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Towards Sustainable Forestry Development in Patagonia: Truths and Myths of Environmental Impacts of Plantations with Fast-Growing Conifers

Javier E. Gyenge^{1,2*} • María Elena Fernández^{1,2} • Verónica Rusch² • Mauro M. Sarasola² • Tomás M. Schlichter¹

¹ Consejo Nacional de Investigaciones Científicas y Técnicas, CONICET, Argentina

² INTA EEA Bariloche, Forest Ecology Research Group, CC 277, Bariloche, Río Negro, Argentina

Corresponding author: * jgyenge@bariloche.inta.gov.ar

ABSTRACT

In NW Patagonia region, Argentina, South-America, there are about 70,000 ha of planted forests replacing from native forests to grasslands with different degrees of deterioration due to previous land use. Although forestry development has been quite slow in this region compared to other regions of the country, it is expected that this activity will be increasing in the future due to provincial and national government policies of forestry incentives. In general, only scarce information is available about the environmental impact of forestation in the region. Our objective was to bring together the knowledge about the changes in biodiversity, water cycle and water resources, soil characteristics and the risk of invasion of introduced exotic fast-growing coniferous species on native ecosystems. The analyses revealed that the greatest changes in biodiversity, water consumption and invasion risk occurred when the introduction of trees was in grasslands compared to forest or shrublands. However, from our results we can conclude that, at the current developmental stage of forestry activity in Patagonia, the negative environmental impact is very low or even nil, with the positive impacts –economic and social– possibly being higher and leading to a more positive balance as a whole. However, we recognize that potential negative impacts, whose magnitude will depend on several aspects discussed in the paper, could increase in the future in relation to the expansion of the forested areas. With the available information we can then formulate prescriptions and management strategies for exotic systems, in order to guarantee the long term sustainability of the activity. In this regard, we have the opportunity of developing a sustainable production activity from the very beginning.

Keywords: biodiversity, environmental impact, forest plantations, invasion risk, water resources

Abbreviations: ENSO, El Niño Southern Oscillation; LAI, Leaf area index (m^2m^{-2})

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INTRODUCTION: THE CHALLENGE OF SUSTAINABLE FOREST MANAGEMENT IN PATAGONIA

At the “1992 Rio de Janeiro Earth Summit”, with the introduction of the term “Sustainable Management”, mankind

agreed to start the process of thinking and acting to reduce the environmental impacts of the human activities. Within the many commitments made at an international level forestry was one of the first activities that introduced the knowledge of ecosystem components and their interaction, into management programmes. In this regard, in order to

estimate the impact of forest management, it is crucial to estimate the changes in the different processes of forest functioning, emphasizing on the changes of biological, ecological and economic key variables which determine the actual magnitude of the impact.

The knowledge necessary to determine key variables and their response to forest management is uneven. This depends on several factors which include the development of forestry activity in a particular region and the development of scientific knowledge itself and they are all related to complex social, economic and political factors. In particular, in Patagonia, Argentina, South-America, there are about 70,000 ha of planted forests, mainly *Pinus ponderosa* Dougl. ex Laws (80% of afforestations), and in a minor degree, *Pseudotsuga menziesii* (Mirb) Franco, *Pinus contorta* Douglas and *Pinus radiata* D. Don. Although forestry development has been quite slow in this region compared to other regions of the country, it is expected that this activity will be increasing in the future due to provincial and national government policies of forestry incentives (e.g. National Law 25080 for Planted Forests promotion).

Coniferous plantations are frequently called "green deserts" referring to their supposed lack of plant and animal life inside them. In the case of Patagonian plantations, this image is mainly due to the fact that most of them were never managed; they received neither thinning nor pruning thus generating dense and highly shaded environments. Many city inhabitants in NW Patagonia, value environmental conservation as they live by choice close to or inside Protected Areas. Natural, heterogeneous, old growth deciduous forests are sometimes compared to dense, rather young, and large areas of exotic coniferous plantations with structural and floristic homogeneity. Some specific elements of biodiversity, as big woodpeckers, are frequently looked for and used as indicators of global biodiversity. Thus, the lack of them inside young forest plantations contributes to the idea of plantations being the "bad guy of the film". These selective observations have produced a strong opposition in part of the regional community against this productive activity because they consider that conifer plantations eliminate the native biodiversity where they grow, they use a lot of water resources and they can invade areas outside their borders. However, less than a decade ago, there was no scientific base demonstrating these perceptions, and that is why we decided to study the potential impacts of this productive activity with the idea of proposing mechanisms to make forest production and environment quality conservation compatible.

Based on this background, the objective of this paper was to review and analyze the available information concerning the environmental function and impact on certain ecosystem components by afforestation with fast-growing species, in order to outline recommendations contributing to the sustainability of this productive activity in temperate systems, with emphasis on the Patagonian region. This study is focused on the issue of the sustainability of afforestation with fast growing exotic species in N.W. Patagonia, South-America, in which the authors have worked during the past decade.

This paper reviews information on the following topics: Productivity, biodiversity, water resources, invasion risk, soil acidification, and then integrates them in order to develop recommendations and conclusions. Each section briefly presents information of other authors in other parts of the World, and then summarizes the available regional data.

ENVIRONMENTAL AND PRODUCTIVE FRAMEWORK

The climate in North Western Patagonia is strongly influenced by the winds coming from the Pacific Ocean. The highest proportion of annual precipitation occurs during the cold season as rain or snow (Paruelo *et al.* 1998). The Andean mountains (Cordillera de los Andes) in the West acts as a barrier producing a sharp precipitation gradient from

West to East. In addition, at the inter-annual time scale, the ENSO phenomenon introduced a higher variability in the amount and time distribution of precipitations (Paruelo *et al.* 1998). Western winds also have produced a heterogeneous distribution of volcanic ashes, determining large differences in soil depth. These variable climatic characteristics at short geographic distances linked to different soil types, depths and aspect, produce a heterogeneous native vegetation distribution, as well constrain productive activities based on primary production.

Patagonian native forests are located in the humid part of the precipitation gradient (at different heights of the Andean Mountains). From West to East, the native ecosystems gradually change from grasslands to steppes, with ecotone areas in the middle (Fig. 1). Recent studies estimated that the surface covered by native forests in N.W. Patagonia is approximately 1.660.000 ha (SAYDS 2007). On the other hand, the amount of surface that could be planted with exotic fast growing species was estimated in 2.000.000 ha, mostly in the ecotone region between native forests and the steppe (SAGPyA 1999). Other more conservative studies suggest that, considering economic and social factors, the potentially forested area is much lower, but even in that case, there is an important area to be considered for this relatively new (see below) productive activity in Patagonia. As mentioned in the introduction, at present, afforestation with fast growing exotic species (mainly with *Pinus* spp. and *P. menziesii*) is still a relatively incipient activity, and although biological-environmental and financial conditions encourage these production systems, people concerned about environmental issues do not like "pine plantations" and look at them as a serious threat to the native ecosystems they replace. In this regard it should be mentioned that pine plantations (i.e. those with *P. ponderosa* and in a lower proportion, *P. contorta*) are installed in the drier sites, usually replacing native grasslands (most of them highly deteriorated by former sheep overgrazing) and in places formerly occupied by the native cypress *Austrocedrus chilensis* (D. Don) Pic. Ser. et Bizarri forests. This is a native conifer species that has similar environmental requirements to the ponderosa pine. Extended areas of this type of native forest was converted to open grasslands at the beginning of the XX century for cattle and sheep raising, which has been since then the most traditional activity in Patagonia. The *P. menziesii* plantations are located in more humid places, replacing *Austrocedrus* forests and shrublands dominated by *Nothofagus antarctica* (G. Forst.) Oerst and several woody species, locally named "ñirantales".

PRODUCTIVITY OF NATIVE SYSTEMS AND FOREST PLANTATIONS

Mean annual productivity in NW Patagonia depends on precipitations but also on the presence of natural wetlands, locally called "mallines". Grassland productivity can vary between 0.5 and 2.5 ton ha⁻¹ year⁻¹ (in places with 150 and 600 mm of mean annual precipitation, respectively, Golluscio *et al.* 1998), but those values can increase to 3-9 ton ha⁻¹ year⁻¹ in the wetlands (Giraud 1997; Golluscio *et al.* 1998).

