Programa de Doctorado en Ecología, Conservación y Restauración de Ecosistemas (D330)

Definición de áreas prioritarias para la restauración forestal en la Cordillera de la Costa de la Región de Los Ríos, Chile

Memoria presentada para optar al grado de Doctor
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A mis hermosos y maravillosos Simona y Emilio

A mis padres y hermana

A mi abuelita Herminia

Se trata entonces que podamos ejercer
la libertad de ser actor y proponer
sobre el camino que queramos emprender
para que el hombre libre pueda florecer.

Mientras un niño por la calle pida un diez
y por escuela tenga el hambre y el neoprén
la libertad seguirá siendo de papel
sueño fecundo que no acaba de nacer.

Mano con mano vamos juntos de una vez
para botar todos unidos la pared
el egoísmo impuesto que no deja ver
que tu problema es mío, es nuestro y es de aquél.

Ya no podemos esperar que alguna vez
haga un milagro lo que debamos hacer
tomar conciencia del derecho y del deber
para que el hombre libre pueda florecer.

"Palabras de nuestro tiempo" (extracto), de Andrés Márquez - Illapu

AGRADECIMIENTOS

Agradecimientos

Y aquí estoy. En mi rincón de la oficina-despacho en que estuve mis primeros dos años de doctorado definiendo el lío en que me quería meter…y ahora he regresado a cerrar el círculo (que por momentos parecía una línea). En aquellos inicios compartí oficina, momentos e historias con tantas y tantos compañeros y compañeras que hoy andan por el mundo siguiendo sus caminos, esos mismos que nuevos estudiantes en mi entorno están ahora comenzando, ocupando escritorios que han soportado el trabajo y perseverancia de tantos otros. Este es un camino difícil, como todo en la vida supongo, y lo decidí recorrer luego de largos años de reflexión. Cuando acabé mi licenciatura allí en la Universidad Austral (UACh) en la hermosa Valdivia, no pensaba en la ciencia como mi camino. Luego de varios años trabajando en la UACh como asistente de investigación del profesor Antonio Lara, decidí estudiar una Maestría con Mario González en el ECOSUR, en San Cristóbal de las Casas, en Chiapas. Fue una de las épocas más maravillosas de mi vida, gracias a la cual conocí a mi Simona. Luego de ese posgrado regresé a Valdivia, y tuve la oportunidad de participar en un proyecto muy lindo que realizaba la ONG Forestales por el Bosque Nativo en el sur de Chile. Gracias a este proyecto conocí en mayor profundidad la realidad de los campesinos forestales y de nuestros bosques nativos, de la degradación y conservación de nuestros ecosistemas forestales, de la diversidad de estas problemáticas, pero al mismo tiempo de los elementos comunes que las caracterizan, tanto a nivel nacional como regional e incluso global. En ese contexto me preguntaba por el real impacto del programa de asistencia técnica forestal a campesinos que por tantos años realiza esta emblemática ONG, me preguntaba por el impacto que nuestras actividades realmente tenían no sólo en la calidad de vida de los campesinos, sino que también en los bosques y en el contexto en que estábamos interviniendo. Fue un período de profundas reflexiones y bueno, me dí cuenta que la ciencia podría darme algunas herramientas para responder estas preguntas y, lo más importante, para hacer otras. Es decir, siempre he visto la ciencia como un medio y no como un fin, como muchas veces he escuchado decir por ahí. Y así es como me metí voluntariamente en este lío llamado doctorado.

No puedo dejar de pensar en tantos momentos vividos, pero fundamentalmente, en mi inspiración. En mi mujer Simona, y en nuestro tesoro Emilio. Este período dejé de vivir tantos momentos junto a ellos que a veces sentí que no tenía derecho a su paciencia, a su complicidad, a su infinita comprensión, a su constante apoyo. No puedo imaginar mi vida sin ellos. ¡CUANTO LOS AMO!

Mi director de tesis, José María, me dió el espacio necesario para desarrollar libremente mi proyecto de investigación, con aquellas ideas que fueron incubadas en mis años junto a la ONG de Valdivia. Tuvo la suficiente confianza en mí como para dejarme partir a Trento durante estos últimos dos años a continuar con mis andanzas, respetando en todo momento mis decisiones y sueños, y para reintegrarme a la atmósfera terrestre cada vez que fue necesario. No puedo dejar de agradecer su apoyo en momentos clave, como aquellos cursos de SIG y de *R* en Granada, o para participar en aquel congreso en Colombia, o más aún, cuando fui a Chile a realizar mi trabajo de campo para mi tesis doctoral y no tenía financiamiento para pagar mi viaje. Le agradezco también por su orientación, en especial en estos últimos meses de locura entre capítulos generales, definición de Tribunal de Tesis, cansancio acumulado, preocupaciones y nerviosismo por las certezas que terminan y por la incertidumbre que vendrá.

Esta tesis, este camino, no habría sido posible sin el apoyo de mi gran maestro y amigo Luis Cayuela. Es difícil escribir mi gratitud hacia ti Luis, como académico y como el gran amigo que eres. Sin tu orientación y apoyo todo habría sido aún más difícil. He aprendido muchísimo en esta etapa, y en gran parte ha sido por tu culpa!. Te estimo y admiro Luis. Agradezco también a Davide Geneletti, quien me recibió en su grupo de investigación en la Universidad de Trento durante esta última etapa de mi postgrado. Aprendí y disfruté muchísimo de mi estancia en la increíble ciudad de Trento. Gracias también a mi amigo Francesco Orsi en la domesticación de los software necesarios para mis análisis. Fueron muchas las conversaciones sobre la vida que habitualmente teníamos al regresar a casa caminando desde la Facultad luego de cada jornada.

Desde que dejé Santiago por el año 1995 me he adaptado a las circunstancias de muchos lugares y países. He conocido gente admirable y grandes amigos con los que hasta el día de hoy mantenemos contacto. En cierto modo, siento que con este doctorado estoy cerrando una larga etapa, y una avalancha de momentos, de sensaciones y de caras se me vienen a la memoria. Mis queridos amigos de Santiago de Chile, de Valdivia, de México, de España, de Italia. De México, además de mi maestro y querido amigo Mario González, pienso en Sofía y Raúl, el pinche José Manuel, Bárbara, Clau, Nadia y Paco Trash Metal. De España…uf….tanta gente, tantos queridos amigos. Sergio de Frutos, aquel experto e irreverente "científico"-comediante que siempre me recibió en su casa en mis visitas a España desde Italia, al igual que mi gran amigo Manuel Loro. Eres un gran tipo Manu. Un grande. Muchas gracias a tu familia por recibirme como uno de vosotros. Siempre estaré agradecido de tu apoyo y amistad. Muchas gracias a tu padre (Don Manu) a tu madre (Sra. Mercedes) a tu hermana Carmen y tu cuñado, el Guille. En Italia también tuve la suerte de hacer muchos amigos con los cuales viví y disfruté el crecimiento de mi pequeño príncipe Emilio. Andrea, Michela y sus pequeños Beatrice y Federico. Valeria, Gabrielle y sus bebés Marco y Carla. Francesca, Dario y la piccola Amelia. Amigos con los cuales vivimos momentos inolvidables, incluyendo algunos partidos de Chile en el mundial de Brasil!.

Para finalizar, quisiera agradecer a mis padres y hermana por su incondicional apoyo y fé en todas las decisiones que he tomado en mi vida. He llegado hasta aquí por su amor y empuje, y lo más importante, por los valores y principios que me han transmitido. Sé que no ha sido fácil para ustedes limitar nuestro contacto a llamadas por Skype (que en mi época de licenciatura eran desde teléfonos públicos en las lluviosas y frías noches de Valdivia), y a pesar de mi "hoja de ruta" por aquí y por allá, siempre han estado conmigo.

No puedo dejar de pensar en mi abuelita Herminia, cada día. No logré regresar a tiempo para despedirme por última vez de ella, de una de las personas más importantes de mi vida. Estaba muy cansada luego de casi 98 años en esta Tierra. Se fue y yo no estaba ahí para despedirme de ella. Pero de algún modo siento que estás acá abuelita, a mi lado mientras escribo estas palabras.

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Regeneración natural de *Dasyphyllum diacanthoides* (trevo).
Todas las fotografías incluidas en esta Tesis Doctoral pertenecen a C. Zamorano-Elgueta.

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Bosques nativos, praderas y pequeñas propiedades, Cordillera de la Costa, Región de Los Ríos.

RESUMEN GENERAL

ABSTRACT

Resumen

Los disturbios de origen antrópico están modificando rápidamente el paisaje y los ecosistemas, y aún existen vacíos importantes en nuestra comprensión del impacto de estos cambios en la ecología del paisaje. Aunque los impactos de los disturbios de origen antrópico en los ecosistemas forestales han sido ampliamente estudiados, se ha prestado menos atención al estudio de cómo la regeneración forestal a nivel de comunidad responde a estas alteraciones, y cómo estos efectos cambian según los factores sociales y ambientales que pueden influenciar el uso del bosque a escala de paisaje. Además, hasta el momento, muy pocos estudios se han enfocado en la expansión y en la configuración espacial de las plantaciones forestales de especies exóticas y en su papel en la dinámica de cambios en la cobertura del suelo. La restauración forestal puede tener una creciente importancia para revertir o mitigar distintos procesos implicados en la destrucción o degradación del bosque. La identificación de áreas prioritarias podría aumentar la eficiencia y el impacto de los recursos disponibles para el diseño, la planificación y el establecimiento de programas de restauración forestal. Sin embargo, el desarrollo de métodos de priorización de restauración forestal para el mantenimiento de la biodiversidad y la provisión de servicios ecosistémicos a escala de paisaje ha sido menos tratado en la literatura científica.

El **objetivo general** de esta Tesis Doctoral es identificar áreas prioritarias para la restauración forestal con el fin de conservar la biodiversidad y la provisión de los servicios ecosistémicos en la Cordillera de la Costa de la Región de Los Ríos, Chile.

Para ello, primero analizamos la influencia de la ganadería en la regeneración de la araucaria (*Araucaria araucana*), una conífera amenazada de los bosques templados de Chile y Argentina (**Capítulo 2**). Utilizamos el número de excrementos como un indicador de la actividad ganadera (el índice de intensidad ganadera, CAI, por sus siglas en inglés). La regeneración fue analizada como una función del CAI, del régimen de propiedad, del área de estudio y de la densidad de adultos. En general, se registró una influencia exponencial negativa del CAI en todas las variables respuesta. En pequeñas propiedades, se detectó una rápida disminución de la regeneración a bajas intensidades de carga ganadera, mientras que en propiedades de empresas forestales este efecto negativo fue mucho menos marcado. El CAI tuvo también efectos cualitativos en la regeneración de la araucaria al aumentar la proporción de regeneración de origen sexual, lo que podría afectar a la conservación de la diversidad genética de la especie en el largo plazo.

Con el fin de investigar si los efectos del disturbio antrópico afectaban también a las comunidades de regeneración en su conjunto, en el siguiente capítulo analizamos los efectos de la ganadería y de la tala selectiva en la composición de la comunidad de regeneración de plantas leñosas en bosques siempreverdes, considerando para ello el estado sucesional del

bosque y el régimen de propiedad (**Capítulo 3**). Nuestros resultados revelan que la ganadería tiene un mayor efecto negativo en la regeneración forestal que la tala selectiva, en especial en pequeñas propiedades y en bosques maduros, los cuales presentaron una mayor sensibilidad a los disturbios evaluados. Los bosques sin alteraciones o sólo expuestos a tala selectiva podrían ser dominados por especies de sucesión tardía como *Saxegothaea conspicua*, *Aextoxicon punctatum* y *Laureliopsis philippiana*. En cambio, la continua presión por ganadería y tala selectiva podría impedir el establecimiento de especies tolerantes y semitolerantes a la sombra y favorecer una composición dominada por especies como *Amomyrtus luma*, *A. meli* y *Gevuina avellana*. Estos resultados confirman que los disturbios de origen antrópico evaluados, en especial la ganadería, pueden disminuir, limitar o impedir el reclutamiento de especies leñosas, con impactos desconocidos en las propiedades funcionales de los ecosistemas forestales.

