFOR), Ministerio de Agricultura. Chile. http://wef.infor.cl/sector_forestal/sectorforestal.php#/0 Accessed: March 2014.

Jobbagy, E. G., Jackson, R.B. (2007). Groundwater and soil chemical changes under phreatophytic tree plantations. Journal of Geophysical Research, 112, 1-15.

Langdon, B., Pauchard, A., Aguayo, M. 2010. *Pinus contorta* invasion in the Chilean Patagonia: local patterns in a global context. Biological Invasions, 12, 3961-3971.

Lantschner, M.V., Rusch, V., Peyrou, C. (2008). Bird Assemblages in Pine Plantations Replacing Native Ecosystems of N.W. Patagonia, Argentina. Biodiversity and Conservation, 17 (5), 969-989.

Lara, A., Veblen, T. T. (1993). Forest plantations in Chile: a successful model? In A. Mather (Ed.), Afforestation policies, planning and progress (pp.118-139). London, U.K: Belhaven Press.

Le Maitre D. C. (1998). Pines in cultivation: a global view. In D. M. Richardson (Ed.), Ecology and Biogeography of *Pinus* (pp.407–31). Cambridge: Cambridge University Press.

Ledgard, N. (2001). The spread of lodgepole pine (*Pinus contorta*, Dougl.) in New Zealand. Forest Ecology and Management, 141, 43–57.

Little, C., Lara, A., McPhee, J., *et al.* (2009). Revealing the impact of forest exotic plantations on water yield in large scale watersheds in South-Central Chile. Journal of Hydrology, 374, 162-170.

Loewe, V., & Murillo, P. (2001). Estudio de ensayos de introducción de especies. Santiago, Chile: Instituto Forestal (INFOR).

Loidy, A., Distel, R. A., Zalba, S. M. (2010). Large herbivore grazing and non-native plant invasions in montane grasslands of central Argentina. Natural Areas Journal, 30, (2), 148-155.

Loidy, A., Zalba, S.M., Distel, R. A. (2012). Viable seed banks under grazing and exclosure conditions in montane mesic grasslands of Argentina. Acta Oecologica, 43, 8-15.

Lusk, C. (2008). Constraints on the evolution and geographical range of Pinus. New Phytologist, 178 (1), 1-3.

MAGPNA (2014). Ministerio de Agricultura Ganadería y Pesca de la Nación Argentina. Nota 485. Dirección General de Forestación.

McGregor, K. F., Watt, M. S., Hulme, P. E., *et al.* (2012a). What determines pine naturalization: species traits, climate suitability or forestry use? Diversity and Distributions, 18, 1013-1023.

McGregor, K. F., Watt, M. S., Hulme, P. E., *et al.* (2012b). How robust is the Australian Weed Risk Assessment protocol? A test using pine invasions in the Northern and Southern hemispheres. Biological Invasions, 14, 987-998.

Ministerio de Ambiente y Desarrollo Sostenible de Colombia (2011). Plan Nacional para la Prevención, el Control y Manejo de las Especies Introducidas, Trasplantadas e Invasoras: Diagnóstico y listado preliminar de especies introducidas, Trasplantadas e invasoras en Colombia. Bogotá, D.C., Colombia: Instituto de Investigación de Recursos Biológicos Alexander von Humboldt, The Nature Conservancy –Colombia, Ministerio de Ambiente y Desarrollo Sostenible.

National Research Council (2002). Predicting invasions of non-indigenous plants and plant pests. Washington D. C., USA: National Academy Press.

Nuñez, M. A., Hayward, J., Horton, T. R., *et al.* (2013). Exotic mammals disperse exotic fungi that promote invasion by exotic trees. PLoS ONE, 8 (6), e66832.

Nuñez, M. A., Horton, T. R., Simberloff, D. (2009). Lack of belowground mutualisms hinders Pinaceae invasions. Ecology, 90, (9), 2352-2359.

Nuñez, M. A., Medley, K. A. (2011). Pine invasions: climate predicts invasion success; something else predicts failure. Diversity and Distributions, 17, 703-713.

Nuñez, M. A., Pauchard, A. (2010). Biological invasions in developing and developed countries: does one model fit all? Biological Invasions, 12 (4), 707-714.

Nuñez, M. A., Raffaele, E. (2007). Afforestation causes changes in post-fire regeneration in native shrubland communities of northwestern Patagonia, Argentina. Journal of Vegetation Science, 18 (6), 827:834.