Less data are available about native forest productivity. Regarding, *N. antarctica* forests, Pablo Laclau (pers. comm.) estimated an annual production of 1.5 ton ha⁻¹ at age 16, 3.3 ton ha⁻¹ at age 25 and 4.4 ton ha⁻¹ at age 34 (mean annual precipitation: 2400 mm). For the same species, but growing in Tierra del Fuego (Southern Patagonia), the annual productivity was estimated as 1.65 ton ha⁻¹ at age 65 (Martínez Pastur *et al.* 1995). Considering the most conspicuous woody species in a mixed *N. antarctica* forest, stem biomass productivity was estimated as 2.4 ton ha⁻¹ year⁻¹, in a site with 1400 mm annual precipitation (Gyenge *et al.* 2008a). Schlichter and Laclau (1998) have reviewed productivity data of *A. chilensis* forests, and they concluded that, considering only the stem production, these forests produce 2 to 2.5 ton ha⁻¹ year⁻¹. This information implies that aerial productivity of native grasslands and that of native forests

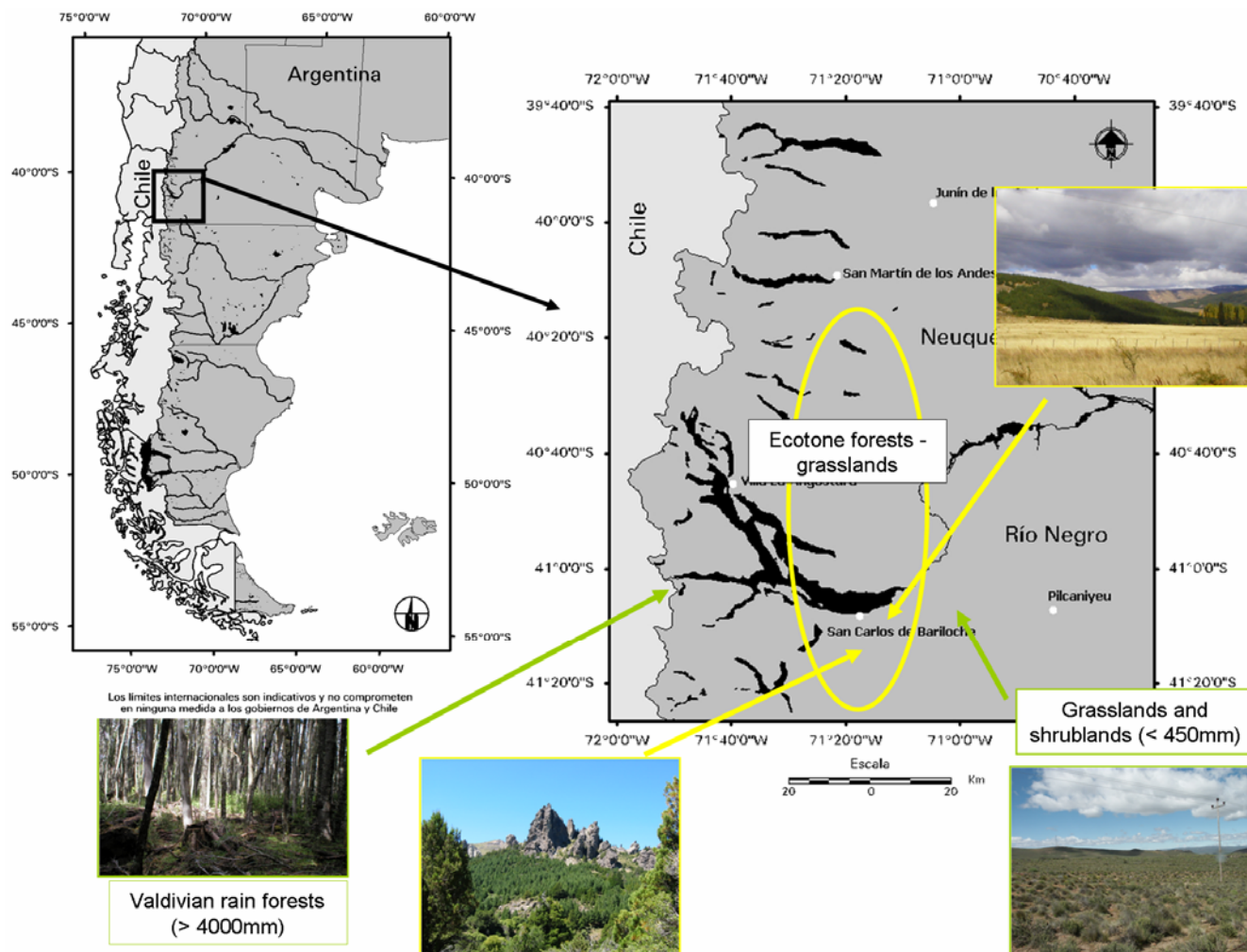


Fig. 1 Study sites location in NW Patagonia, Argentina. Below each type of ecosystem it is indicated its corresponding mean annual precipitation.

are quite similar.

On the other hand, estimations of pine plantation productivity in the area between 500 and 900 mm of mean annual precipitation (always outside the wetlands) varies between 4 and 10 ton ha^{-1} (Schlichter and Laclau 1998). This means that, as expected, productivity of plantations with fast growing exotic species is much higher than that of all types of the native systems they replace, suggesting important changes in systems function and amount of used resources.

BIODIVERSITY ASPECTS

What is known in the world?

Though afforestation determines changes in biodiversity, the activity was related more to biodiversity than to other activities, like agriculture or pastures (Hobbs *et al.* 2003; Stephens and Wagner 2007). However, diversity and abundance do not always decrease in forest plantations. In some cases, such as degraded systems, plantations even benefit native diversity. Abandoned plantations of *Pinus resinosa* Sol. ex Aiton in Canada, for example, allowed the establishment of shade tolerant, native tree species belonging to the late successional stages of the original system, the most valuable elements of a conservation strategy (Mosseler 2005). In areas where the forest has been transformed into a pasture, they are also useful to improve ecological connectivity, allowing animal species to move from one patch of natural forests to another (Brockerhoff 2005; Pawson *et al.* 2008).

In contrast to what is commonly thought (Lang 2005; Overbeek 2005; Jiang 2009), what has been found is that

plantations are not “green deserts”, but that they are new habitats (Carnus *et al.* 2006; Brockerhoff *et al.* 2008; Cummings and Reid 2008). Species guilds vary with the forest age, botanical species and management (amount of herbicides and intensity of soil preparation, number of planted species, length of the rotations, amount of dead wood or shrubs, thinning intensity, among others) and factors at a greater scale (for example climate and lithology, or landscape context, distance to native forests, design of open areas, etc.).

The light environment of the plantation is critical to determine vegetation diversity. Plantations with a high canopy cover eliminate many mid-tolerant species, some of them of great value for ecosystem processes. Humphrey *et al.* (2002) recommend intense thinning in dense plantations of pine and fir in Britain for the protection of lichens though early thinning seems to be better as original vegetation is preserved in the stand. The light environment is also related to plantation age. Nagaike *et al.* (2003), for example, found the greatest vegetation diversity in 15 years old *Larix kampfieri* (Lamb.) Carr. plantations in Japan. Very old plantations can also recover many species and the understory becomes more similar to that of native forests (Brockerhoff *et al.* 2003). The vertical structure of the stand also differs between ages, and also influences vegetation composition (Ferris *et al.* 2000) so partial cutting of the stand (and deadwood permanence) are recommended as good management practices in Britain. When all the community is analyzed it is commonly found that “generalist” species are found in plantations (Pomeroy and Dranzoa 1998), referring to species that can use many different trophic or habitat resources. In some forests, many of those species may be also valuable for conservation (Hobbs *et al.* 2003). One of the most af-

ected groups is that of birds that need cavities for nesting, an habitat element that is not frequently found in a commercial plantation, so old growth stands (Humphrey 2005) and dead logs standing or on the ground (Land *et al.* 1989) are needed. In many areas with altered soil, vegetation lower layers cover increase during the first decades after planting, and numbers of little mammals also increase (Atkeson and Johnson 1979). Vertical structure of the vegetation is also highly related to different fauna components (Humphrey *et al.* 1999). In consequence, intense thinning of plantations is also recommended as a practice for fauna diversity maintenance, two examples of this are the bird “urogallo” in Spain (MacMillan and Marshall 2004) and the arthropod species in Japan (Ohsawa 2004). Early thinning and long rotations are practices even more recommended for the maintenance of key original elements (Hartley 2002).

Another way to maintain fauna diversity is the combination of different tree species, as demonstrated by Clout and Gaze (1984) in New Zealand and Gjerde and Saetersdal (1997) in Norge. Original ecosystem composition is also crucial: when grasslands are planted, changes of flora, fauna or both are usually greater than when original forests are replaced by plantations.

The spatial scale is strongly influenced by the dispersive ability of the biological group under analysis. Carabide abundance and diversity in *Pinus pinaster* Ait. plantations in France, for example, are related to tree height and not to landscape features as occurs in bird diversity, which is usually strongly determined by the presence of native forest patches close to plantations (Barbaro *et al.* 2005, 2008). Leaving remnant patches of the original vegetation inside the plantation area is also a useful tool for native species conservation (Lindenmayer *et al.* 2002, 2009). Other landscape variables have to be considered, as size, form of the patches and amount of border, proportion of the habitat replaced by plantation, fragmentation or distance to native systems, among others.

So, a new paradigm has been generated, where small changes in the design and management can improve biodiversity without reducing productivity. But a key point that is seldom considered is the relative importance of the different elements of diversity for conservation objectives or because of its key role in ecosystem functioning (Cardinale *et al.* 2009). There is no consensus about which elements should be preserved or managed in plantations. In our region, we are working with a proposal: take care of priority sites, habitats and species, and maintain a connected matrix of natural ecosystems.

What is known in Patagonia?

Different types of natural vegetation have been replaced and planted with *P. ponderosa* (pines) in NW Patagonia. At the

beginning many plantations were established on areas coming from xerophytic *A. chilensis* forests (cypress) and sometimes *Nothofagus* forests or shrublands, while in the last decades, semiarid steppes were the main lands occupied by these plantations. As mentioned before, most plantations received no management and as a consequence they became dense and highly shaded. We analyzed the changes in flora and fauna determined by pine plantations, looking at three spatial scales: the region, the landscape, and the site. Site variables included low density or low canopy cover plantations as we were especially interested in studying the consequences of early thinning practices.