Una vez analizados los impactos de las actividades humanas sobre la comunidad de regeneración, analizamos la dinámica de cambios de la cobertura del suelo bajo la hipótesis de que las plantaciones forestales de especies exóticas han causado la mayor transformación en la superficie de los bosques templados en el sur de Chile en las últimas tres décadas (**Capítulo 4**). Usamos para ello imágenes Landsat de 1985 (TM), 1999 (ETM+) y 2011 (TM), y seleccionamos diversas variables para el análisis de los cambios del paisaje. Nuestros resultados indican que los mayores cambios se generaron principalmente como una conversión dinámica entre bosques, plantaciones y matorrales. Durante el período analizado la superficie de plantaciones aumentó un 168%, con una tasa anual del 3.8%, principalmente por la sustitución del bosque nativo y matorrales. Se registró una pérdida bruta de bosques del 30%, pero una pérdida neta de sólo el 5.1% de la superficie inicial, con una tasa anual de deforestación del 0.2%. La diferencia entre la pérdida bruta y la neta de los bosques se debe a la conversión de matorrales y de áreas agrícolas y pastizales a bosques secundarios por la regeneración natural de los bosques. En general, nuestros resultados indican la expansión y compactación de las plantaciones de especies forestales exóticas y un incremento en la pérdida y fragmentación de los bosques nativos, en particular para el período 1985-1999. Mientras la deforestación afectó tanto a los bosques adultos como a los secundarios, los bosques regenerados corresponden exclusivamente a estos últimos. Ello puede influir en la capacidad de los bosques para proveer servicios ecosistémicos, incluyendo aquellos relacionados con el suelo y el agua, con impactos que van más allá del uso inmediato de la tierra.

Por último (**Capítulo 5**), identificamos áreas prioritarias para la restauración forestal con el fin de conservar la biodiversidad y la provisión de servicios ecosistémicos. Utilizamos un enfoque multicriterio para evaluar la idoneidad ecológica y la factibilidad socioeconómica de la restauración de bosques. Los bosques degradados fueron definidos por la evidencia empírica de impactos en la regeneración forestal por la ganadería y la tala selectiva detectada (**Capítulos 2 y 3)**, mientras que las áreas deforestadas fueron definidas a partir de una imagen Landsat (TM) del año 2011 (**Capítulo 4**). El área prioritaria de restauración fue definida de acuerdo a las áreas de mayor idoneidad y factibilidad según diferentes perspectivas (por ejemplo, con una orientación en la biodiversidad). Las áreas prioritarias de restauración se distribuyeron a través de toda el área de estudio pero se concentraron en las zonas Central y Este de la región, donde presentaron una distribución continua. Las áreas prioritarias para la restauración forestal tienen una alta biodiversidad y una severa erosión potenciales, y están localizadas en cuencas hidrológicas caracterizadas por bajos coeficientes de escorrentía, áreas de elevada accesibilidad y expuestas a una menor presión productiva. Su superficie total se estimó en un 10.7% del área de estudio, de la cual el 7.4% correspondieron a áreas deforestadas y el 3.3% a bosques degradados en zonas cercanas a los bosques bien conservados, en propiedades de empresas forestales certificadas por el *Forest Stewardship Council*, en pequeñas propiedades sustentables y en áreas protegidas.

El enfoque de análisis propuesto por esta Tesis Doctoral contribuye a la comprensión de la influencia variable de los disturbios de origen antrópico en la regeneración forestal a escala de paisaje. Estos resultados podrían apoyar el diseño de políticas y acciones de restauración y conservación, las cuales deberían primero concentrarse en limitar o eliminar los principales factores de disturbio y luego proteger y recuperar las especies más sensibles a estas alteraciones. Además, el análisis de la dinámica de cambio de la cobertura del suelo y del papel de las plantaciones forestales de especies exóticas podría apoyar el desarrollo de estrategias de restauración y conservación de bosques nativos, las cuales son actualmente obligatorias para las empresas forestales de la Región para la certificación forestal. Este enfoque y los resultados generados no sólo permitirán a los expertos y planificadores identificar dónde restaurar, con el fin de aumentar los valores ecológicos de un territorio, sino también definir la factibilidad socioeconómica para implementar acciones de restauración en el mediano y largo plazo, incluyendo tanto las áreas deforestadas como los bosques degradados.

Abstract

Human-induced disturbances are rapidly changing landscapes and ecosystems, yet significant gaps in our understanding of the spatial ecology of these changes remain. Although the impacts of human-induced disturbances on forests have been extensively studied, less attention has been paid to understanding how tree regeneration at the community level responds to such disturbances and how these effects change according to major social and environmental factors that can influence forest use at the landscape scale. In addition, to date few studies have focused on the expansion and spatial configuration of exotic tree plantations and in their role on the dynamics of regional land cover change. Forest restoration can play an increasingly important role to reverse or mitigate these processes. Moreover, identifying priority areas would increase the efficiency and impact of available resources to design, plan and establish forest restoration programs. However, the problem of developing methods to identify priority areas for maintaining and enhancing biodiversity and ecosystem services through forest restoration at the landscape-scale level is less frequently addressed in the scientific literature.

The **main goal** of this PhD Thesis is to identify priority areas for forest restoration in order to maintain and enhance biodiversity and the provision of ecosystem services in the Coastal Range of Región de Los Ríos, Chile.

To achieve this goal, we first analyzed the influence of cattle on the regeneration of monkey puzzle tree (*Araucaria araucana*), an endangered conifer of the temperate forests of Chile and Argentina (**Chapter 2**). We used the number of cattle dung pats as a surrogate of cattle activity (the cattle intensity index, CAI). Regeneration ot the monkey puzzle tree was analyzed as a function of the CAI, land tenure regime, the study site, and the density of parent trees. Overall, there was a negative exponential influence of the CAI on all response variables. In small landowner forests, even low cattle intensities caused regeneration to drop rapidly to zero, whereas in plots owned by timber companies regeneration decreased smoothly as the CAI increased. The CAI also affected regeneration of the monkey puzzle tree qualitatively by increasing the regeneration by root suckering, which may lead to problems of genetic drift in the long-term.

To further investigate whether human-induced changes might also affect biotic communities as a whole, in the next chapter we analyzed the effects of cattle grazing and selective logging on the composition of tree regeneration communities in evergreen forest in southern Chile, considering these effects in relation to forest successional stage and land tenure regime (**Chapter 3**). Our results revealed that cattle had a more negative effect on forest regeneration than selective logging, especially in small properties and old-growth forests, which appear to be more sensitive to human-induced disturbances. Undisturbed old-growth forests or forests associated only with selective logging would be dominated by late-successional species like *Saxegothaea conspicua*, *Aextoxicon punctatum* and *Laureliopsis philippiana*. Instead, the occurrence of cattle and selective logging could prevent the establishment of these shade-tolerant and shade-semitolerant species and favour a composition dominated by *Amomyrtus luma*, *A. meli* and *Gevuina avellana*. These results confirmed that human-induced disturbances, particularly cattle, can diminish, damage or prevent the recruitment of tree species, which could generate unknown impacts on functional ecosystem properties.

After analyzing human-induced impacts on regeneration communities, we moved on to the landscape scale and analyzed the dynamics of land cover change under the hypothesis that exotic tree plantations have caused a major transformation of temperate forest cover in southern Chile in the last three decades (**Chapter 4**). To achieve this, we used Landsat scenes taken in 1985 (TM), 1999 (ETM+), and 2011 (TM), and selected landscapes indices. Our results showed that major changes took place as mainly a dynamic conversion among forest, exotic tree plantation and shrubland. During the studied time span, the area covered by exotic tree plantations increased by 168% at an annual rate of 3.8%, mostly at the expense of native forest and shrubland. There was a total gross loss of 30% of native forest, but a net loss of only 5.1% of its initial cover, at an average annual deforestation rate of 0.2%. The difference between gross and net loss of native forest is mostly the result of conversion of shrubland and agricultural and pasture land to secondary forest following natural regeneration. Overall, the observed trends indicate expansion and compactness of exotic tree plantations, and increasing native forest loss and fragmentation, particularly during the 1985-1999 period. Whereas forest loss include both old-growth and secondary forests, native forest regenerated after natural succession correspond to the latter. This can influence the native forest capacity to provide ecosystem services, including those related to soil and water. These alterations will affect humans in ways that go beyond the immediate land-use situation.

Finally, and based on the results of previous chapters, we focused on identifying priority areas for forest restoration for maintaining and enhancing biodiversity and the provision of ecosystem services (**Chapter 5**). We used a multicriteria approach to assess the ecological suitability and socioeconomic feasibility of forest restoration. Forest degradation was defined based on empirical evidence of alterations in forest regeneration by cattle grazing and selective logging (**Chapter 2 and 3**), whereas deforested areas were defined using a Landsat image for the year 2011 (TM, **Chapter 4**). The area to be restored was defined according to the best suitability and feasibility areas for forest restoration according to different perspectives, e.g. biodiversity oriented. The priority areas for forest restoration were distributed across the entire study area, but concentrated in the central and eastern parts of the region, where they showed a continuous distribution. The priority areas for forest restoration have high potential biodiversity and severe potential erosion, located in watersheds characterized by low runoff coefficients, in more accessible areas, and exposed to low pressure on forest. The total amount of priority areas for restoration accounted for 10.7% of the study area, of which 7.4% corresponded to deforested areas and 3.3% to degraded forests near well-conserved forests, in land owned by forest companies certified by the Forest Stewardship Council (FSC, 5.85%), in sustainable small properties (3.25%), and protected areas (0.87%).

Our analytical approach contributes to the understanding of the differential influence of human-induced disturbances on the tree regeneration community at the landscape scale. It can inform restoration and conservation policies and actions, which should focus first on addressing the main disturbance factors and then on developing strategies to conserve the most sensitive species to such disturbances. In addition, understanding the dynamics of land cover change and the role of exotic tree plantations will help to restoration and conservation strategies of native forest, which is now mandatory for forest companies in the region to obtain timber certification. This approach and the results will not only allow practitioners to understand where to restore in order to enhance the ecological values of a region, but also to define the socioeconomic feasibility of restoration activities in the medium and long term, including deforested areas and degraded forests.

Bosque maduro siempreverde, Cordillera de la Costa, Región de Los Ríos.

CAPÍTULO 1

Introducción general

Introducción general

La degradación de los bosques y sus impactos en la biodiversidad

La deforestación y la degradación de los bosques son reconocidas como unas de las principales causas de pérdida de biodiversidad y de otros cambios ambientales a nivel global (Sasaki & Putz 2009, Laestadius et al. 2011), y la biodiversidad constituye la base de los servicios ecosistémicos de los cuales depende estrechamente el bienestar humano (MEA 2005). Se ha estimado que más de dos billones de hectáreas de bosques han sido completamente deforestadas o degradadas en todo el mundo (Laestadius et al. 2011). Cerca de 130 millones de ha fueron deforestadas durante la última década, de las cuales 40 millones eran de bosque primario (**Figura 1.1**, FAO 2010). A nivel regional, Sudamérica presenta la mayor pérdida neta de bosques entre los años 2000 y 2010 – cerca de 4 millones de ha al año – seguida por África, con una pérdida anual de bosques de 3.4 millones de ha (FAO 2010).