Nuñez, M. A., Simberloff, D., Relva, M. A. (2008). Seed predation as a barrier to alien conifer invasions. Biological Invasions, 10, 1389-1398.

Paritsis, J. Aizen, M. A. (2008). Effects of exotic conifer plantations on the biodiversity of understory plants, epigeal beetles and birds in Nothofagus dombeyi forests. Forest Ecology and Management, 255, 1575-1583.

Pauchard, A., Kueffer, C., Dietz, H., *et al.* (2009). Ain't no mountain high enough: Plant invasions reaching high elevations. Frontiers in Ecology and the Environment, 7, 479-486.

Pauchard, A., Langdon, B., Jiménez, A., *et al.* (2014). Pináceas invasoras en el sur de Sudamérica: patrones, mecanismos e impactos potenciales. In F. Jaksic, S. Castro (Eds.), Ecología de Invasiones Biológicas (pp.283-308). Santiago: Editorial Universidad Católica.

Pauchard, A., Langdon, B., Peña, E. (2008). Potencial invasivo de *Pseudotsuga menziesii* (Mirb.) Franco en Bosques Nativos del Centro-Sur de Chile: patrones y recomendaciones. In: R. Mujica, H. Grosse, B. Muller-Using (Eds.) Bosques Seminaturales: una opción para la rehabilitación de bosques nativos degradados (pp.89-114). Santiago, Chile: Instituto Forestal.

Pauchard A., Nuñez, M., Raffaele, E., Bustamante, R., Legard, N., Relva, M., Simberloff, D. (2010). Introduced conifer invasions in South America: an update. Frontiers of Biogeography, 2 (2), 34-36.

Pheloung, P. C., Williams, P. A., Hallo, S. R. (1999). A weed risk assessment model for use as a biosecurity tool evaluating plant introductions. Journal of Environmental Management, 57, 239-251.

Peña, E., Hidalgo, M., Langdon, B., *et al.*, (2008). Patterns of spread of Pinus contorta Dougl. ex Loud. invasion in a Natural Reserve in southern South America. Forest Ecology and Management, 256, 1049-1054.

Peña, E., Langdon, B., Pauchard, A. (2007). Árboles exóticos naturalizados en el bosque nativo chileno. Bosque Nativo, 40, 3-7.

Peña, E., Pauchard, A. (2001). Coníferas introducidas en áreas protegidas: un riesgo para la biodiversidad. Bosque Nativo, 30, 3-7.

Portz, L., Manzolli, R. P., Saldanha, D. L., *et al.* (2011). Dispersão de espécie exótica no Parque Nacional da Lagoa do Peixe e seu entorno. Revista Brasileira de Geografia Física, 4, 33-44.

Quiroz, C., Pauchard, A., Cavieres, L. A., *et al.* (2009). Análisis cuantitativo de la investigación en invasiones biológicas en Chile: tendencias y desafíos. Revista Chilena de Historia Natural, 82, 497-505.

Rejmánek, M., Richardson, D. M. (1996). What attributes make some plant species more invasive? Ecology, 77, 1655-1661.

Relva, M. A., Nuñez, M. A., Simberloff, D. (2010). Introduced deer reduce native plant cover and facilitate invasion of non-native tree species: evidence for invasional meltdown. Biological Invasions, 12, (2), 303-311.

Richardson, D.M. (2006). Pinus: a model group for unlocking the secrets of alien plant invasions? Preslia, 78, 375–388.

Richardson, D., Higgins, S. (1998). Pines as invaders in the southern hemisphere. In D. Richardson (Ed.), Ecology and biogeography of Pinus (pp.450–473). Cambridge, UK: Cambridge University Press.

Richardson, D., Williams, P., Hobbs, R. (1994). Pine invasions in the southern hemisphere: determinants of spread and invadability. Journal of Biogeography, 21, 511-527.

Richardson, D. M., Van Wilgen, B. W., Nuñez, M. A. (2008). Alien conifer invasions in South America: short fuse burning? Biological Invasions, 10, 573-577.

Sarasola, M., Rusch, V., Schlichter, T., *et al.* (2006). Invasión de coníferas forestales en áreas de estepa y bosques de ciprés de la cordillera en la Región Andino Patagónica. Ecologia Austral, 16, 143–56.

Schlichter, T., Laclau, P. (1998). Ecotono estepa-bosque y plantaciones forestales en la Patagonia norte. Ecologia Austral, 8, 285–96.