1. A look at the whole region

The ways to address the impact of pine plantations at the regional scale were: the replacement of biodiversity key zones by plantations, and the level of occupancy or fragmentation of the natural area with them. Priority areas for conservation were determined through workshops (Bran *et al.* 1999; Vila *et al.* 1999; Rusch *et al.* 2008), based on published (Chehébar *et al.* 1986; Monjeau *et al.* 1994; Aizen and Ecurra 1998; Úbeda *et al.* 1999; Marcelli and Gallo 2000; Prémoli and Kitzberger 2000) and unpublished works of scientists from the region. Those sites were important for their high diversity, level of endemism, presence of endangered species, unique processes or ensembles. Thirty five sites were identified, described and mapped, mainly because of the presence of endangered trees like “larch” (*Fitzroya cupressoides* (Molina) I.M. Johnst.), “Guaitecas cypress” (*Pilgerodendron uviferum* (D. Don) Florin), “araucaria” (*Araucaria araucana* (Molina) K. Koch), and relict populations of *A. chilensis*. Other important elements were the “huemul deer” (*Hippocamelus bisulcus* Molina), the otter “huillin” (*Lontra provocax* Thomas) and several endemic frogs. Those priority areas did not overlap with pine plantations (Rusch *et al.* 2005). “Risk zones”, i.e. those that have a great possibility of being replaced with plantations, were also drawn and in two cases they overlapped areas for biodiversity conservation (one of araucaria forest and another one with the unique ensemble of *A. chilensis*, *P. uviferum* and *F. cupressoides*).

The analysis of landscape metrics showed that pine plantations, do not represent an important factor of landscape fragmentation. The proportion of planted areas is less than 1% at the regional level, and is close to 10% when circles of 10 km radio are drawn around the centre of the plantations. Natural systems connectivity (connectivity index, Turner *et al.* 2001) is also kept between high values, being reduced by plantations from 0.5 for natural grasslands, to 8% for dense *Austrocedrus* forest areas and intermediate values for *Nothofagus* forests and shrublands (Rusch *et al.* 2004, Fig. 2). All these results show that the relative area currently

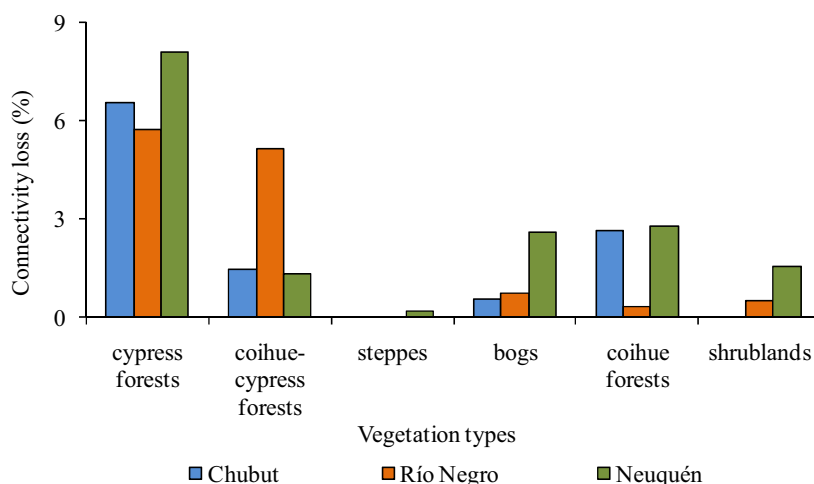


Fig. 2 Changes in connectivity index (Turner *et al.* 2001) due to pine plantations in six vegetation types, through the three NW Patagonian provinces. Adapted from Rusch and Vila (2008).

occupied is very small, the native ecosystems maintain their connectivity and these plantations are not planted on critical areas for biodiversity conservation.

As detailed in previous sections, huge areas could be planted in the future. As we have a sound analysis of relevant sites for conservation and distribution of endangered species, plantation prescriptions are given in order to respect those areas. Consideration of these priority sites for conservation could be the first step when analyzing where and how much plantations could be developed in a region.

2. A look at plantations, management and design

Vegetation: In the region, dense monospecific plantations of pine are associated with the reduction of the understory. This leads people to assume that other components of the community are reduced in the same degree. Nevertheless, it has been demonstrated that animal populations do not change in the same degree (Rusch *et al.* 2005; Paritsis and Aizen 2008). Let us analyze changes in each group. Pine canopy covers close to 90%, determine reductions of vegetation species (grasses, shrubs and forbs) richness from 24.5 to 12.0 species of *N. dombeyi* (Mirb.) Oerst. (“coihue”) forests are replaced (Paritsis and Aizen 2008); 31.7 to 10.4 species in steppe areas and 42.4 to 15 species in *A. chilensis* forests (Rusch *et al.* 2005). Impact on the understory cover is even greater: from 39.6 to 4.0% when *N. dombeyi* forests are replaced (Paritsis and Aizen 2008), from 45.5% to less than 0.5% in steppe areas and from 35.9 to 2.7% in *A. chilensis* forests (Lantschner *et al.* 2008).

Nevertheless, intense early thinning practices change the situation. The most interesting point of these situations is that many key species are maintained in the system and their condition even improved. *Festuca pallescens* (St. – Yves) Parodi, the palatable and dominant grass of the original steppe is the one which showed a higher increase in their cover under plantations. Though shade commonly reduces grass production, Fernández *et al.* (2004) demonstrated that pine plantations may facilitate the growth of *F. pallescens* through the effect of shading on the reduction of water demand. On the other hand, the spiny shrub “neneo” (*Mulinum spinosum* (Cav.) Pers.) and the more water resistant grass but less palatable *Stipa speciosa* Trin. & Rupr. (syn. *Jarava speciosa* (Ruiz & Pav.)) were significantly reduced. While the most abundant native grazers of these steppes used to be the guanaco (*Lama guanicoe* Muller), that preferred shrubs to grasses, domestic cattle and sheep introduced on the early XXth prefer the grasses. It is stated that overgrazing by domestic animals has promoted “the replacement of the highly palatable and dominant tussock grass *F. pallescens* by the spiny shrub *M. spinosum*” (sic) (Aguilar and Sala 1998). As Perelman found (in Aguiar and Sala 1998), non palatable grass species (like *S. speciosa*) also increase with overgrazing. Pine plantations include grazing enclosures and create microclimates that are less water demanding. We suggest that both factors partially allow the restoration of important elements of the original system and the most important component of these steppes for cattle raising because of their abundance and relative high quality. The combination of enclosure and a new microclimate created by the plantation, also favored other species of even higher value, such as *Bromus spp.* So this practice helps not only to maintain part of the natural biodiversity, but also as a forage reserve. The persistence of these key elements with moderate covers also contributes to erosion control and the possibility of recovering the original system.

When comparing native *A. chilensis* forest systems and sparse pine plantations occupying those areas, canopy covers are similar (in our study: 62.5 and 66.9% respectively). Many shrubs are maintained (mainly *Schinus patagonicus* (Phil.) I.M. Johnst. and *Baccharis spp.*). Other shrubs, like *Aristotelia spp.* are very abundant in dense plantations (and an exotic weed, *Rosa eglanteria* L. also tolerates dense shade). One of the most remarkable findings is that more

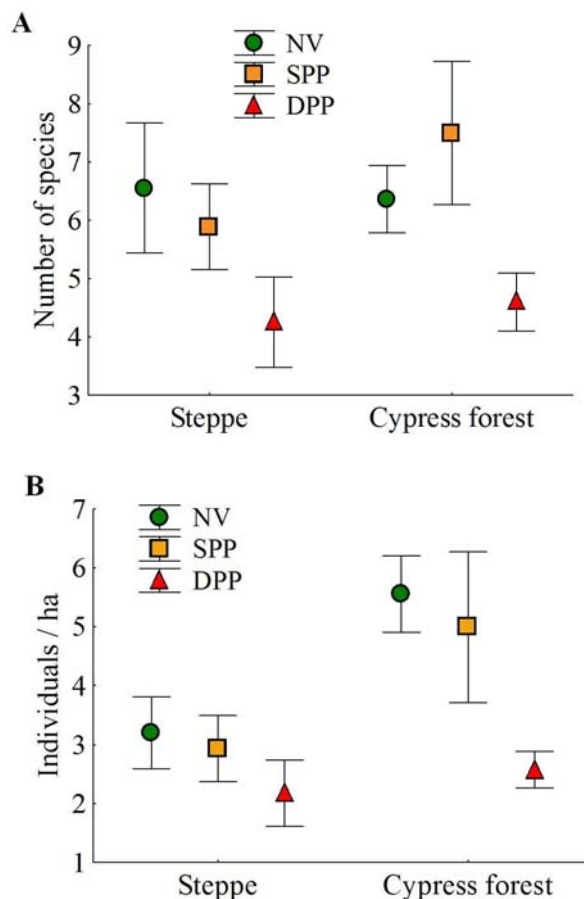


Fig. 3 (A) Richness (N° of species per site) and (B) density (individuals/ha) of birds (means \pm SE) in the different types of vegetation (NV: native vegetation, SPP: sparse pine plantations, DPP: dense pine plantations) in the steppe and in the *Austrocedrus chilensis* forest.

tree regeneration is also found under the sparse pine canopy. Kitzberger *et al.* (1994) described the strong relationship between *A. chilensis* juveniles with shrubs, indicating that shrubs favorably influence tree regeneration during dry habitats and periods of drought. These shrub/nurse plants appear to be required for successful tree seedling establishment. The nursery effect was also studied in the region on different species after fire (Raffaele and Veblen 1998) and precisely microsites beneath *S. patagonicus* were characterized by lower and more heterogeneous light levels but by greater soil moisture. This nursery effect is useful, and outweighed the negative effects of light competition until plants are 4 m height in dry areas or periods, though this positive net effect ends when plants reach 0.50 m on wet areas or periods (Letourneau 2005). So we could assume that planting in a sparse way would also favor the possibility of natural forest recovery, which is the basal and dominant element of the ecosystem and may favor the resilience of the system as well. This is a second step.