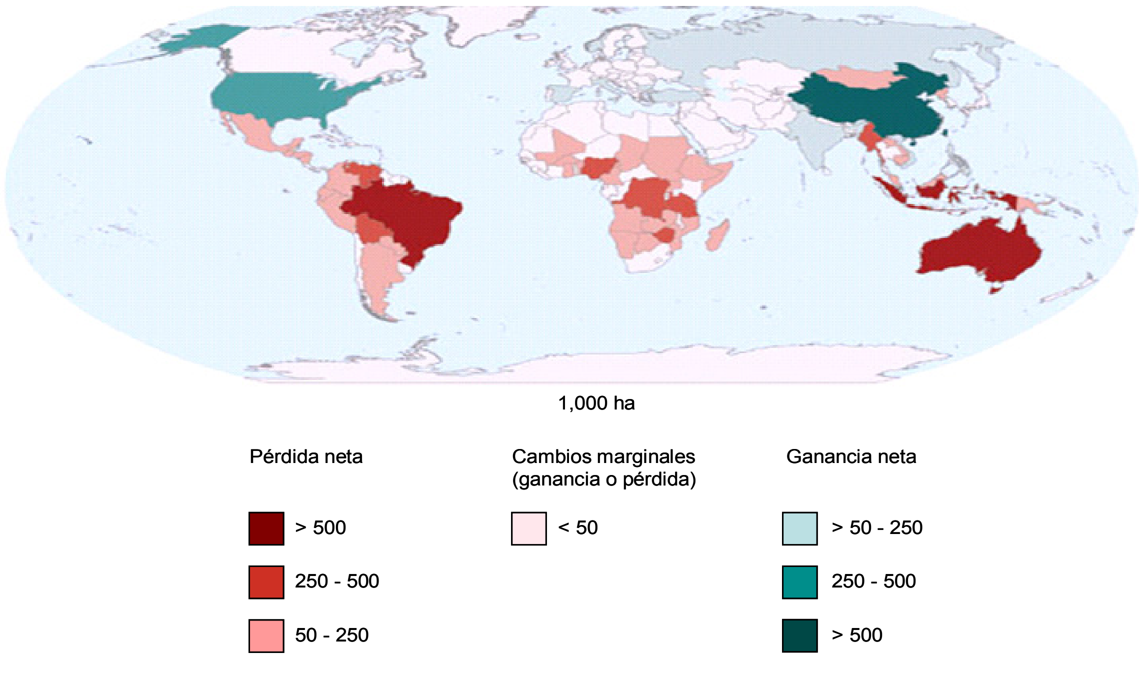

Figura 1.1. Cambio anual de la superficie forestal por país para el período 2005-2010. Fuente: FAO 2010.

En Chile se ha estimado una pérdida de bosque nativo de cerca de 8 millones de ha desde el inicio de la colonización europea en el siglo XVI (es decir, un 44% de la superficie forestal existente antes de la colonización, Lara et al. 1999), principalmente por la transformación de áreas para el uso agrícola y ganadero y por la demanda de leña de la naciente industria minera (Camus 2006). Además de la disminución de la superficie de los bosques nativos, también se alteró su estructura y composición, destacando la alta proporción de bosques secundarios o renovales que superan el 50% de los bosques remanentes del país y que indican la intensidad del impacto antrópico sobre ellos (Lara et al. 1999). Se han determinado recientemente tasas de deforestación para diferentes regiones del país. Así, en la Cordillera de la Costa del sur de la VII Región y norte de la VIII Región, Echeverría et al. (2006) estimaron en un 67% la disminución de la cobertura de bosques entre los años 1975 y 2000. Por otra parte, en la zona central de Chile Schulz et al. (2010) determinaron una pérdida del 42% de la superficie de bosques secos para el período 1975-2008. En la Cordillera de la Costa de la Región de Los Ríos, donde se ha realizado esta Tesis Doctoral, aún no existen estimaciones de las tasas de deforestación ni de procesos de fragmentación de los bosques nativos

A diferencia de la deforestación, la cual es definida como la sustitución permanente de bosques por otros usos, la degradación implica el mantenimiento de una superficie forestal pero con una capacidad reducida del ecosistema para generar bienes y servicios (MEA 2005, **Figura 1.2A**). La degradación de los bosques por disturbios de baja intensidad (ganadería, tala selectiva e invasión de especies exóticas, entre otros), modifica la composición, diversidad y funcionamiento de la comunidad de organismos (Chapin III et al. 1998, Cadotte et al. 2011, Baraloto et al. 2012). Estas alteraciones afectan negativamente a millones de personas que dependen total o parcialmente de los servicios generados por los ecosistemas forestales a escala local, regional o global (FAO 2011). Estos servicios incluyen el almacenamiento de carbono, la regulación del clima, la provisión de agua y la fertilidad de suelos, entre muchos otros (MEA 2005). La degradación de los bosques es, de este modo, un problema ecológico, social y económico crítico (Simula & Mansur 2011). Los procesos de degradación forestal producidos por disturbios de baja intensidad como la ganadería (**Figura 1.2B**) y la tala selectiva (**Figura 1.2C**) son más difíciles de detectar que aquellos generados, por ejemplo, por una explotación forestal excesiva, incluso mediante observaciones en el campo o terreno (FAO 2009).

Determinar el impacto de estas alteraciones en los ecosistemas permitiría detectar de forma temprana los procesos de degradación forestal, disminuyendo notablemente el costo de las actuaciones de restauración y el tiempo necesario para revertir sus consecuencias. La ganadería y la tala selectiva representan dos de los factores de alteración más comunes en los bosques a nivel global (Belsky & Blumenthal 1997, Ramírez-Marcial et al. 2001, Stern et al. 2002, Timmins 2002, Baraloto et al. 2012, Clark & Covey 2012). Estas perturbaciones generan distintos impactos dependiendo del tipo de ecosistema, tipo y cantidad de ganado y cantidad de biomasa extraída, entre otros factores (Vázquez 2002, Clark & Covey 2012). Dichos impactos incluyen la alteración de la estructura y la composición de la vegetación, la reducción de la tasa de infiltración de agua, la compactación y la erosión de los suelos (Belsky & Blumenthal 1997) e incluso la alteración de la polinización (Vázquez 2002), lo que puede derivar en profundos impactos en la funcionalidad ecosistémica (Chapin III et al. 1998, Cadotte et al. 2011). Los ecosistemas con más biodiversidad son comúnmente más estables y eficientes debido a la presencia de una mayor diversidad funcional para el flujo de la energía y el reciclaje de los nutrientes (Tilman 1997, Sekercioglu 2010).

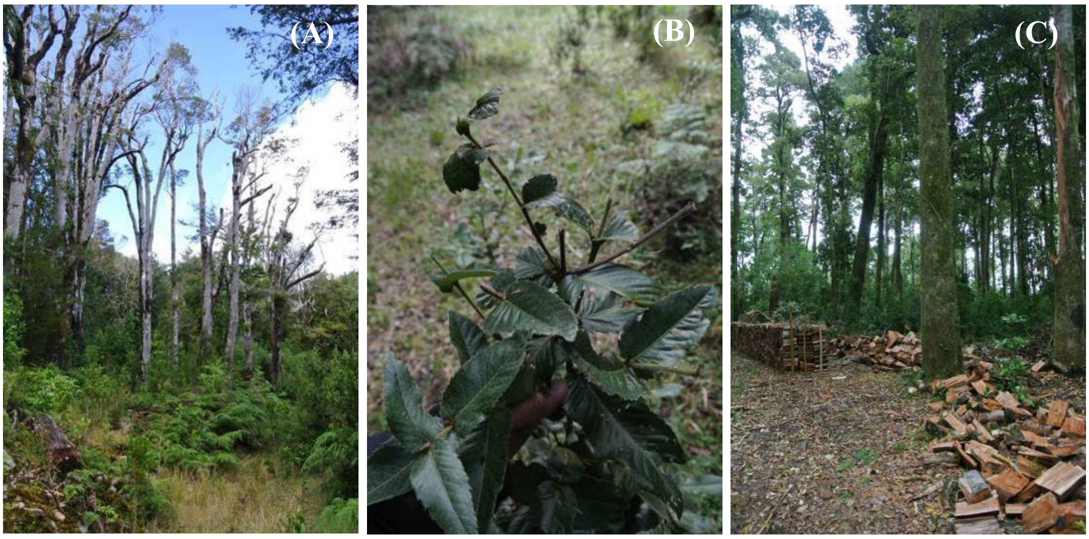

Figura 1.2. Disturbios de origen antrópico en bosques nativos siempreverdes de la Cordillera de la Costa del sur de Chile. De izquierda a derecha: **(A)** bosque degradado por extracción excesiva de madera, **(B)** regeneración dañada por ramoneo de ganado y **(C)** producción de leña en un bosque maduro.

En Sudamérica, y particularmente en Chile, los escasos estudios que han evaluado el impacto de la ganadería y de otros mamíferos introducidos en los bosques templados se han concentrado en la regeneración de algunas especies de árboles como *Nothofagus dombeyi* (Veblen et al. 1989, 1992) y *Austrocedrus chilensis* (Veblen et al. 1992, Relva & Veblen 1998, Relva & Sancholuz 2000). No obstante, se ha prestado menos atención al impacto que estos factores de alteración pueden tener en la regeneración a nivel de comunidad en bosques sometidos a una mayor presión productiva. Ello es un problema relevante porque la creciente presión generada por las actividades humanas está rápidamente modificando el paisaje y la capacidad de los ecosistemas de generar bienes y servicios (Holl 2006, MacLaren et al. 2014). En consecuencia, la detección de los impactos de disturbios de origen antrópico en la comunidad de regeneración forestal podría servir de apoyo a la toma de decisiones para el diseño, implementación y monitoreo de estrategias de restauración y conservación del bosque nativo.

El papel de las plantaciones de especies forestales exóticas y de la regeneración natural de los bosques en la dinámica del paisaje.

Los cambios del uso del suelo han sido definidos como uno de los principales objetivos de investigación en el desarrollo de estrategias para la gestión y la planificación sustentable del paisaje (Vitousek 1994). La transformación del paisaje genera profundos impactos en las funciones ecológicas y la pérdida de hábitats y la fragmentación de los bosques han aumentado pérdida de biodiversidad y la vulnerabilidad de estos ecosistemas (Laurance et al. 2006). La regeneración natural de los bosques podría limitar estos impactos (Morrison & Lindell 2010, FAO 2011), en especial en áreas donde la recolonización natural es más rápida debido a una mayor disponibilidad de semillas, la protección de la vegetación remanente y la conservación de los suelos (Prach et al. 2007, Chazdon 2008).

Figura 1.3. Plantación de *Eucalyptus spp*. en la Cordillera de la Costa de la Región de Los Ríos, sur de Chile.

La rápida expansión de las plantaciones forestales en el mundo es uno de los principales problemas para el desarrollo sustentable contemporáneo debido a los significativos conflictos ambientales y socioeconómicos asociados a éstas (Gerber & Veuthey 2011). En la actualidad, las plantaciones forestales representan cerca del 7% de la masa forestal a nivel global (FAO 2011). Al contrario de lo que sucede con la superficie de bosques naturales, los cuales han disminuido a una tasa anual de 5.2 millones de ha durante la última década, las plantaciones forestales están aumentando rápidamente, en especial en las áreas de más difícil acceso. Ello se debe a tres factores principales: el agotamiento de los recursos no renovables, los límites naturales para aumentar la productividad del suelo y el desarrollo técnico necesario para el establecimiento de plantaciones en condiciones de campo más restrictivas (Kröger 2013). Las plantaciones forestales comúnmente se han establecido en áreas agrícolas o pastizales, pero también se han expandido sustituyendo a los bosques naturales, constituyendo una de las causas más importantes de deforestación y fragmentación forestal (Echeverría et al. 2006, Brockerhoff et al. 2008, **Figura 1.3**).

En Chile, la explotación del bosque nativo ha sido objeto de diversas controversias durante las últimas décadas (Reyes & Nelson 2014). Así, extensas áreas de bosques han sido convertidas en plantaciones forestales de especies exóticas, sobre todo en áreas prioritarias para la conservación de la biodiversidad como la Cordillera de la Costa de las regiones central, centro-sur y sur del país (Echeverría et al. 2006, Schulz et al. 2010, Nahuelhual et al. 2012). Además de la deforestación, la fragmentación de los bosques nativos es uno de los principales resultados de estas prácticas productivas, y ambos fenómenos han supuesto una serie de conflictos entre las comunidades rurales, los conservacionistas, las empresas forestales privadas y el Estado (Reyes & Nelson 2014). Las plantaciones forestales de especies exóticas en Chile están dominadas por *Pinus radiata* (D. Don) y *Eucalyptus* spp., ocupan una superficie de 2.3 millones de ha (INFOR 2013) y la tasa de plantación supera las 37,000 ha anuales (CONAF 2014).