Simberloff, D., Nuñez, M. A., Ledgard, N. J., *et al.* (2010). Spread and impact of introduced conifers in South America: lessons from other southern hemisphere regions. Austral Ecology, 35, 489-504.
Simberloff, D., Relva, A., Nuñez, M. (2002). Gringos en el bosque: introduced tree invasion in a native nothofagus/austrocedrus forest. Biological Invasions, 4, 35-53.
Simberloff, D., Relva M. A., Nuñez M. A. (2003). Introduced Species and Management of a *Nothofagus/Austrocedrus* Forest. Environmental Management, 31 (2), 263- 275.
Simonetti, J. A., Grez, A. A. Estades, C. F. (2013). Providing Habitat for Native Mammals through Understory Enhancement in Forestry Plantations. Conservation Biology, 27 (5), 1117-1121.
Urrutia, J., Pauchard, A., García, R. A. (2013). Diferencias en la composición vegetal de un bosque de *Araucaria araucana* (Molina) K.Koch y *Nothofagus antarctica* (G. Forst.) Oerst. asociadas a un gradiente de invasión de *Pinus contorta* Douglas ex Loudon. Gayana Botanica, 70 (1), 92-100.
Veblen, T. T., Kitzberger, T., Raffaele, E., *et al.* (2003). Fire history and vegetation changes in northern Patagonia, Argentina. In T. T. Veblen, W. L. Baker, G. Montenegro, *et al.* (Eds.), Fire and Climatic Change in Temperate Ecosystems of the Western Americas (pp.265–296). New York: Springer.
Visser, V., Langdon, B., Pauchard, A. *et al.* (2014). Unlocking the potential of Google Earth as a tool in invasion science. Biological Invasions, 16 (3), 513-534.
Voltolini, J. C., Zanco, L. (2010). Densidade de plântulas e jovens de espécies nativas de Floresta Atlântica em áreas com e sem o pinheiro americano (*Pinus elliottii*). Revista Biociências, 16, 102-108.
Yezzi, A., Nebbia, A. J., Zalba, S. M. (2011). Efectos de fragmentación en dunas generada por una forestación implantada. Boletín de la Sociedad Argentina de Botánica, 46, (Supl.) 79.
Yezzi, A., Nebbia, A. J., Zalba, S. M. (2013). Efectos de una plantación forestal sobre la composición de la vegetación nativa en pastizales costeros bonaerenses. Boletín de la Sociedad Argentina de Botánica, 48, (Supl.) 83.
Zalba, S. M. (2000). El pastizal pampeano, los árboles exóticos y la fauna silvestre: un problema con múltiples dimensiones. In C. Bertonatti, J. Corcuera (Eds.), Situación Ambiental Argentina 2000 (pp.332-337). Buenos Aires: Fundación Vida Silvestre Argentina.
Zalba, S. M. (2010). Controle de espécies exóticas invasoras em áreas protegidas naturais: aprender fazendo. Cadernos de Mata Ciliar, São Paulo, 3, 23-27.
Zalba, S. M. (2013). Incorporando el desafío de las invasiones biológicas a los proyectos de restauración ecológica. In D. Pérez (Ed.), Restauración ecológica en la diagonal árida de la Argentina (pp.61-72). Buenos Aires: Editorial Vazquez Mazzini.
Zalba, S. M., Cuevas, Y. A., de Villalobos, A. (2009). Lecciones aprendidas durante siete años de control de pinos invasores en pastizales naturales. In N. J. Cazzaniga, H. M. Arelovich (Eds.), Ambientes y recursos naturales del sudoeste bonaerense: Producción, contaminación y conservación. (Actas de las V Jornadas Interdisciplinarias del Sudoeste Bonaerense) (pp.325-340). Bahía Blanca: Ediuns.
Zalba, S. M., Villamil, C. B. (2002). Invasion of woody plants in relictual native grasslands. Biological Invasions, 4, 55-72.
Zalba, S. M., Ziller, S. R. (2007). Adaptive management of alien invasive species: putting the theory into practice. Natureza & Conservação, 5 (2), 86-92.
Zenni, R. D., Simberloff, D. (2013). Number of source populations as a potential driver of pine invasions in Brazil. Biological Invasions, 15, 1623-1639.
Zenni, R. D., Ziller, S. R. (2011). An overview of invasive plants in Brazil. Brazilian Journal of Botany, 34, 431-446.
Zenni, R. D., Bailey, J. K., Simberloff, D. (2014). Rapid evolution and range expansion of an invasive plant are driven by provenance-environment interactions. Ecology Letters, 17, 727-735.