The effect of plantations on the “movement” of plants (the possibility of pollen to cross the planted area) has not yet been studied. Most of the plants of the steppe area are carried by wind, so plantations may act as filters, but not as barriers for plant connectivity.

Fauna: Bird communities are usually used as environmental indicators because of the high mobility of their individuals and the possibilities of measurement standardization. Bird communities in pine plantations were similar to the replaced forests of *N. dombeyi* (Paritsis and Aizen 2008) and *A. chilensis* (Lantschner *et al.* 2008), and *Nothofagus* shrublands (Lantschner and Rusch 2007). When plantations on steppes and areas of *A. chilensis* forests are dense, richness may decrease, but bird community composition remained

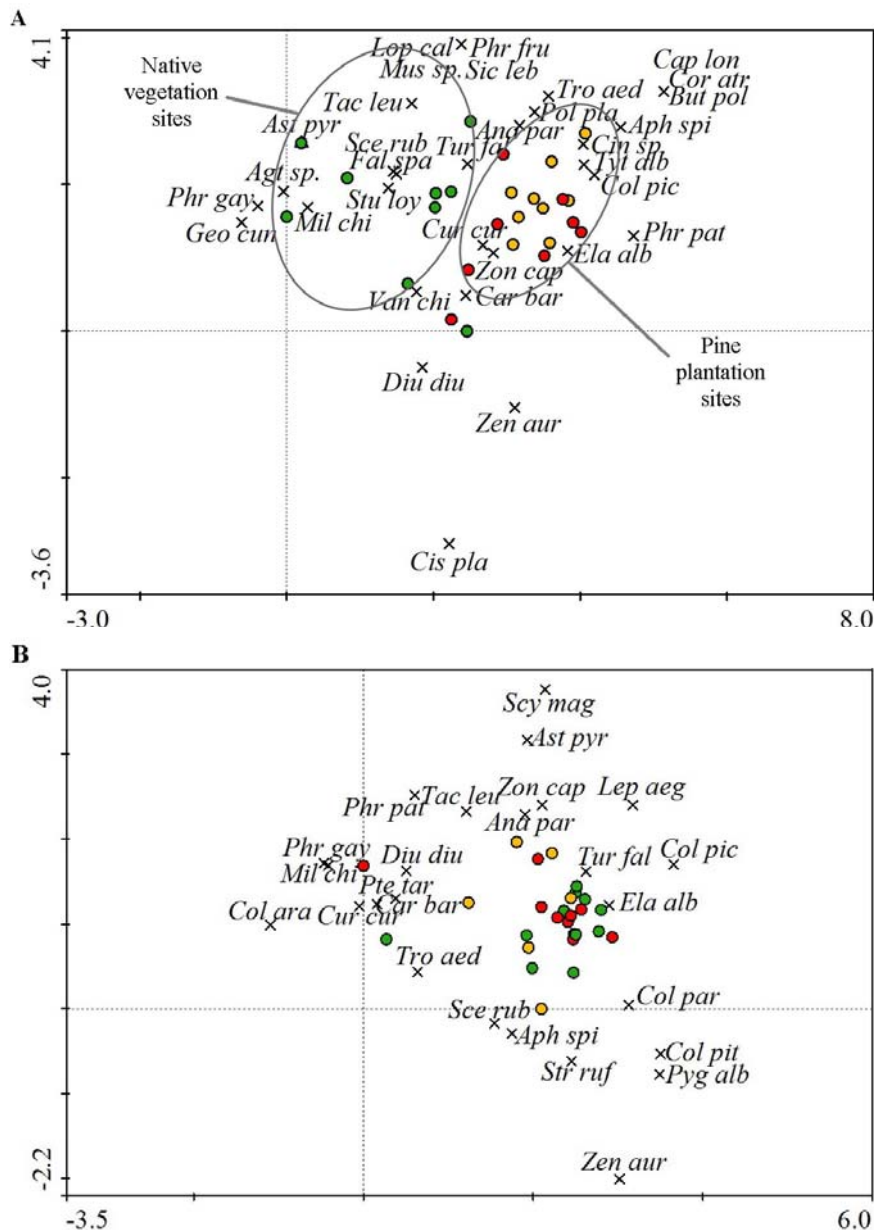


Fig. 4 Ordination plots for the first two axes of the detrended correspondence analysis of bird abundance data in the studied sites. (A) Steppe area. Sites: steppe native vegetation (green circle), sparse pine plantation on steppes (orange circle), dense pine plantation on steppes (red circle); (B) *A. chilensis* forest area. Sites: *A. chilensis* forest native vegetation (green circle), sparse pine plantation on *A. chilensis* forest (orange circle), dense pine plantation on *A. chilensis* forest (red circle). Species: (x) are indicated with the three first letters of the genus and the species: *Agriornis* sp.; *Anaethes parulus*; *Aphrastura spinicauda*; *Asthenes pyrrholeuca*; *Buteo polyosoma*; *Caprimulgus longirostris*; *Carduelis barbata*; *Cin* sp.; *Cinclodes* sp.; *Cistothorus platensis*; *Colaptes pitiuus*; *Colorhamphus parvirostris*; *Columba araucana*; *Columba picazuro*; *Coragyps atratus*; *Curaeus curaeus*; *Diuca diuca*; *Elaenia albiceps*; *Falco sparverius*; *Geositta cunicularia*; *Leptasthenura aegithaloides*; *Lophortyx californica*; *Milvago chimango*; *Muscisaxicola* sp.; *Phrygilus fruticeti*; *Phrygilus gayi*; *Phrygilus patagonicus*; *Polyborus plancus*; *Pterotochos*; *Pygarrhichas albogularis*; *Scelorchilus rubecula*; *Scytalopus magellanicus*; *Sicalis lebruni*; *Strix rufipes*; *Sturnella loyca*; *Tachycineta leucopyga*; *Troglodytes aedon*; *Turdus falcklandii*; *Tyto alba*; *Vanellus chilensis*; *Zenaida auriculata*; *Zonotrichia capensis*.

relatively constant. These similarities were even greater when plantations were sparse (Fig. 3).

When plantations are established in a structurally highly different system like steppes, bird richness and abundance do not change significantly either but the community composition does (Fig. 4). The community of birds from the steppe is partially replaced by a new community, similar to that of ecotonal forests (with dominant species like *Aphrastura spinicauda* Gmelin, Thorn-tailed “Rayadito”). Steppe species like *Sturnella loyca* Molina-Long-tailed Meadowlark-, *Asthenes pyrrholeuca* Lesser “Canastero”-, *Phrygilus gayi* Gervais-Gray hooded Sierra Finch- and *Diuca diuca* Molina -Common “Diuca”-Finch- disappear in plantations (Lantschner et al. 2008).

When comparing bird communities in *P. menziesii* plantations with those of *N. antarctica* mixed shrubland and low

forests, only small changes were found, as the native system loses some rare species. Nevertheless, other activities (like logging and grazing) determine strong changes, though they incorporate some open-area species to the forested area (Lantschner and Rusch 2007), increasing γ diversity.

So vegetation structure seems to be the driving factor in determining bird community, and both dense and sparse plantations are similar to native forests but they are unsuitable for most steppe bird species. At the stand-scale, the maintenance of some forest structural elements could also allow the conservation of forest birds in planted forests. In this sense, the presence of native understory vegetation is of great importance, thus ideal management of pine plantations should enhance the native understory vegetation in order to facilitate their beneficial effects on the remaining native habitats, planting at low densities or with early thin-

ning.

To evaluate the relative importance of site and landscape species, habitat models can be constructed. A habitat model was developed for one keystone bird species (determined by Aizen *et al.* 1999), the “Chucao” (*Scelorchilus rubecula* Kittlitz), considering site and landscape variables (Rusch and Lantschner 2006). Site variables included cover of different layers, cover of shrubs like “calafate” (*Berberis* spp.) or “bamboo” (*Chusquea culeou* Desvieux), altitude, aspect, latitude and longitude (index of precipitation in the region). Landscape variables include diversity and proportion of native vegetation types, number of plantations, among others and were analyzed at concentric circles of 500, 1000 and 2000 m around each site. The logistic model to predict the chucao’s presence showed that “shrub cover” was the most important variable explaining near 85% of the variability, while bamboo growth sharply increased the probability of finding the species. This information, together with the knowledge of the low dispersion capability of the chucao, determines that connectivity of its habitat should be defined through the continuity of a dense shrub strata or the presence of bamboo. On the other hand, another keystone species (Aizen *et al.* 1999), the “magellanic woodpecker” (*Campephilus magellanicus* King) needs mainly mature *Nothofagus* spp. trees, but their ability to fly up to 5 km can also transform a landscape composed by “forest patches” in a “continuous forest” for these individuals, as open areas are not barriers for the species.