La distribución geográfica de los bosques templados chilenos ha disminuido notablemente durante el último siglo, principalmente como resultado de la conversión de los bosques nativos a otros tipos de uso del suelo (Smith-Ramírez 2004). La pérdida de bosques y la fragmentación de los ecosistemas forestales remanentes representan dos de las principales amenazas para la conservación de estos bosques en el sur de Chile (Echeverría et al. 2006, Lara et al. 2011, Nahuelhual et al. 2012). Ello es relevante, ya que estos bosques representan una de las ecorregiones globalmente reconocidas por su interés para la conservación de la biodiversidad mundial (Smith-Ramírez 2004, Myers et al. 2000) y han sido definidos como un objetivo de conservación urgente por diversas agencias y organizaciones mundiales como el World Bank y la World Wildlife Fund (Dinerstein et al. 1995), entre otras.

Priorización para la restauración forestal

En las últimas décadas se han establecido diversos acuerdos y compromisos globales para la restauración de los bosques, la cual ha sido reconocida como una prioridad mundial (Aronson & Alexander 2013). Una de estas iniciativas ha sido el acuerdo de la Conferencia de las Naciones Unidas sobre el Desarrollo Sustentable (Río+20) celebrada en Río de Janeiro, Brasil, en el cual se estableció como uno de los principales objetivos para el desarrollo sustentable global restaurar 150 millones de ha de bosques degradados para el año 2020 (IUCN 2012). Sin embargo, aún es necesario generar las bases que permitan abordar con éxito iniciativas de esta magnitud. Por ejemplo, utilizar de la manera más eficiente posible los recursos disponibles para la implantación de proyectos de restauración. Más aún si se considera que a una escala global más de dos billones de ha de bosques que han sido deforestadas o degradadas ofrecen oportunidades de restauración (Laestadius et al. 2011).

La identificación de áreas prioritarias podría aumentar la eficiencia y el impacto de los recursos disponibles para el diseño, la planificación y el establecimiento de programas de restauración forestal en los cuales las diferentes intervenciones podrían generar mayores beneficios. El problema de la priorización ha sido abordado de diversas maneras (Mittermeier et al. 1998). Para la identificación de áreas prioritarias de restauración de bosques es necesario desarrollar métodos que puedan ser aplicados en diferentes escalas espaciales, de tal modo que los resultados sean relevantes para la planificación del uso del suelo y para las estrategias de restauración y conservación de los bosques (Geneletti 2008, Moilanen et al. 2011).

Los métodos de análisis multicriterio han sido desarrollados para apoyar procesos de toma de decisiones que integren la comparación de diferentes alternativas posibles frente a un conjunto de objetivos y criterios (Belton & Steward 2002). Estos métodos normalmente son utilizados en la planificación territorial, para la cual es necesario evaluar de manera integral las diferentes alternativas de decisión, en especial por los múltiples criterios y los conflictos de intereses que afectan a la toma de decisiones (French et al. 2009). La planificación para la intervención en los bosques requiere integrar diversos factores y escenarios complejos, los cuales en la actualidad se pueden abordar a través de métodos de análisis multicriterio (Mills & Clark 2001, Kangas & Kangas 2005). Se han propuesto enfoques basados en estos métodos para identificar las zonas más idóneas y factibles para la reforestación a escala de paisaje (Uribe et al. 2014) y para la restauración de servicios ecosistémicos a escala local (Trabucchi et al. 2014). Otros estudios han definido como áreas potenciales para la restauración forestal específicamente zonas deforestadas (Orsi & Geneletti 2010), o se han basado en la percepción local de la degradación de los bosques determinada a partir de entrevistas (Ianni & Geneletti 2010). Sin embargo, a fecha de hoy no se han propuesto aproximaciones empíricas para definir áreas de bosques degradados como objeto de restauración. Además, el desarrollo de métodos para identificar áreas prioritarias de restauración forestal para el mantenimiento de la biodiversidad y la provisión de servicios ecosistémicos a escala de paisaje ha sido menos tratado en la literatura científica (Trabucchi et al. 2014).

Objetivos y organización de la Tesis Doctoral.

El objetivo general de esta Tesis Doctoral es identificar áreas prioritarias para la restauración forestal con el fin de conservar la biodiversidad y la provisión de los servicios ecosistémicos en la Cordillera de la Costa de la Región de Los Ríos, Chile. **Los objetivos específicos son los siguientes:**

(1) Determinar si la ganadería tiene un efecto cuantitativo (densidad) y/o cualitativo (origen por reproducción sexual o asexual) en la regeneración de la especie *Araucaria araucana*, y si este efecto varía según el régimen de propiedad de la tierra. Este objetivo se abordará en el **Capítulo 2.**

Los ecosistemas forestales están expuestos a diversos factores de alteración de origen antrópico, los cuales pueden influir directa e indirectamente en su composición y conservación. Si bien algunas especies forestales están protegidas por la legislación nacional e internacional, limitando o impidiendo su explotación maderera, otras alteraciones continúan actuando en los ecosistemas en los cuales estas especies se desarrollan, con impactos poco conocidos en su conservación. En un estudio piloto, analizaremos el impacto de la ganadería en la regeneración de *Araucaria araucana*, una de las especies forestales más emblemáticas de los bosques templados del sur de Chile y Argentina, cuya tala está prohibida por la legislación de ambos países y que ha sido incluida en el Apéndice I de la Convención sobre el Comercio Internacional de Especies de Flora y Fauna Silvestre en Peligro (CITES). El enfoque de análisis utilizado nos permitirá modelar el efecto que los factores de disturbio de baja intensidad tienen en la regeneración forestal. De igual modo, se analizará si estos efectos varían según el régimen de propiedad, incorporando así en nuestros análisis un factor socioeconómico de gran relevancia para la conservación de los bosques a nivel global, en especial en los países menos industrializados.

(2) Evaluar la influencia de la ganadería y de la tala selectiva en la regeneración forestal a nivel de especie y de comunidad, y si estas influencias varían según el estado sucesional del bosque (definido como bosque adulto, adulto/secundario, y secundario) y el régimen de propiedad (pequeñas y grandes propiedades). Este objetivo se abordará en el **Capítulo 3.**

La ganadería es una actividad económica creciente en los bosques del sur de Chile. Si bien los resultados generados a partir del **Objetivo específico 1** permitirán demostrar el efecto de la ganadería en la regeneración de la araucaria, nos preguntamos si dicho efecto será similar en otro tipo de bosques y si afectará a la regeneración tanto a nivel de especie como de comunidad. Además, junto con la ganadería, la tala selectiva es una de las intervenciones más comunes en los bosques para la producción de leña, una de las principales fuentes de energía en las áreas rurales y urbanas del sur de Chile. En este **Objetivo específico 2** analizaremos si la tala selectiva influye en la regeneración forestal y si esta influencia depende a su vez del régimen de propiedad y el estado sucesional del bosque. El estado sucesional podría influir en los efectos de los factores de disturbio al presentar los diferentes estados sucesionales grandes diferencias en cuanto a diversidad, estructura y funcionalidad, entre otros atributos. Los resultados permitirán proponer una definición de bosques degradados a partir de la evidencia empírica de las alteraciones en la regeneración forestal, lo que será útil para la identificación de áreas prioritarias para la restauración forestal (**Objetivo específico 4**).

(3) Analizar la dinámica de cambio de uso del suelo y determinar el papel de las plantaciones forestales de especies exóticas en la transformación de los bosques templados del sur de Chile durante las últimas décadas (años 1985-2011). Partimos de la **hipótesis** de que

las plantaciones forestales de especies exóticas han causado la mayor transformación de los bosques templados en el sur de Chile durante este período. Este objetivo se abordará en el **Capítulo 4.**

El reconocimiento de las alteraciones en los procesos que regulan un ecosistema a escala de paisaje es la base para definir estrategias de restauración forestal. Las causas de estas alteraciones son en gran parte generadas por la intensiva y creciente presión productiva, como la habilitación de áreas agrícolas y pastizales, la ganadería, la extracción de leña y carbón, la tala selectiva y la sustitución de los bosques nativos por plantaciones de especies forestales exóticas. El análisis de la dinámica del paisaje es fundamental para la evaluación histórica del estado de conservación del bosque. En este objetivo específico nos centramos en evaluar los principales cambios en el paisaje, en especial en los procesos de deforestación y fragmentación de los bosques nativos y en la expansión de las plantaciones forestales. A partir de estos resultados, se podrán determinar las áreas deforestadas que serán definidas como áreas potenciales para la restauración (**Objetivo específico 4**).

(4) Proponer un enfoque metodológico multicriterio para evaluar la idoneidad ecológica y la factibilidad socioeconómica de la restauración forestal. Este objetivo se abordará en el **Capítulo 5.**

La restauración forestal a escala de paisaje requiere de un adecuado marco de análisis que permita aumentar la eficiencia y los impactos de los recursos disponibles para diseñar, planificar y establecer políticas y acciones de restauración. La identificación de áreas prioritarias de restauración forestal puede orientar la selección de aquellas áreas en las cuales las intervenciones producirán los mayores beneficios y cuyas características socioeconómicas aumentarán sus posibilidades de éxito en el medio y largo plazo. En este objetivo específico utilizaremos una definición empírica de bosques degradados como área potencial de restauración. Para ello nos basaremos en los resultados de los **objetivos específicos 1 y 2**, a partir de los cuales se determinarán los mayores impactos por factores de disturbio en la regeneración forestal. Como área potencial para la restauración consideraremos las áreas deforestadas que fueron identificadas a escala de paisaje a partir del **Objetivo específico 3**. Esta información, junto con otras variables ecológicas y socioeconómicas espacialmente explícitas, servirá de base para definir las áreas más idóneas y factibles para el desarrollo de acciones de restauración forestal mediante un enfoque metodológico multicriterio.

Esta Tesis Doctoral está organizada en seis capítulos: un capítulo introductorio (la presente **Introducción general**); cuatro capítulos de **Resultados** que abordan los cuatro objetivos específicos detallados anteriormente, se presentan en formato de artículo científico y reproducen los contenidos publicados o en revisión en diferentes revistas científicas internacionales; un capítulo que presenta una **Discusión general** que integra las discusiones de los cuatro anteriores; y un capítulo con las **Conclusiones generales** resultantes de este trabajo (**Tabla 1.1**).

Tabla 1.1. Organización de la Tesis doctoral. Se indican los objetivos específicos, los capítulos en los que se desarrollan y las publicaciones resultantes, si procede.

Objetivos específicos	**Capítulos**	**Publicación**
	Capítulo 1 Introducción general	
1. Determinar si la ganadería tiene un efecto cuantitativo (densidad) y/o cualitativo (origen sexual/asexual) en la regeneración de la especie *Araucaria araucana*, y determinar si este efecto varía según el régimen de propiedad de la tierra.	**Capítulo 2** Impacts of cattle on the South American temperate forests: Challenges for the conservation of the endangered monkey puzzle tree (*Araucaria araucana*) in Chile.	Zamorano-Elgueta et al. 2012. **Publicado en *Biological Conservation.***
2. Evaluar la influencia de la ganadería y de la tala selectiva en la regeneración forestal a nivel de especie y de comunidad, y si estas influencias varían según el estado sucesional del bosque (definido como bosque adulto, adulto/secundario, y secundario) y el régimen de propiedad (pequeñas y grandes propiedades).	**Capítulo 3** The differential influences of human-induced disturbances on tree regeneration community: a landscape approach.	Zamorano-Elgueta et al. 2014. **Publicado en *Ecosphere.***
3. Analizar la dinámica de cambio de uso del suelo y determinar el papel de las plantaciones forestales de especies exóticas en la transformación de los bosques templados del sur de Chile durante las últimas décadas (años 1985-2011).	**Capítulo 4** Seeing the glass as half-full or half-empty: a story of native forest replacement by exotic plantations in southern Chile (1985-2011), and how natural regeneration partly compensate these losses.	Zamorano-Elgueta et al. **En revisión en *Forest Ecology and Management*.**
4. Proponer un enfoque metodológico multicriterio para evaluar la idoneidad ecológica y la factibilidad socioeconómica de la restauración forestal.	**Capítulo 5** Restoring forests for biodiversity and ecosystem services: A spatial multicriteria approach to identify priority areas.	Zamorano-Elgueta et al. **En revisión en *Landscape & Urban Planning***
	Capítulo 6 Discusión general	
	Capítulo 7 Conclusiones generales	

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Árbol adulto de *Araucaria araucana*, Cordillera de Nahuelbuta, Región de La Araucanía.