Estades and Temple (1999) analyzed the effect of plantations on “hualo” (*N. glauca* (Phill.) Krasser) forests birds also employing circular plots, and they found a negative relationship between the size of forest fragment and bird species richness per unit area. Fragmentation effects were mostly species-specific. A few large-sized bird species tended to be absent from the smaller fragments, while the presence or abundance of most birds showed no relationship with fragment size. The type of vegetation adjacent to forest fragments had a significant effect on the composition of the bird community inhabiting them. Though the abundance of most cavity-nesting species in pine plantations was related positively to the proximity of either the nearest native forest fragment or the nearest creek, most open-nesting species in pine plantations depended mostly on the characteristics of the vegetation in the understory.

Nevertheless, the maintenance of steppe species in a forest planted area should be assured at landscape scales, maintaining the connectivity of the native matrix, minimizing the fragmentation of bird populations.

Ant assemblages were also analyzed in native systems and in plantations with dense and sparse canopy covers. Ant abundance and diversity within dense plantations were low but the more open plantations sustained ant assemblages which resembled those of the native steppe samples as found with vegetation and birds. Though tree presence may be responsible for the changes in the ant assemblages, the presence of native vegetation cover predicts ant assemblage composition in these arid zones. A more open plantation design will be accompanied by higher ant abundance and species richness, which in turn may be favorable for sustainable forest management in the area (Corley *et al.* 2007).

We are now analyzing carnivores’ response to plantations. Though larger home ranges surely determine important relationships between movement and landscape structure, many important site variables seem to influence their presence (Victoria Lantschner, pers. comm.). Studies on the Chilean Patagonia, where plantations are more extensive, and inserted in humid forest regions, *Oncifelis guigna* (“guigna” cat, *Felidae*) preferred habitats with dense bush cover, far from roads and close to large patches of native forests, almost exclusively restricted to this type of habitat. On the other hand, *Pseudalopex culpaeus* (culpeo fox, *Canidae*) preferred open habitats, close to roads, relying extensively on pine plantations. “Thus, forest size reduction and isolation may reduce the *O. guigna* survival in a pine dominated landscape. In contrast, *P. culpaeus*, a generalist spe-

cies that would use habitat depending on its abundance, may adapt to changes in this landscape” (sic) (Acosta and Simonetti 2004).

As a general conclusion, pine plantations can provide habitat for a substantial number of native species, and this feature varies both with management practices and with the landscape context where afforestation occurs. Consideration of the structure of the landscape, combined with the stand structure that allows the persistence of the fauna, would be the second step to analyse when considering where and how many areas can be converted to forest plantations in a region. Nevertheless there is still not enough knowledge to determine the maximum proportion of converted land in a given landscape, or the minimum size of corridors to make them functional.

WATER CYCLE AND WATER RESOURCES

What is known in the world?

Measurements and modeling at the catchment scale indicate that, although the response of mean annual evaporation to vegetation change depends on the particular limits and controls over water fluxes in each region, a reduction in forest cover in general increases water yield (Calder 1998; Zhang *et al.* 2001; Andréassian 2004). In this regard, on an annual basis, grasslands always show a lower annual evapotranspiration than forests when the annual rainfall exceeds 500 mm (Zhang *et al.* 2001). In drier places, models presented by those authors (Zhang *et al.* 2001) indicate that evapotranspiration of grasslands and forests can be similar. On the other hand, generalizations about water cycle and water use of different vegetation systems is difficult because limits on evaporation may be related to climatic factors (radiation, advection, soil water availability), physiological factors (hydraulic resistance, canopy conductance, root development) and specific site factors (soil texture and depth) (Calder 1998; Roberts 2000; Andréassian 2004). The model of Zhang *et al.* (2001) agreed with Farley *et al.* (2005) who found that depending on the tree species introduced, the afforestation on grasslands or shrublands may produce reductions in runoff, and they suggested that this could be most severe in drier regions. In other words, the probability of finding an increase in evapotranspiration in response to the afforestation of grasslands is higher than that of replacing forests, the magnitude of changes depending on the local factors limiting water evaporation.

Any change in the type, cover or structure of vegetation, produces changes in the relationship between surface- and ground-water (Le Maitre *et al.* 1999). Rain interception, i.e. the adsorption of rain in shoots and stems followed by direct evaporation to the atmosphere, depends on the total aboveground area of the plants. In this sense, trees develop higher leaf areas than grasses and shrublands, and this can lead to a lower amount of water reaching the soil, decreasing also the infiltration and/or the runoff rate. In addition, part of the rain could concentrate in the base of the tree following the vertical direction of branches and stems (stemflow), changing the spatial distribution of soil water. On the other hand, due to their high root development, trees (but also other life forms) could redistribute water between soil layers with different soil water content which could change the resource availability for other species (e.g., Dawson 1993). Also, and due to their higher canopy and root surface, trees can use more water than grasses, changing the yield from the system and altering the rate of percolation that could migrate to other economies. Finally, shade produced by the trees decreases the evaporation from the soil and from the grasses growing under the canopies. This also can change the water cycle within the system.

The magnitude of the changes in the fluxes depends on both the tree and the replaced species. In a global synthesis, annual runoff was reduced in a higher magnitude in grasslands than in shrublands when they were replaced by forest plantations, and also, a larger reduction was produced by

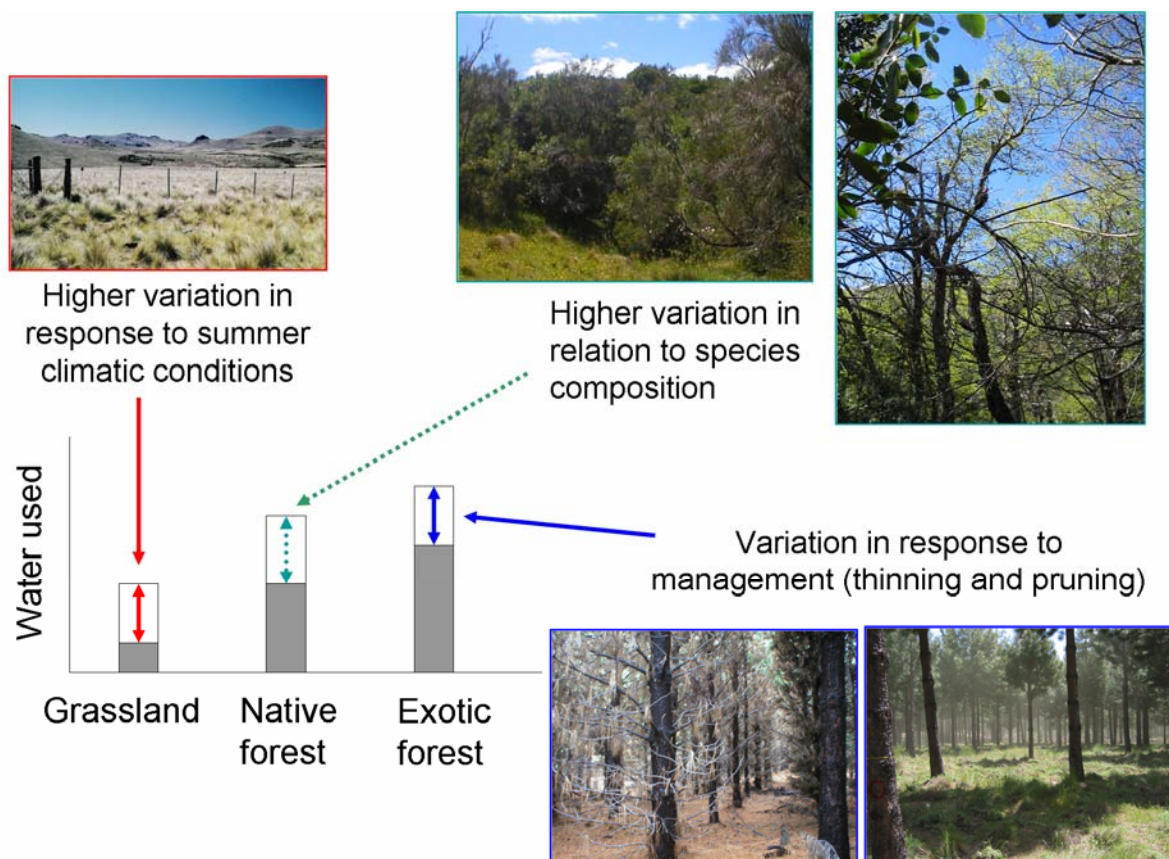


Fig. 5 Schematic representation of the amount of water used by grasslands and forests. Variations in each type of systems depends primarily on a different factor: climatic conditions in the case of grasslands (great inter-annual variability), species composition in the case of native forests and management options in the case of forest plantations (exotic forests).

Eucalyptus plantations than by pine plantations (Farley *et al.* 2005). In particular, rain interception by canopy and litter, and water used by the trees (both factors which produce a decrease in water outflows) change with the species, size of the individuals and the degree of intra- and inter-specific competition (Eastham *et al.* 1990; Putuhena and Cordery 1996, 2000; Lagergren and Lindroth 2004).

What is known in Patagonia?

Based on the differences in annual productivity between forest plantations and native systems, and due to the close relationship between water loss by transpiration and carbon fixation by photosynthesis, the hypothesis driving our studies was that exotic fast growing species introduced in N.W. Patagonia use more water than the replaced native species, the differences being water use of a similar magnitude to those found in productivity. In addition, we expected additional changes in the water balance (rain interception, deep drainage), mainly comparing the most different systems: plantations and grasslands.

The studies reviewed in this paper were carried out during several years at two main sites: Meliquina valley (684 mm of annual precipitation; 40° 30' S, 71° 10' W) for ponderosa pine studies and El Foyel valley (1414 mm of annual precipitation; 41° 39' S, 71° 33' W), for *P. menziesii* plantations. Both sites are important areas with forest plantations of each species in Neuquén and Río Negro provinces, NW Patagonia, Argentina. In addition we carried out additional studies in other complementary sites with ponderosa pine plantations (Arroyo del Medio, Chall-Huaco valley, both in Río Negro province, Argentina). The main variables studied were soil water use, canopy interception, water sources and transpiration in native and exotic species. To assess the mentioned variables we measured soil water content (with TDR and gravimetric methods), water reaching the soil during a rain event with several rain gauges outside and below tree canopies, sapflow density with Gra-

nier's method (Granier 1985), and we used stable isotopes of H and O for water source determination. We also performed water balances for each site (for particular details of each methodology, see the cited references).

The compared systems in Meliquina valley were *P. ponderosa* plantations (growing at three plantation densities), *A. chilensis* forests (isolated trees and two stand densities) and native grasslands (dominated by *F. pallescens* and *S. speciosa*). In El Foyel valley, we compared *P. menziesii* plantations (with a maximum leaf area index in the region) with the mixed native forests called "ñirantales" of *N. antarctica*, *Lomatia hirsuta* (Lam.) Diels ex J.F. Macbr. ("radal"), *S. patagonicus* and *Diostea juncea* (Gillies & Hook.) Miers. ("retamo").

As expected, the higher differences in water use (mm day^{-1} , mean growing season value) were found comparing forests (irrespectively of the species) with grasslands (**Fig. 5**). This was in accordance with the model of Zhang *et al.* (2001). However, the differences in the amount of water used by the different systems decreased with decreasing leaf area index of the forest (Gyenge *et al.* 2003; Licata *et al.* 2008; **Fig. 5**). Mean daily evapotranspiration of grasslands varied between 0.9 and 2.8 mm day^{-1} depending on climatic conditions of the growing season (Gyenge *et al.* 2002; Gyenge 2005). For similar periods, ponderosa pine plantations transpired in average 12 to 90% more water than grasslands, depending on their LAI (see below for absolute values; **Fig. 5**). Significant differences in soil water extraction below 60 cm of soil depth were observed from January of each year (middle of the growing season) comparing the grassland and the forested systems, demonstrating the more exhaustive use of soil water resources by trees compared to the native grasses (Gyenge *et al.* 2002). Stable isotope determinations indicated that when growing together, pines and grasses use water from different soil layers, thus minimizing interspecific competition for water resources (Fernández *et al.* 2008). Comparing forests of exotic and native species growing in Meliquina valley (the driest site), dense

plantations of *P. ponderosa* used more water than dense forests of *A. chilensis* (3.6 and 2.2 mm day⁻¹, respectively, LAI of both forests ≈ 9 m² m⁻²; Licata *et al.* 2008). However, differences between systems were not constant comparing different years (Licata *et al.* 2008). In this regard, when comparing a wet and a dry growing season, differences in transpiration between dense pine and *A. chilensis* forests increased in a wet year, suggesting a higher ability of the exotic species to use available resources when these are abundant (Licata *et al.* 2008). In both systems, daily transpiration was reduced when LAI was lower (due to a lower plantation density or thinning in the native forest). Pine plantation transpiration was reduced to 2.2 mm day⁻¹ (LAI: 3), and *A. chilensis* transpiration to 1.4 mm day⁻¹ (LAI: 5). It is important to note that whole stand transpiration reduction was not proportional to the reduction in leaf area (or number of trees ha⁻¹) because each individual tree transpired more water per day when intraspecific competition was reduced (Gyenge 2005). Another means of reducing transpiration of a forest is by green pruning of branches. We observed that there was no complete compensation of transpiration by the remnant foliage after a pruning of ponderosa pine (Gyenge *et al.* 2009b).

On the other hand, comparisons carried out between systems located in more humid sites (*P. menziesii* plantations vs mixed ñirantales), indicated that there were no statistical differences between the exotic and the native systems (however, a mean difference of about 1 mm day⁻¹ was observed in mean transpiration between systems) (Gyenge *et al.* 2008; Fernández *et al.* 2009; Gyenge *et al.* 2009b). It should be noted that LAI differences were very high between systems (even when both systems presented the highest LAI reported for each one), 5 and 13 in the ñirantal and *P. menziesii* plantation, but this did not result in a high difference in transpiration (the coniferous plantation transpired in average 33% more water than the ñirantal). Different ecophysiological behaviors in response to climate and forest structure in exotic vs native species could explain the observed low differences in water use (Fernández *et al.* 2009). Important differences were also observed between species of the native forest, therefore the particular species composition and size distribution of trees within each system could lead to important differences in water use of mixed native forests (Fernández *et al.* 2009). Because a high heterogeneity between stands of mixed native forests was observed (Reque *et al.* 2007), the relative impact over water fluxes produced by the introduction of an exotic species will depend on the particular history and current use of this type of native forests.

Considering other components of the water balance, net precipitation (total precipitation minus the amount intercepted in the canopy) was consistently lower below the canopies of *A. chilensis* (40% of the precipitation) than below the exotic species, *P. ponderosa* (63%; Licata 2007). In our studies, *P. ponderosa* plantations used more water in the productive pathway (transpiration) than in the unproductive one (evaporation from the canopy) comparing to *A. chilensis* forests. Differences in transpiration between pine plantations and *A. chilensis* forests are compensated by interception losses, thus both systems have similar total water use (transpiration plus interception). On the other hand, similar rain interception values were observed in *P. menziesii* plantations and ñirantales (approx. 50% of growing season precipitation) in spite of the high differences in their LAI (Gyenge *et al.* 2009). It is important to note that in this case, one of the most conspicuous species of the ñirantales (*N. antarctica*) is a deciduous species. Therefore, it is expected that interception values during winter were lower in the native system than in the evergreen coniferous plantations. In all studied systems, the stemflow component of water balance was almost negligible, with values ranging 0.04% to 3% of total precipitation (Pérez 1989; Licata 2007).

Finally, studies with stable isotopes in soil and xylem water, together with simultaneous and automatic record of soil water content at different soil depths, indicated that, in

spite of the differences in the magnitude of used water, both *P. ponderosa* and *A. chilensis* trees use similar water sources (Fernández *et al.* 2007; Licata *et al.* 2008).

Returning to our original hypothesis, it seems that even when exotic systems use in most cases more water than native systems they replace, affecting also other components of the water balance, it is evident that the magnitude of the differences is not the same as the magnitude of differences in productivity. Water use efficiency (that is, the ratio between biomass production and water used) of exotic systems was always much higher than that of native systems, indicating that the production of a cubic meter of wood or a kg of dry biomass is much cheaper in terms of water both in ponderosa pine and *P. menziesii* compared to all native studied systems. We observed, making comparisons between systems or within them, that more productive individuals/species/stands were those with higher water use efficiency (Gyenge 2005; Gyenge *et al.* 2008; Fernandez and Gyenge 2009). This finding opens very interesting questions about the underlying mechanisms explaining the observed patterns of efficiency of resource use within and between species and systems.

SOIL CHANGES

Another aspect of the environmental impact of plantations is soil change, mainly acidification. Soil pH is also an integrated measure of fertility, as it is strongly related to nutrients amount and availability, and potential toxicity as well as an important factor for the biological activity of soils. Some results of studies in other regions of the world (Binkley 1994; Olsson 1999) have alerted citizens of concern about the environment. Nevertheless the intensity of these activities and the type of soils are very different in each region, determining different responses. In NW Patagonia, Andisols, young rich soils, dominate the landscape, and Molisols are more frequent towards the east, in the western steppe region. Broquen *et al.* (1995) found no differences in the acidity of the solution neither between moist *Nothofagus* forests systems and plantations nor between 27 pair of sites when looking at the first 5 cm (though the interchangeable acidity – measured in KCl – slightly changed -0.2 points- Broquen *et al.* 2002). Nevertheless, Gobbi *et al.* (2002) found that some changes were detected when comparing plantations with dry forests (*A. chilensis*).

The analysis of 33 pairs of natural and planted sites (from moist forests in the west, xeric forests, and steppes in the east of the rain gradient), showed statistical differences, but of a very low magnitude (0.18 pH points) in plantations older than 20 years (Rusch *et al.* 2004b). As mean pH of original systems was 6.0, the reached values were still in the range of the most productive soils (Schlichter *et al.* 2004). So, results show that very little changes occur in soil pH under plantations. It would then be desirable to follow up the process and monitor long term changes (Fig. 6).

INVASION RISK

What is known in the world?