CAPÍTULO 2

Impacts of cattle on the South American temperate forests: Challenges for the conservation of the endangered monkey puzzle tree (*Araucaria araucana*) in Chile

Este capítulo reproduce íntegramente el texto del siguiente manuscrito:

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Impacts of cattle on the South American temperate forests: Challenges for the conservation of the endangered monkey puzzle tree (*Araucaria araucana*) in Chile

Abstract

Notwithstanding the increasing cattle activity on the South American temperate forests, its impacts on the forests regeneration are yet poorly understood. We investigated the influence of cattle on the regeneration of monkey puzzle tree (*Araucaria*), an endangered conifer of the temperate forests of Chile and Argentina, on properties of small landowners and of timber companies. In 36 100 x 20 m plots, we recorded the number of seedlings and saplings from seeds and resprouts, the number of cattle dung pats and the density of parent trees. We used the cattle dung pats as a surrogate of cattle activity (the cattle intensity index, CAI). The regeneration was analyzed as a function of the CAI, land tenure regime, the study site, and the number of parent *Araucaria* trees. We used likelihood methods and model selection for data analysis. Overall, there was a negative exponential influence of the CAI on all response variables. In small landowner forests, even low cattle intensities caused regeneration to drop rapidly to zero, whereas in plots owned by timber companies regeneration decreased smoothly as the CAI increased. The CAI affected regeneration of *Araucaria* qualitatively by decreasing the ratio of sexual/asexual regeneration, which may lead to problems of genetic drift in the long-term. Our results suggest that conservation of a single species does not necessarily ensure its long-term persistence It is necessary to protect the ecosystems in which the species grows and involve local stakeholders in the development of management strategies that reduce the impacts of cattle ranching.

Keywords: South American temperate forests, cattle impact, *Araucaria araucana*, land use, Nahuelbuta, Chile.

Introduction

Araucaria araucana (Molina) K. Koch (Araucariaceae) is an impressively large and long-lived conifer, reaching 50 m in height, 2.5 m in diameter and up to 1300 years in age (Montaldo 1974). This conifer is commonly known as the monkey puzzle tree, Pehuén or *Araucaria*, and its current distribution spans only three degrees of latitude and is divided between a main area straddling the Chilean and Argentinian slopes of the Andean cordillera, and two unconnected populations in the Coastal Range of Chile (**Fig. 2.1**). The current distribution derives from a previously more extensive range, which has been severely reduced and fragmented by logging, man-made fires and land clearance since European colonization (Veblen 1982). The ecology of *Araucaria* is disturbance-driven, principally by volcanism, natural and human ignited fires, landslides and wind, and it has effective adaptations to thrive under these disturbance regimes such as thick bark and epicormic buds (Burns 1993, González et al. 2006). *Araucaria* is generally dioecious, but may occasionally be monoecious with predominantly gravity-dispersed seeds and wind-dispersed pollen. Most of the *Araucaria* seeds fall directly under the canopy or a few meters away from the parent tree, due to their large size (2-4 cm long, 1-2 cm wide) and heavy weight (3.5-5.0 g, González et al. 2006). *Araucaria* seeds can also be dispersed over greater distances by birds, rodents and other animals (Veblen 1982, González et al. 2006). Asexual reproduction by root suckering has been reported on the Andes and the Coastal Range (Schilling and Donoso 1976, Cortés 2003), particularly under severe disturbance regimes (Cortés 2003, González et al. 2006). Yet it is unknown how important this process is for population maintenance and expansion (Veblen et al. 1995).

Araucaria is a socially significant species, producing high-quality timber and providing a unique resource for tourism and recreation. The tree has a relevant role in the culture of the indigenous Pehuenche people and is also valued for its large, edible seeds, which are extensively collected for local markets (Aagesen 1998a). *Araucaria* has been classified under the IUCN guidelines as vulnerable (Farjon and Page 1999), and is currently officially protected in both Chile and Argentina as well as internationally through its listing in Appendix I of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). Despite its protected status and outstanding ecological, economic and cultural significance, *Araucaria* forests continue to experience intense human-induced pressures, such as grazing and harvesting, both for timber and seeds (Aagesen 1998b). Its thick bark and resprouting ability may confer the *Araucaria* a competitive advantage under fire regimes relative to other coexisting species with thin bark such as *Nothofagus pumilio* and *N. dombeyi* (Schilling and Donoso 1976, González and Veblen 2007). Nevertheless, grazing and high-frequency human set fires may have profound effects on the species long-term persistence, mostly by altering its regeneration capacity. Nowadays, the most obvious sign of *Araucaria* forest degradation is the lack of natural regeneration, which might be exacerbated by livestock and feral exotic animals as wild boar (*Sus scrofa*) and red deer (*Cervus elaphus*), which consume seeds in autumn and seedlings in spring, and trample seeds, seedlings and saplings during grazing (Gallo et al. 2004, Shepherd and Ditgen 2005, Sanguinetti and Kitzberger 2009a). In addition, cattle ranching can cause soil compaction, which may change soil structure and contribute to the increased incidence of water stress, tree mortality during dry periods and erosion (Fleischner 1994, Hobbs 2001).

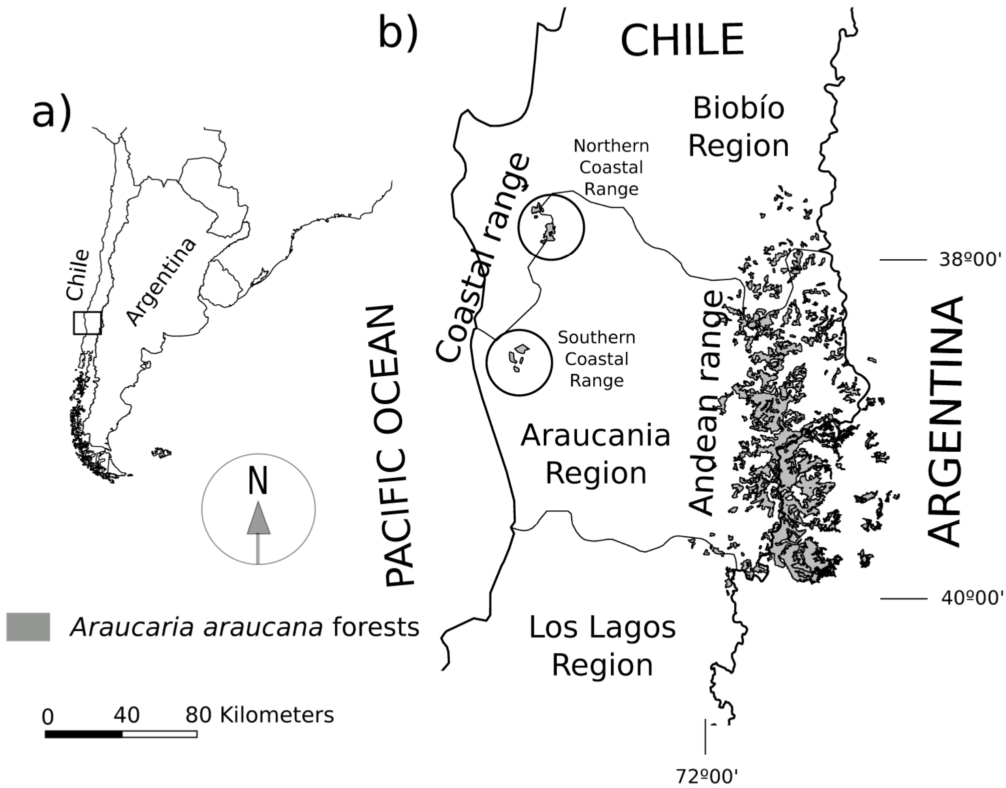

Figure 2.1. (a) Location of the study area and; (b) distribution of *Araucaria* forests within the Coastal and Andean ranges, and location of the two main populations of monkey puzzle that were sampled in this study in the Coastal Range.

Studies about the regeneration of the species include the effects of biophysical factors (Veblen 1992, Christie and Armesto 2003) and stand variables (Donoso and Nyland 2005) on seedling densities in late-successional and old-growth evergreen (Valdivian) forests in Chile. More recently, the effects of masting, seed predation and understory vegetation on seedling establishment (Sanguinetti and Kitzberger 2008, 2009a, b), and the long-term implications of fire in the regeneration of the species (Burns 1993, González et al. 2005, González and Veblen 2007) have been addressed. However, notwithstanding the widespread coincidence of cattle and the distribution of *Araucaria* forests, the impacts of animal husbandry on the regeneration of this tree species are yet poorly understood. The effects of grazing on *Araucaria* may vary under different land tenure regimes, namely rural properties of small landowners and large properties of timber companies. Currently, most of Chile's forests on the Coastal Range are owned by a few number of large timber companies. During the last decades, these companies have been responsible for the replacement of large tracts of native forests by exotic *Pinus radiata* and *Eucalyptus spp*. plantations (Taylor 1998, Lara and Veblen 1993, Echeverría et al. 2006, Lara et al. 2010). Paradoxically, some of the properties owned by timber companies, where exotic plantations are expanding, are nowadays refuge to some of the last remnants of *Araucaria* forests in the Coastal Range. Improving our understanding of how *Araucaria* regeneration responds to cattle intensity under these two major land tenure regimes might therefore contribute to manage the conservation and sustainable use of this endangered species more efficiently.

This study is aimed to elucidate whether there is a negative effect of cattle on the regeneration of *Araucaria araucana* in the Coastal Range, which are more exposed to human disturbance compared to the Andes. We analyzed how the impact of cattle on regeneration varies according to land tenure regime, site (northern and southern) and density of *Araucaria* parent trees. Specifically we investigated the impacts of cattle ranching on: (1) the number of seedlings and saplings; (2) the number of seedlings and saplings originated from seed (sexual reproduction); (3) the number of resprouting seedlings and saplings (asexual reproduction); and (4) the ratio between seedlings and saplings originated from seed and those from resprout.

Methods

Study area

The study was carried out in the Nahuelbuta mountain range, within the coastal distribution range of *Araucaria* in Chile (**Fig. 2.1**). We selected this study area for three main reasons: (1) *Araucaria* populations in the southern limit of its distribution in the Coastal Range are genetically distinct to the other populations: populations in the northern Coastal Range do not differ genetically from those in the Andean range (Bekessy et al. 2002). Thus, our study includes two genetically distinct extant populations; (2) human disturbance represents a serious threat for forest conservation in the Coastal Range, mainly due to its physiographical features; elevations rarely exceed 1500 m, and therefore provide greater accessibility and represent higher vulnerability (Armesto et al. 1995); and (3) the scarcity of protected areas in this region (Armesto et al. 1998) poses an additional threat for *Araucaria* populations. Most *Araucaria* forests are included within large properties of timber companies, and to a lesser extent, are within the properties of small landowners that extract firewood and timber for charcoal from associated species, and use the forest for cattle grazing.