Invasion is a process that occurs when one species is introduced in a new environment, is established in self-sustaining populations without direct and deliberate human help and spreads in extensive areas far away from the original dispersal point. From all the introductions in an ecosystem, only a minute percentage becomes invader. (Williamson 1996). There is no general theory that explains all cases of invasion, but there is consensus that two aspects are involved in the invasion process: traits that enable a species to invade a habitat and the habitat characteristics that determine an ecosystem's susceptibility to the establishment and spread of a species (Lonsdale 1999). Some authors define a species as "invasive" when it can propagate itself in nature over a distance > 100 m from the site of introduction in < 50 years for species spreading by seeds

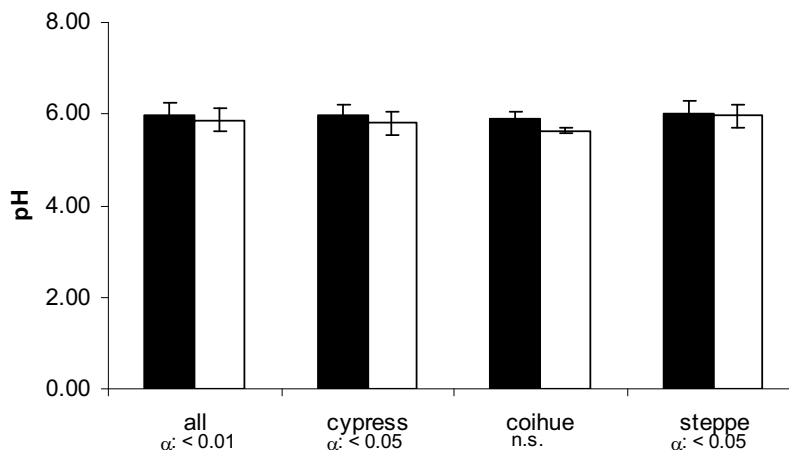


Fig. 6 Soil pH in the first 20 cm in native systems (black bars) and pine plantations (white bars) replacing them.

(Richardson *et al.* 2000).

Biological invasions are considered one of the great threats to biodiversity throughout the world (Williamson 1996). Several impacts related to displacement of native species, alteration of natural regime disturbances (e.g. fire), higher consumption of resources (water and nutrients), alteration in nutrient cycles and others, have been studied and described for different ecoregions (Hobbs and Mooney 1986; Ewel *et al.* 1999; Le Maitre *et al.* 2000). Besides the impact on natural systems, invasion produces several impacts on productive activities (e.g. weed in cultivations) leading to expensive expenditures for its control (Pimentel *et al.* 2000)

Tree invasions and particularly conifer invasion has acquired special attention within biological invasions. This relevance is associated principally to two aspects, the capacity of many pines to invade different habitats and the extensive use of conifers in commercial plantations around the world. In different countries or regions, conifer invasion has become an environmental and economic problem. In the south Hemisphere, South Africa, New Zealand and Australia are good examples where conifer species proceeding from plantations have colonized extensive areas (Richardson and Higgins 1998; Le Maitre *et al.* 2000; Ledgard 2001). These invasion cases generated populations of high or sparse density, but in both kinds of colonizations invasion of these species has been reported to cover hundreds of thousands of hectares.

Many studies have analyzed why *Pinus* species produce invasions (Richardson and Cowling 1994; Rejmanek and Richardson 1996). Where conifer invasions have had success, these species were introduced and planted in extensive areas and a long time ago (Richardson 1999), invasion was positively associated to propagules pressure and the time elapsed to explore and reach favorable conditions. In addition to this, there are some species characteristics that favor their invader behavior as being a pioneer species in the natural habitat, having a natural capacity to colonize marginal areas and being a drought resistance species. Biological traits related to seed production are good predictors of invasiveness of woody species: small seed mass, short age of reproductive maturity and short intervals between large seed crops (Rejmanek and Richardson 1996) favor invasion capacity of a species.

Independently of specific traits of invasive species, there are factors that influence an ecosystem's susceptibility to conifer invasions. In general, invasions are preceded by disturbances. In areas with alterations in natural regime of disturbances, pine invasions are recorded. Fires, deforestation, grazing and browsing, land use changes, fire suppression are disturbances associated with pine invasions (Richardson *et al.* 1991). Associated with the intrinsic resistance of an ecosystem to invasion, Davis *et al.* (2000) postulated a gradient of resistance to pine invasion, in which

closed forests without disturbances are the most resistant systems, followed by open forests, shrublands and finally, the more susceptible grasslands.

The list of conifer species that have become invaders in different habitats is extensive, including *P. ponderosa*, *P. contorta* and *P. menziessi*. The habitats invaded by these species include grasslands, steppes, scrublands, woodlands, temperate open forests and disturbed forests (Richardson and Higgins 1998; Simberloff *et al.* 2008).

One of the principal impacts of conifer invasions is the conversion of native areas to conifer forests with the local disappearance of many native plants (Wardle 1985; Richardson *et al.* 1994; Richardson and Higgins 1998; Ledgard 2001, 2002). In New Zealand, *P. menziesii* and *P. contorta* also have invaded open native forests and canopy gaps (Wardle 1985; Ledgard 2002). This invasion into gaps may threaten regeneration of the native dominant trees (Ledgard 2002). In Australia, New Zealand and specially South Africa, pines cause important problems in the management of pastures, basins and protected areas (Richardson *et al.* 1994; Higgins and Richardson 1998; Richardson *et al.* 1998). Potential impacts could be related to an increase in the intensity of fires and change in the hydrology, particularly where forests replace non-forest vegetation (Le Maitre 2000).

We should differentiate impacts of conifer plantations from impacts of self-sown conifer invasion. Many effects of invasions can be similar to effects of plantations (Richardson *et al.* 1994) but the similarity will depend on the way the site was occupied, i.e. if the invasion generated mono-specific and dense populations or if the occupation was sparse. We can suppose that the effects of these invasions will have some similarities to the effects of dense or sparse plantations, respectively. However, the ecological similarities, including impacts, between plantations and forests that establish outside plantations (due to invasion processes) merit much more research (Simberloff *et al.* 2008).

What is known in Patagonia?

As mentioned before, in Patagonia, the more planted conifer is *P. ponderosa*, and in a minor degree *P. radiata*, *P. contorta* and *P. menziessi*, all of these species are mentioned as invaders in other countries. Therefore, it is very common to hear say in both scientific and non scientific environments: "pines are invading". This generalization probably originated in the cases in other countries (already mentioned) and in the observation of natural pine seedlings and saplings outside conifer plantations in Patagonia. Road margins, highly disturbed sites and small areas close to mature plantations are commonly colonized by conifers, contributing to generalize the idea of pine invasion as a widespread fact. However, in Patagonia, there are few studies that have evaluated this colonization in time, space and in natural or

seminatural systems (Simberloff *et al.* 2002; Sarasola *et al.* 2006). There are also few studies evaluating causes or factors that influence –positively or negatively– conifer invasions (Nuñez *et al.* 2008). The available studies are relatively new, some of them carried out during the last ten years and some of them are still in course.

1. How is the advance of conifer wildings in different ecosystems in Patagonia?

Considering pine invasion in steppe ecosystems, Sarasola *et al.* (2006) evaluated in 22 sites the natural recruitment of pines outside pine plantations of mean age of 22 years. *Pinus contorta* had the greatest recruitment, reaching more distances and higher densities than *P. ponderosa* (>300 vs 50 m, and 420 vs 100 individuals ha⁻¹ with an age > 4 years). Plantations of *P. contorta* producing wildings were more frequent and started recruitment earlier than *P. ponderosa* (at age 12 vs 18 years, respectively). In both species, areas located downwind presented greater recruitment. The distribution pattern of wildings presented higher densities close to plantations decreasing logarithmically toward greater distances from the plantation. Seven-nine years old wildings of *P. contorta* had already produced cones and in some sites their satellite recruitment has been recorded, in some cases up to 8 km (Cecilia Monte, pers. comm.).

There are few measurements of conifer invasions in shrubland ecosystems, showing that in plantations of *P. menziessi* (30-35 years old) the natural recruitment can reach a distance of more than 300 m from the plantation edge. Mean density of recruitment was variable but low (125 to 360 trees ha⁻¹) and 60 to 91% of these individuals were < 0.50 m height. Highest density and dispersion distance were in the downwind sides of the plantation. Other study comparing relative invasiveness in *A. chilensis* and shrublands, showed that colonization in shrublands is lower than in the native conifer forest (Orellana and Raffaele 2008). Our studies in xeric and mesic areas with *A. chilensis* forests, demonstrated that the xeric environments present a quite high susceptibility invasion by *P. menziessi*. This species shows a better capacity to colonize these native forests than *P. ponderosa*. Thirty year old, *P. menziessi* plantations presented wilding fronts of advance of 150 m with mean densities of 1500 individuals (age: > 4 years) ha⁻¹, while wildings of *P. ponderosa* only reached 70 m from the plantation, with densities significantly lower (250 trees ha⁻¹ with age: > 4 years; Orellana *et al.* 2004; Sarasola *et al.* 2006) All studied *P. menziessi* plantations presented recruitment of new seedlings outside their edges, and the greatest were in downwind areas. Both *P. ponderosa* and *P. menziessi* started to produce descendants at age 17-18 years (Sarasola *et al.* 2006) while other studies reported that recruitment was observed in 10-15 year old plantations (Orellana and Raffaele 2008).