Nahuelbuta is characterized by abundant endemic flora and fauna, which probably reflects the location of vegetation refuges during the last glacial period (Armesto et al. 1995). The main forest association in the study area is *Araucaria-Nothofagus dombeyi* forest. Sampling was conducted in two different sites: the northern Coastal Range (NCR), and the southern

Coastal Range (SCR, **Fig. 2.1**). The NCR site bordered the Nahuelbuta National Park (37°44'S, 72°55'O; 37°51'S, 73°03'O, elevation between 700 and 1400 m). The SCR site matches the southern limit of *Araucaria* distribution in the Coastal Range (38° 29' lat S, 73° 12' long O, elevation between 450 and 700 m). The predominant climate is temperate with Mediterranean influence, with average temperatures between -1ºC in winter and 9°C in summer, and an average annual precipitation of 1500-2500 mm (Di Castri and Hajek 1976, González et al. 2006). Soils derived from metamorphic material and granitic rocks are thin to moderately deep (15 - 180 cm), loamy-clay to clay at depth, extremely acidic to very strongly acidic (pH = 3.0 - 5.5), and have moderate to high erodibility, and low nutritient levels (IREN-CORFO 1964, Montaldo 1974).

Sampling methods

We established 36 100 x 20 m sampling plots: 20 plots were located in rural properties of small landowners (sampling size: NCR = 17 plots, SCR = 3 plots) and 16 plots in large properties of timber companies (sampling size: NCR = 9 plots, SCR = 7 plots). Within each plot, we counted monkey puzzle seedlings (< 1.3 m in height) and saplings (> 1.3 m tall and < 5 cm in diameter at breast height, DBH) in 30 2 x 2 m sub-plots set at 10 m intervals from each other along three 100 m line transects running parallel to the longest axis of the plot. Using a non-destructive technique we dug around each sapling and seedling by hand and hand shovel to differentiate between sexual (seeds) and asexual (resprouts, *sensu* González et al. 2006) regeneration. For the sake of brevity we will refer henceforth to seedlings and saplings from seeds and resprouts, although seedlings in a strict sense refer only to young plants grown from seeds. Once we made the observations, we put back and compacted the soil around the seedlings and saplings. In each 100 x 20m plot we counted the number of cattle dung pats and estimated the number of dung pats/ha as a surrogate of the current cattle density and trampling pressure. The dung pats has a very slow decomposition rate in this region due to the cold temperatures that prevail during most of the year. Therefore our estimation of cattle intensity is likely to include the accumulated activity of cattle for a broad period of time, spanning at least for the last one to two years (C. Zamorano, pers. obs.). Henceforth we will refer to this variable as the cattle intensity index (CAI). Measurements of damage by grazing were not considered since *Araucaria* is not a palatable species. In each of these plots we also counted the number of parent trees, defined as those individuals > 5 cm in DBH. Even though 5 cm in DBH might seem a low size for trees to become reproductive, we must note that reproductive capacity in this species is more related to age than to size. *Araucaria* reaches sexual maturity between 15 to 25 years old (Montaldo 1974), and since it is a slow growing species, individuals > 5 cm in DBH often attain this age (Cortés 2003). We did not determine the sex of adult *Araucaria* trees because of the difficulties to do such distinction in tall trees and/or in dense stands.

Statistical analyses

We fitted models for the following response variables: total number of seedlings and saplings; number of seedlings and saplings originated from seed (sexual reproduction); number of resprouting seedlings and saplings (asexual reproduction); and the ratio between seedlings and saplings originated from seed and resprouts (sexual/asexual ratio). The latter was calculated as follows for seedlings (we applied the same formula to saplings):

$$\textit{Sexual/asexual ratio} = \frac{\textit{Number of seedlings from seed} + 1}{\textit{Number of resprouting seedlings} + 1}$$

These response variables were analyzed as a function of the cattle intensity index (CAI), land tenure regime (small landowner, timber company), the study site (NCR, SCR), and the number of parent trees. We used likelihood methods and model selection as an alternative to traditional hypothesis testing (Johnson and Omland 2004, Canham and Uriarte 2006) for data analysis. Following the principles of likelihood estimation, we estimated model parameters that maximized the likelihood of observing the regeneration measured in the field, given a suite of alternative models. We examined eight different nested models. For each model, we conducted separate analyses for each individual response variable, namely overall seedling and sapling regeneration, as well as analyses considering sexual and asexual seedling and sapling regeneration. For each response variable (Y), our simplest model takes the form:

$Y = a \cdot e^{b \cdot CAI}$ (Equation 1)

The first term in the model, a, is an estimated parameter that represents the average *Araucaria* regeneration in the absence of cattle effects. The second term, $e^{b \cdot CAI}$, controls for the effect of cattle intensity. This model implies that regeneration changes exponentially as a function of the number of dung pats, where b is an estimated parameter that controls for the slope of the curve.

We tested three variants of Equation 1. A first variant allowed the effect of the cattle intensity on *Araucaria* regeneration to vary depending on land tenure:

$Y = a \cdot e^{b_i \cdot CAI}$ (Equation 2)

where b_i defines the slope of the exponential curve between regeneration and cattle intensity for the two classes of land tenure analyzed. To test for the possibility of a site effect independent of cattle intensity, we ran a modified version of the simplest model in which the average *Araucaria* regeneration (i.e. term b in Equation 1) was estimated separately for the NCR and the SCR study sites:

$Y = a_j \cdot e^{b \cdot CAI}$ (Equation 3)

The last variant included both the effects of land tenure on the slope of the exponential curve, and the inclusion of separate terms for average *Araucaria* regeneration in the two study sites:

$Y = a_j \cdot e^{b_i \cdot CAI}$ (Equation 4)

We also ran variations of Equation 1 to 4 including a density effect of parent trees on regeneration. The most complex model, a variant of Equation 4, takes the form:

$Y = a_j \cdot e^{b_i \cdot CAI + c \cdot Number\ of\ trees}$ (Equation 5)

We used simulated annealing, a global optimization procedure, to determine the most likely parameters (i.e. the parameters that maximize the log-likelihood) given our observed data (Goffe et al. 1994). We used a Poisson error structure for variables involving a count number of seedlings and saplings, and a normal error structure with the variance as a power function of the mean for the sexual/asexual ratio of seedlings and saplings. The latter required estimating an additional parameter, *delta*, to determine the scaling of the variance to the mean. Alternative models were compared using the Akaike Information Criterion (AIC_c) corrected for small sample sizes (Burnham and Anderson 2002). Models with a difference in AIC_c < 2 units are considered to have equivalent empirical support, whereas a difference value within only 4-7 units of the best model has considerably less support. Differences in AIC_c > 10 indicate that the worse model has virtually no support and can be omitted from further consideration. We used asymptotic two-unit support intervals to assess the strength of evidence for individual maximum likelihood parameter estimates (Edwards 1992). These are simply the range of parameter estimates for which 'support' (log-likelihood) is within two units of the maximum log-likelihood, and were determined by incrementally varying parameter estimates above and below the maximum likelihood estimated until log-likelihood had dropped by two units. The R^2 of the model fit (1 – SSE/SST, sum of squares error (SSE); sum of squares total (SST)) of observed versus predicted was used as a measure of goodness-of-fit. All analyses were performed using the 'likelihood' package (Murphy 2008) written for the R environment (R Development Core Team 2010).

Results

Seedlings were almost threefold more abundant than saplings (**Table 2.1**). Sexual regeneration in seedlings was more frequent than asexual regeneration, whereas saplings showed the opposite trend. Mean density of seedlings and saplings was considerably higher in plots of timber companies than in those of small landowners (**Table 2.1**).

Table 2.1. Mean density and standard deviation (in brackets) of *Araucaria* seedlings and saplings in sampled plots of small landowners and timber companies, including sexual (i.e. from seed) and asexual (i.e. from resprouting) regeneration.

	Mean density (plants/ha)		
	All plots	Small landowners	Timber companies
Total Seedlings	861 (761)	613 (874)	1172 (1870)
Sexual regeneration	571 (938)	467 (729)	703 (1160)
Asexual regeneration	289 (504)	146 (145)	469 (710)
Total Saplings	333 (241)	229 (392)	463 (517)
Sexual regeneration	134 (261)	96 (154)	182 (353)
Asexual regeneration	199 (218)	133 (238)	281 (164)

Comparison of alternate models revealed the best-fits for the different response variables (**Table 2.2**). For total and sexual seedling, the best-fit models showed site-specificity and an effect of land tenure on the response of *Araucaria* regeneration to cattle intensity, without an effect of the parent tree density (R^2 = 0.40 and 0.33, respectively). For total saplings and asexual regeneration of seedlings, the best-fit model included the effects of land tenure and tree density (R^2 = 0.25 and 0.27, respectively). Site specificity and density-dependence were the main effects included in the best-fit model for regeneration of sapling resprouting (R^2 = 0.28). For seedlings, the results suggest a slight response of the sexual/asexual ratio to cattle intensity of land tenure regimes and sites (R^2 = 0.16), whereas for saplings this response was even lower, with the best-fit model included only site-specificity (R^2 = 0.13, **Table 2.2**). There was a density effect in total saplings, and seedling and sapling resprouts (**Table 2.2**), indicating that the higher the number of parent trees, the more regeneration from resprouting. Overall, there was a negative influence of cattle intensity on all response variables, except for sapling resprouts (**Fig. 2.2**).

The northern Coastal Range (NCR) had, on average, greater regeneration than the southern Coastal Range (SCR, see model coefficients in **Table 2.3**). In addition, the estimated sexual/asexual ratio of seedlings was about threefold larger in the NCR than in the SCR, and almost eightfold larger for saplings (**Fig. 2.3**). It must be noted that the estimated mean value of the sexual/asexual ratio of saplings in the SCR (parameter a_{SCR} in **Table 2.3**) was below one, thus indicating a predominance of asexual as compared to sexual regeneration even in the absence of cattle (**Fig. 2.3**). Parameter b in the exponential term of the models was approximately one order of magnitude higher in small properties than in forest companies

(**Table 2.3**). As a result, in small landowner forests, low cattle intensities (100-200 dung pats/ha) caused regeneration to drop rapidly to zero, whereas in plots of timber companies, regeneration decreased smoothly as cattle intensity increased, and models predicted some degree of regeneration even at high levels of cattle intensity (> 1000 dung pats/ha), particularly for seedlings (**Fig. 2.2**). The same applied to the sexual/asexual ratio of seedlings (**Fig. 2.3).** No density effect was detected in the sexual/asexual ratio of seedlings and saplings.

Table 2.2. Comparison of alternate models (using AIC_C) for seedling and sapling *Araucaria* regeneration, including sexual (i.e. from seed) and asexual (i.e. from resprouting) regeneration. The best model (lowest AICc) is indicated in boldface type. The number of parameters k and R2 refer to the best model (see table 2.3).

Response variable	AIC_c[a]									
	Basic model[b]		Land tenure model[c]		Site-specific model[d]		Site-specific land tenure model[e]			
	No-density effect	Density effect	No-density effect	Density effect	No-density effect	Density effect	No-density effect	Density effect	k[f]	R^2
Seedlings density	516.25	478.29	414.97	378.43	476.44	406.31	**360.70**	432.20	4	0.40
Sexual regeneration	485.20	480.43	395.32	391.59	426.93	405.38	**318.13**	355.23	4	0.33
Asexual regeneration	285.75	240.23	268.09	**223.39**	286.74	232.02	268.65	288.48	4	0.27
$Ratio_{S/A}$	185.95	188.27	171.81	174.60	180.31	183.16	**165.40**	168.83	5	0.16
Saplings density	227.16	211.44	209.11	**192.97**	227.56	214.08	210.37	230.25	4	0.25
Sexual regeneration	166.81	164.98	145.33	144.25	159.28	152.67	**133.06**	146.64	4	0.24
Asexual regeneration	161.11	148.25	159.47	146.02	576.55	**141.19**	144.71	149.96	4	0.28
$Ratio_{S/A}$	110.71	113.29	111.22	114.16	**90.85**	93.46	93.00	99.66	4	0.13

[a] Akaike Information Criterion corrected for small sample sizes.
[b] This model predicts *Araucaria* regeneration only as a function of the CAI.
[c] This model allows variations in the response of *Araucaria* regeneration to the number of cattle dung pats between the two land tenure classes.
[d] This model estimates different average responses of *Araucaria* regeneration in the two study sites.
[e] This model is a combination of the previous two models.
[f] Number of model parameters.

Discussion

Cattle impacts on the regeneration of Araucaria

During the last decades, the native forests of the Nahuelbuta mountain range have been exposed to substantial loss as a result of the expansion of commercial plantations, agricultural activities and urban and industrial sprawl (Aguayo et al. 2009). Despite legal protection, *Araucaria* forests in Chile are still exposed to constant direct and indirect human disturbances, similarly to what has been reported in mixed ombrophyllus forests with *Araucaria angustifolia* in southern Brazil (Vibrans et al. 2011). Cattle, fire, and selective logging of forest species associated with this conifer, especially *Nothofagus pumilio* and *N. dombeyi* for timber and *N. antarctica* for firewood and charcoal, are common disturbances in these forests. Cattle ranching may have profound impacts on forest ecosystems (Hobbs 2001, Floyd et al. 2003). Free-ranging cattle grazing on seedlings and saplings can diminish, damage or prevent the recruitment of many tree and other species by trampling, thus promoting changes in species composition (Hobbs 2001, Floyd et al. 2003; Wassie et al. 2009). In Nahuelbuta, soil compaction caused by intensive cattle ranching since the late 1940's in the NCR and early 1950's in the SCR has probably aggravated the restrictive natural soil conditions that characterize the region, especially in properties of small landowners (Torrejón and Cisternas 2003, Zamorano et al. 2008).

Our study reveals that *Araucaria* regeneration is severely affected by cattle ranching, both in properties of timber companies and of small landowners. In the large properties of private timber companies, forests are used by neighbouring farmers and even by those from nearby towns as a source of fodder and refuge for cattle. In some cases, companies allow this use through the "leasing of grazing rights", but this does not include any control over livestock density or the amount of forest area that is grazed. The negative exponential models show that small increases in cattle intensity may lead to a substantial decrease in the regeneration of the species, particularly for seedlings. The effect of cattle ranching on the regeneration of *Araucaria* appears much less dramatic in plots owned by timber companies than in small rural properties (**Fig. 2.2**). Two complementary explanations may account for such differences in the response to cattle ranching. On the one hand, forests owned by private companies are often used by cattle as a winter refuge, whereas cattle ranching in small rural properties take place throughout the entire year. Thus, although not directly accounted for in our study, the frequency of a disturbance might be as important as its intensity on the regeneration of a species. On the other hand, firewood and timber extraction are one of the main sources of income in small rural properties (Zamorano et al. 2008). Although current legislation prohibits logging of *Araucaria*, extraction of forest species associated with this conifer, especially *Nothofagus* spp. is allowed. Such disturbances, which occur less intensively in lands of timber companies (C. Zamorano, unpublished results), may act synergistically with cattle ranching (Hobbs 2001, Laurance and Useche 2009), amplifying its impact on monkey puzzle regeneration.

The response of regeneration to cattle intensity was also site-specific in most cases. The SCR displayed, in average, lower values of seedlings and saplings than the NCR (**Figure 2.2**). Overall differences found in regeneration between the NCR and the SCR can be explained by two main reasons: (1) soils in the NCR have better physical and chemical properties than those in the SCR (Cortés et al., 2001). Unfortunately, soil measures were not directly accounted for in this study; and (2) *Araucaria* forests in the NCR are, in general terms, better conserved, display more complex structure and holds a larger number of plant species than forests in the SCR (Cortés et al. 2001, Cortés 2003, Zamorano et al. 2008).

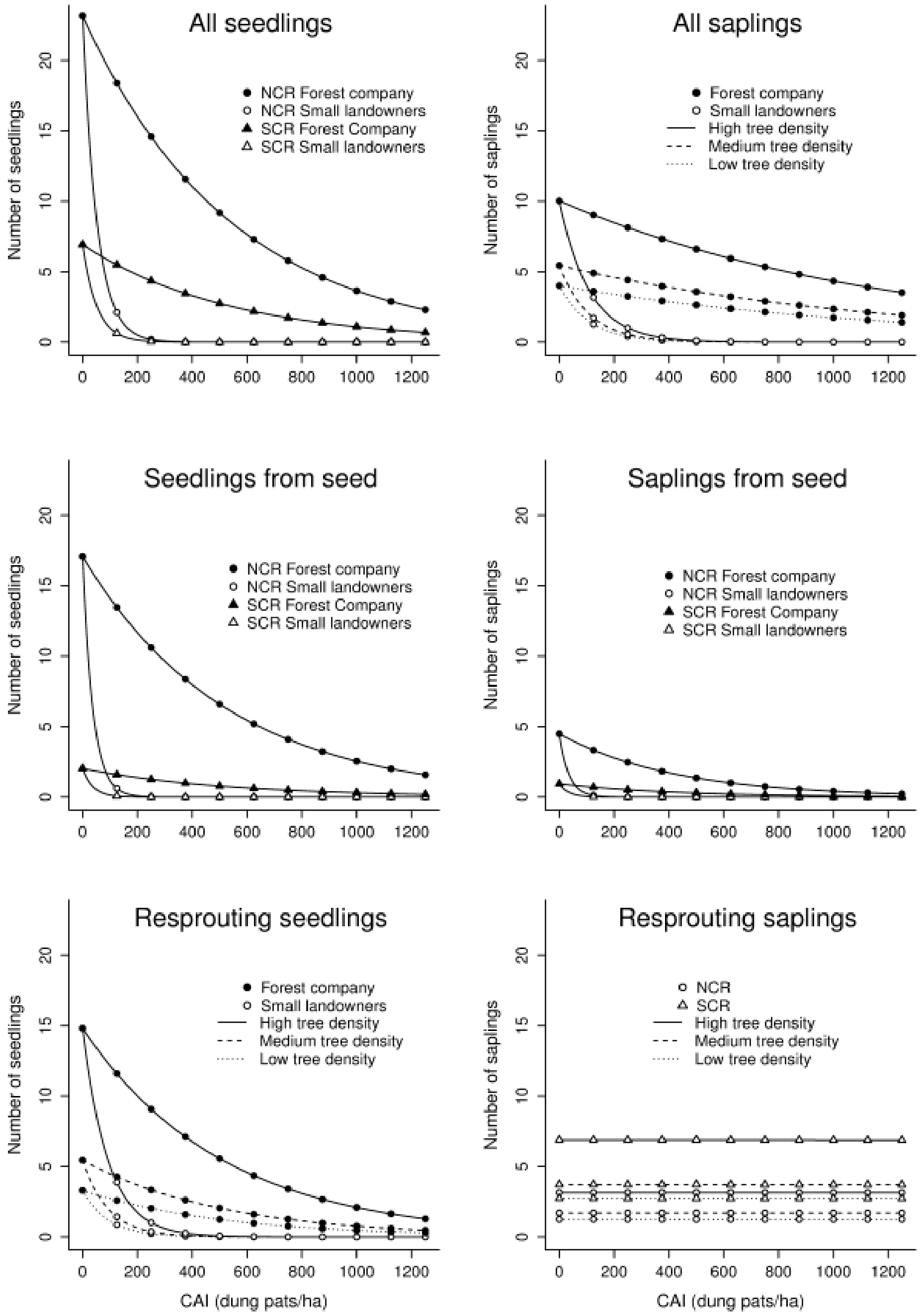

Figure 2.2. Predicted decrease of seedling and sapling density (number per ha) as a function of the Cattle Intensity Index (CAI) for the best selected models (see Table 1). For simplicity, only three tree densitiy classes were represented in those models for which this term was important: high density = 200 *Araucaria* trees/ha; medium density = 100 trees/ha; low density = 50 trees/ha.

Among seedlings, the proportion of resprouts, was relatively high (ca. one third of the total number of seedlings) compared to other studies in *Araucaria* stands at the Andes that describe ca. 10% of seedlings from asexual regeneration (e.g. Sanguinetti and Kitzberger 2009b). This figure was even higher for saplings, with more than half the total number of saplings having an asexual origin (**Table 2.1**).

Table 2.3. Parameter estimates and two-unit support intervals (in brackets) for the most parsimonious models of seedling and sapling *Araucaria* regeneration, including sexual (i.e. from seed) and asexual (i.e. from resprouting) regeneration, as well as the sexual/asexual ratio ($Ratio_{S/A}$).

Response variable	*a*	a_{NCR}	a_{SCR}	*B*	$b_{small\ property}$	$b_{forest\ company}$	*c*	*delta*[a]
Seedlings		23.18 [20.67, 25.59]	6.92 [4.84, 9.64]		-0.0096 [-0.0116, -0.0077]	-0.0009 [-0.0011, -0.0006]		
Sexual regeneration		17.09 [14.85, 19.42]	2.02 [0.92, 3.54]		-0.0133 [-0.0166, -0.0105]	-0.0009 [-0.0013, -0.0006]		
Asexual regeneration	2.00 [1.70, 2.38]				-0.0053 [-0.0075, -0.0033]	-0.0009 [-0.0016, -0.0004]	0.005 [0.004, 0.006]	
$Ratio_{S/A}$		3.94 [3.22, 5.13]	1.46 [0.92, 2.40]		-0.0038 [-0.0046, -0.0025]	-0.0002 [-0.0008, 0.0006]		1.147 [0.902, 1.466]
Saplings	2.95 [2.52, 3.53]				-0.0046 [-0.0046, -0.0027]	-0.0004 [-0.0008, -0.0001]	0.003 [0.002, 0.004]	
Sexual regeneration		4.49 [3.35, 5.77]	0.93 [0.26, 2.08]		-0.0146 [-0.0217, -0.0090]	-0.0012 [-0.0024, -0.0005]		
Asexual regeneration		0.92 [0.66, 1.21]	2.01 [2.75, 0.71]	-9.19e-07 [-0.0005, 0.0004]			0.003 [0.002, 0.004]	
$Ratio_{S/A}$		1.75 [1.39, 2.15]	0.31 [0.23, 0.51]	-0.0006 [-0.0009, -0.0002]				0.853 [0.662, 1.081]

[a]Parameter delta determines the scaling of the variance to the mean

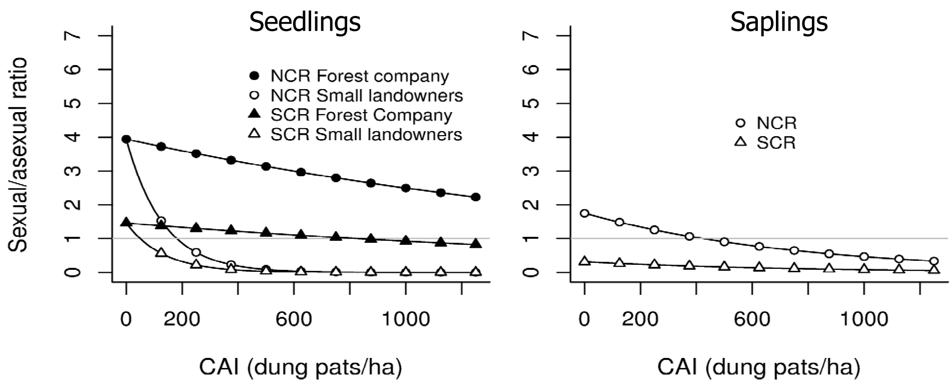

Figure 2.3. Predicted decrease of seedling and sapling sexual/asexual ratio as a function of the Cattle Intensity Index (CAI) for the best selected models (see Table 2.2). Horizontal grey lines indicate the point below which asexual regeneration surpasses sexual regeneration.

Best fit models of the sexual/asexual ratio of seedlings and saplings (**Fig. 2.3**) might indicate that, in the face of disturbance, resprouts are expected to have higher chances of survival than plants germinating from seeds. Resprouts have advantages, including an already established root system with a large surface area for water and resource acquisition and high stored energy reserves (Simões and Marques, 2007; Miller and Kauffman, 1998), which enables them to withstand disturbances and compete more efficiently for resources. It is also important to remark that density of parent trees does not play an important role in explaining the observed regeneration of seedlings and saplings from seed, but might be important in determining regeneration from resprouts (**Fig. 2.2**). This involves that *Araucaria* densities of about 50 trees/ha, the minimum number recorded in our plots, would suffice to produce the current observed figures of seedling and sapling regeneration from seed.