Finally, we have to consider conifer invasions in humid areas with *A. chilensis*- *N. dombeyi* forests. It is important to note that currently very few plantations are installed in these humid areas. The first massive introduction of conifers in Patagonia was (more than 40 species) in the Victoria Island (3700 ha, in the Nahuel Huapi Lake), 80 years ago. Simberloff *et al.* (2002) evaluated in meticulous way what happened with wildings of introduced species. Eighty years after species introduction, *P. menziessi* and *Juniperus sp.* were the more extended species reaching up to 1 km in distance from the original plantation, but wilding densities were very low. Higher densities were in areas close to plantations (150 to 250 trees ha⁻¹ including all species).

2. Which causes can be affecting positively or negatively these processes?

Considering species characteristics, the three studied conifer species present a certain capacity to invade considering only their biological traits identified as good predictors of invasiveness (Rejmánek and Richardson 1996). From the

literature and our data, the invasiveness gradient started with *P. ponderosa*, as the species with lower invasive character, followed by *P. menziessi* and finally, *P. contorta*, the species with highest invasive potential (Sarasola *et al.* 2006)

In addition to species characteristics, we have to consider ecosystems characteristics, such as the degree of disturbance and vegetation cover. Forests and shrubland, areas with highest cover (areas relatively undisturbed) did not present introduced trees, while the presence of recruitment was associated to areas with lowest cover (Chauchard *et al.* 1988; Simberloff *et al.* 2002; Sarasola *et al.* 2006). The more colonized areas were natural or anthropogenic gaps, roads, trails, roadsides and old roads, while natural patches of high tree density or understory vegetation presented the lowest recruitment. Presence of natural and anthropogenic gaps, roads, land use changes, the presence of cattle and exotic deer, abandoned pastures, logging and fires are frequent in Patagonia and are direct or indirectly related to the decrease in vegetation cover. Therefore, all these medium-scale disturbances could be facilitating the start of local invasions. In addition, manipulative experiments simulating microdisturbances showed higher germination and lower seed predation than seeds without microdisturbance. These results presented similar patterns in steppe, shrubland and *A. chilensis* and *N. dombeyii* forests (Mauro Sarasola unpublished data).

Experiments carried out in steppes, shrublands, *A. chilensis* and *N. dombeyii* forests demonstrated high rates of conifer seed predation, reaching in some cases more than 90% of the seeds (Nuñez *et al.* 2008). Shrublands and xeric *A. chilensis* forests presented the lowest predation values (Mauro Sarasola unpublished data). In addition, predation was higher in zones far away than close to the plantation installed in *Nothofagus sp.* areas (Nuñez *et al.* 2008). Post dispersal seed predation could be reducing or limiting invasion processes in Patagonia.

On the other hand, in the steppe, the susceptibility to invasion will be directly associated to pasture use. Sites where cattle were excluded presented more recruitment while sites more intensely grazed did not have wildings (Sarasola *et al.* 2006). In contrast, in forest areas, the presence of cattle should be facilitating recruitment by means of reduction of native vegetation cover and generation of trails and microdisturbances.

Another topic to be considered as influencing invasion processes is the presence or absence of symbiont organisms. In *A. chilensis* and *N. dombeyii* forests in Victoria Island the absence of specific micorrhizas far away from plantations would be an explanation of the low colonization by pines (Nuñez *et al.* 2009). In the steppe, micorrhiza inoculums were found up to 400 m from *P. ponderosa* plantations (Salgado-Salomón *et al.* 2007). Presence of micorrhiza inoculums could be crucial for the success or failure of pine colonization because these introduced conifers are ectomycorrhizal, and these fungi are generally highly host-specific.

Considering the influence of fire, we can mention that conifers and particularly pines are well adapted to fire and that this type of disturbance is common in Patagonia. In a survey of 18 burned plantations, most of them presented high recruitment and recolonization within the same plantation area but they did not present recruitment outside plantation edges, except two cases in humid sites (Raffaele and Nilsson 2006).

Finally, as mentioned in the previous section, wind appears to be important in invasion processes in Patagonia. In this region, the direction of winds is mostly NW/W to E. Conifer recruitment from plantation was strongly influenced by this factor, presenting a higher degree of recruitment (distance from plantation and seedling density) in downwind sides (Sarasola *et al.* 2006).

MANAGEMENT IMPLICATIONS AND GENERAL CONCLUSIONS: IS SUSTAINABILITY AN ACHIEVABLE GOAL?

From our results, we can conclude that, at the current developmental stage of forestry activity in Patagonia region, the negative environmental impact – considering biodiversity losses, alteration of water balance and soil pH, as well invasion processes – is very low or even nil, the positive impacts probably being – economic and social – higher leading to a positive balance as a whole. However, we recognize that potential negative impacts, whose magnitude will depend on several aspects, could increase in the future in relation to the expansion of the forested areas. With the available information we can then formulate prescriptions and management strategies in order to guarantee the long term sustainability of the activity. In this regard, we have the opportunity of developing a sustainable production activity from the very beginning.

As general prescriptions we can suggest that, from the stand point of biodiversity conservation, landscapes comprising mosaics of native vegetation and forest plantations are more desirable from a conservation perspective than other land uses that are more structurally simplified, like agriculture (Moore and Allen 1999), or intensive livestock grazing (Lantschner and Rusch 2008). Thus, when analyzing the impact of plantation forestry on biodiversity, the ecological context of planted forest development must be considered. The definition of management objectives linked to sustainability, considering endangered and functional keystone species, and the integrated analysis of different spatial scales are important finding a balance between intensive land use and biodiversity conservation.

Considering water resources consumption and water balance alteration, we can conclude that in the humid portion of the precipitation gradient, forest plantations would not constitute a problem. However, in the other extreme of the gradient, pine plantations can use all the available water resources, decreasing the yields to external economies. In this area, a reduction of leaf area (through intense thinning, or installation of low density plantations) is recommended if excess water resources have to be used for other purposes. Sparse plantations are also desirable for biodiversity conservation and forage production in the understory.

From invasion results we can conclude that *P. contorta* in steppe and *P. menziessi* in xeric *Austrocedrus* forests present auto self-seeded wildings with invasive dynamics, while *P. ponderosa* is not showing an invasive behavior in either system yet. Continuous monitoring following maturity of plantations and more studies are necessary to define the actual invasion risk for this species because may be invasive at a lower level. The observed rates of recruitment indicate that at the present time, conifer invasion is an incipient process focalized in some areas. At a regional scale, an invasion risk exists but it is low. This “low risk” appreciation is based on two main aspects: total planted area in the region is very low and more than 80% of this area is occupied by plantations of *P. ponderosa*, species with the lowest invasive behaviour. In addition, in steppes where cattle raising are the main productive activity, grazing may be controlling invasion, whereas in native forest areas legal regulations constrain the expansion of the planted areas. Finally, unlike two other strongly invasive species in Patagonia as *Rosa rubiginosa* (rosa mosqueta – sweet briar) or *Cytisus scoparius* (retama – Common broom), rates of advance of conifers recruitment are relatively low. However, probability of invasion success could increase with the increment in planted area within the steppe and aging of current plantations (95% of them are younger than 35 years), both factors contributing to increase propagule pressure. In addition, particular diseases affecting native forests, such as the “Cypress disease” and/or or events of tree mortality due to drought, as occurred in *Nothofagus* forests, could increment the opportunities for invasions. Finally, climate change would also contribute to conifer invasions (Richardson and

Bond 1991), through differential responses of native vs exotic species to drought and high temperature. Therefore, in Patagonia we still are in an ideal stage for implementing prevention plans in order to avoid or mitigate conifer invasion success. Prevention is the least expensive strategy for invasion control. Once installed and highly spread, eradicating invasions is a very expensive and frequently impossible task. We currently have an opportunity of learning the lesson of other regions where conifer invasions have succeeded. In this regard, an important factor contributing positively to the implementation of effective control or prevention plans is the predictability of the conifer invasion process.

Returning to the question if sustainability is an achievable goal for forestry, we suggest that the response is “yes”, however, it is not intrinsically sustainable but depends on where and how the plantations are installed and managed. The current unmanaged plantations are not good examples of what has to be done. The current low impact is linked to the few hectares occupied by plantations and not because they were well managed. Future sustainability will be guaranteed only if current plantations begin to be managed, and future plantations are well planned from the beginning. In this regard, both the State –through legal regulations and control- and private producers –by respecting the law- have the responsibility of take care of the natural resources for the welfare of current and future human generations.

ACKNOWLEDGEMENTS

Special thanks to Peyrou C, Lantschner V, Licata J, Bond B, Rivero D, Weigandt M, Varela S, Caballé G who helped us in the field and laboratory work. We also thank the economic support of Consejo Nacional de Investigaciones Científicas y Técnicas (CONICET, Argentina), Instituto Nacional de Tecnología Agropecuaria (INTA, Argentina), Secretaría de Agricultura, Ganadería, Pesca y Alimentos (SAGPyA, Argentina), Secretaría de Ambiente y Desarrollo Sustentable (SAyDS, Argentina), Agencia Nacional de Promoción Científica y Tecnológica (Argentina), National Science Foundation (NSF, USA).

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