Conservation prospects

For some species, a limited level of grazing can increase regeneration by removing competitive vegetation, reducing fire hazards and through fertilization from manure (Kuiters et al. 1996, Blackhall et al. 2008). Although *Araucaria* is highly tolerant to disturbances (Burns 1993, González et al. 2006), increasing levels of cattle ranching have a rapid response in its regeneration, both quantitatively, by reducing the number of seedlings and saplings, and qualitatively, by decreasing the ratio of sexual/asexual regeneration. These impacts may lead to problems of genetic drift which would ultimately have profound implications for the conservation of this species, especially for populations in the southern limit of *Araucaria* distribution, which are genetically distinct from the other populations (Bekessy et al. 2002).

Overall, the results of this study reveal that partial conservation actions focused on a single species and limited to cutting prohibitions do not necessarily ensure its long-term persistence. It is necessary to develop scientific-based conservation plans focused on the species autoecology, considering its particular germination and growth characteristics, as well as the optimal environmental conditions that determine its distribution. This involves developing synergies between single-species (Simberloff 1998) and ecosystem approaches (Walker and Salt 2006, Lindenmayer et al. 2007).

Even though our results suggest the incompatibility of the conservation of *Araucaria* with cattle ranching, cattle eradication is not a feasible solution. It is unpractical to control livestock access to the properties of timber companies or even the National Park, given their wide ranges and lack of fences. A tighter control in small and medium-sized properties would inevitably increase pressure of illegal cattle ranching in larger uncontrolled properties, as well as unsustainable productive pressure on small farms and rural poverty of families that depend on livestock as their main source of income (Zamorano et al. 2008). Current policies oriented towards poverty reduction in rural areas of Chile mainly target agriculture, livestock or planting of exotic forest species, whereas conservation and sustainable management of native forests has been systematically neglected (Tecklin and Catalán 2005). As a result, inevitably contradictory interests arise between current production-oriented agricultural and silvicultural policies and partial and limited species-focused conservation policies for *Araucaria*. So what is the path forward? An important step is to promote the regulation of livestock densities and

Araucaria protection among local actors, instead of relying on policies imposed that have small chances to assure sustainable long-term solutions for the conservation of this species. Multiscale approaches offer nowadays a unique opportunity to conciliate policies for rural development and *Araucaria* conservation through the concerted interaction of all relevant stakeholders (Reed, 2008) in order to attain long-term sustainable land use planning at local and landscape scales, considering both the local interests and the environmental features of the region. To guarantee the success of multiscale approaches, stakeholder participation must be institutionalised, promoting organisational cultures that can facilitate processes where goals are negotiated and outcomes are necessarily uncertain (Reed 2008).

Notwithstanding that timber companies have been responsible for the conversion of large extensions of native forests to exotic plantations during the last decades (Taylor 1998, Echeverría et al. 2006, Lara et al. 2010), the lower cattle impact recorded on *Araucaria* regeneration as compared to properties of small landowners offer an excellent opportunity for the conservation of these forests, as these remnants are less exposed to the negative effects of cattle and other disturbances. Some practices to foster conservation of this species include active restoration by planting *Araucaria* seedlings on patches where regeneration is very scarce or absent, promoting reconversion of exotic plantations established in formerly *Araucaria* forest patches, and implementing monitoring programmes to control the invasion of exotic forest species within *Araucaria* remnants. In addition, timber companies might contribute to conservation of the species by identifying pasturelands or areas more suitable for cattle ranching in order to reduce cattle pressure on the extant *Araucaria* remnants (Polasky et al. 2008), particularly when leasing grazing rights.

In Chile, it has recently been promulgated a legal instrument to the conservation and sustainable management of native forests (Law N° 20283 of Development and Recuperation of the Native Forests). This law defines subsidies for afforestation with threatened forest species, which may represent a first step towards developing further forest restoration initiatives. In addition, there are currently several ongoing governmental subsidies that support productive systems of small landowners, such as pastureland improvement through fertilization and erosion control. In the long-term this could help regulate and even exclude cattle ranching activities from *Araucaria* forests.

Further investigation is needed to fully understand the multiple factors that affect the long-term maintenance of *Araucaria* populations as well as the interactions between natural and human-driven disturbances, in order to develop and promote effective measures for the restoration and conservation of these endangered unique forests. Although there are still many unsolved questions, our results can help identify urgent policies and promote initiatives to reverse or mitigate the degradation processes that affect *Araucaria* regeneration.

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CAPÍTULO 3

The differential influences of human-induced disturbances on tree regeneration community: a landscape approach

Este capítulo reproduce íntegramente el texto del siguiente manuscrito:

Zamorano-Elgueta, C., Cayuela, L., Rey Benayas, J. M., Donoso, P. J., Geneletti, D., Hobbs, R .J. The differential influences of human-induced disturbances on tree regeneration community: a landscape approach. Ecosphere 5(7), 90.

The differential influences of human-induced disturbances on tree regeneration community: a landscape approach

Abstract

Understanding the processes shaping biological communities under interacting disturbances is a core challenge in ecology. Although the impacts of human-induced disturbances on forest ecosystems have been extensively studied, less attention has been paid to understanding how tree regeneration at the community level responds to such disturbances. Moreover, these previous studies have not considered how these effects change according to major social and environmental factors that can influence forest use at a landscape scale. In this study, we investigate the effects of cattle grazing and selective logging on the composition of tree regeneration communities in relation to forest successional stage and land tenure regime in Chilean temperate forests, a global biodiversity hotspot. We recorded seedlings, saplings and basal area of stumps of tree species (as a surrogate for selective logging), and number of cattle dung pats (as a surrogate for cattle pressure) in 129 25 x 20 m plots in small (<200 ha) and large properties in different successional stages (old-growth, intermediate, secondary forests). The regeneration of the ten more abundant species as predicted by human disturbance, land tenure, forest successional stage, and number of parent trees was modelled using generalised linear models. Predictions for each individual model were made under different scenarios of human disturbance. The predicted regeneration results were assembled and subjected to ordination analyses and permutation multivariate analyses of variance to determine differences in regeneration composition under each scenario. In most cases, best-fit models contained at least one of the explanatory variables accounting for human disturbance. The effects of selective logging on tree regeneration varied depending on land tenure regime, but cattle grazing always exhibited a negative effect. Our results revealed that cattle have a more negative effect on forest regeneration than selective logging, especially in old-growth forests and small properties. Our analytical approach contributes to the understanding of the differential influence of human-induced disturbances on the tree regeneration community at a landscape scale. It can inform conservation policies and actions, which should focus on addressing the main disturbance factors and on developing strategies to conserve the most sensitive species to such disturbances.

Keywords: Cattle grazing; Chile; community composition; forest successional stages; land tenure; low-intensity disturbance; 'predict first, assemble later' modeling; temperate forest; selective logging.

Introduction

The impacts of human-induced disturbances on forest ecosystems have been extensively reported throughout the world (Belsky and Blumenthal 1997, Augustine and McNaughton 1998, Wisdom et al*.* 2006, Baraloto et al*.* 2012, Clark and Covey 2012). Forest degradation by both intense, episodic disturbances (e.g., extensive logging or forest conversion to other land uses) and low intensity, chronic disturbances (e.g., grazing, selective logging or invasion of exotic species and fires) have been studied in several forests (Belsky and Blumenthal 1997, Ramírez-Marcial et al*.* 2001, Stern et al*.* 2002, Timmins 2002, Fisher et al. 2009, Baraloto et al. 2012, Clark and Covey 2012). These disturbances, particularly cattle grazing and selective logging, change species diversity and composition, which may have a major influence on community and ecosystem functioning (Chapin III et al. 1998, Cadotte et al. 2011, Baraloto et al. 2012). However, less attention has been paid to understanding how tree regeneration at the community level responds to such disturbances. Moreover, previous work has not considered how these effects change according to the major environmental and social factors that can influence use of forests at the landscape scale, namely forest successional stage and land tenure regime. For example, in South America, the few published studies have focused on analysing the impacts of introduced mammals on single forest species, such as *Nothofagus dombeyi* (Veblen et al*.* 1989, 1992), *Austrocedrus chilensis* (Veblen et al*.* 1992, Relva and Veblen 1998, Relva and Sancholuz 2000), and *Araucaria araucana* (Zamorano-Elgueta et al. 2012). Understanding the impacts of these disturbances on forest communities and how they vary according to forest successional stage and land tenure provides more comprehensive information to guide conservation efforts. Furthermore, Turner (2010) identified as research priorities the study of disturbances as catalysts of rapid ecological change, interactions among disturbances and relationships between land use and disturbance.

To evaluate the effects of human-induced disturbances on community composition, the study of regeneration of forest species can be particularly informative, as seedlings and saplings respond more rapidly to most low intensity and chronic human disturbances than long-lived adult trees do (Cayuela et al*.* 2006, Helm et al. 2006). Forest regeneration is the process that ensures successive generations of trees (Barnes et al. 1998), and is essential in maintaining the long-term ecological functions and values of forests (Donoso and Nyland 2005). Therefore, investigating the effects of human disturbance on tree regeneration can provide critical information about how these impacts will shape forest community composition and influence ecosystem functioning, as well as the resistance and resilience of these ecosystems to environmental change in the long term (Cadotte et al. 2011).

In this study we investigated (1) the influences of cattle grazing and selective logging on the composition of regeneration communities of a temperate forest ecosystem; and (2) whether these influences vary according to forest successional stage (old-growth, intermediate, and secondary forests) and land tenure regime (small and large properties). Taking into account the successional context is important because forest successional stage is one of the main factors that determine the baseline communities (Nyland 2002). Additionally, property size is also important because frequency and intensity of alterations are often related to this feature (Burschel et al. 2003, Zamorano et al*.* 2012). We investigated these effects in the Coastal Range of southern Chile, which is more exposed to human disturbance compared to the Andes. The Chilean temperate forests have been recognized as a global hotspot of biological diversity (Myers et al*.* 2000, Funk and Fa 2010) and were selected as a target for urgent conservation efforts by the World Wildlife Fund and the World Bank (Dinerstein et al*.* 1995).

Methods

Study area

The study was carried out in the Coastal Range of the Región de los Ríos, southern Chile (39°28′S, 73°16′W; 40°20′S, 73°18′W). The area covers ca. 2,700 km^2 (**Fig. 3.1**) and elevation ranges from 4 to 684 masl. The Coastal Range has abundant endemic flora and fauna, which probably reflects the location of vegetation refuges during the last glacial period (Armesto et al*.* 1995). Evergreen forests are the dominant vegetation type and occupy 79% of the total forest cover in the study area (CONAF et al*.* 1999). The predominant climate is temperate with Mediterranean influence. The average annual temperature and precipitation is 11.9 ºC and 2,500 mm, respectively (Di Castri and Hajek 1976). Soils derive from metamorphic material and granitic rocks, are thin to moderately deep (15-180 cm), have a loamy-clay to clay texture at depth, are very strongly to moderately acidic (pH = 4.6-5.7), and have moderate to high erodibility and low nutrient levels (IREN-CORFO 1964).

Land tenure in the study area is characterized by a mosaic of different land use types, productive activities and local actors, including indigenous communities, national protected areas, large private protected areas, and forest companies. The dominant types of land tenure correspond to small properties (67,656 ha, 25% of the study area) owned by "campesinos" (Spanish name for rural people leaving in small-sized properties with a subsistence economy), and properties owned by forest companies (79,852 ha, 29%) that concentrate the area covered by exotic tree plantations. Although these plantations have increased during the last decades (CONAF-CONAMA 2008), a significant proportion of well-conserved old-growth forest (92,558 ha, 55%) still remains. High frequency and intensity of alterations are typically associated with small properties (i.e. <200 ha as defined by the Chilean laws) owned by peasants or "campesinos" due to the need to achieve levels of production to ensure family subsistence (Zamorano et al*.* 2012). In this study we used small (<200 ha) and large (≥200 ha) properties belonging to private landowners to evaluate logging and cattle grazing.

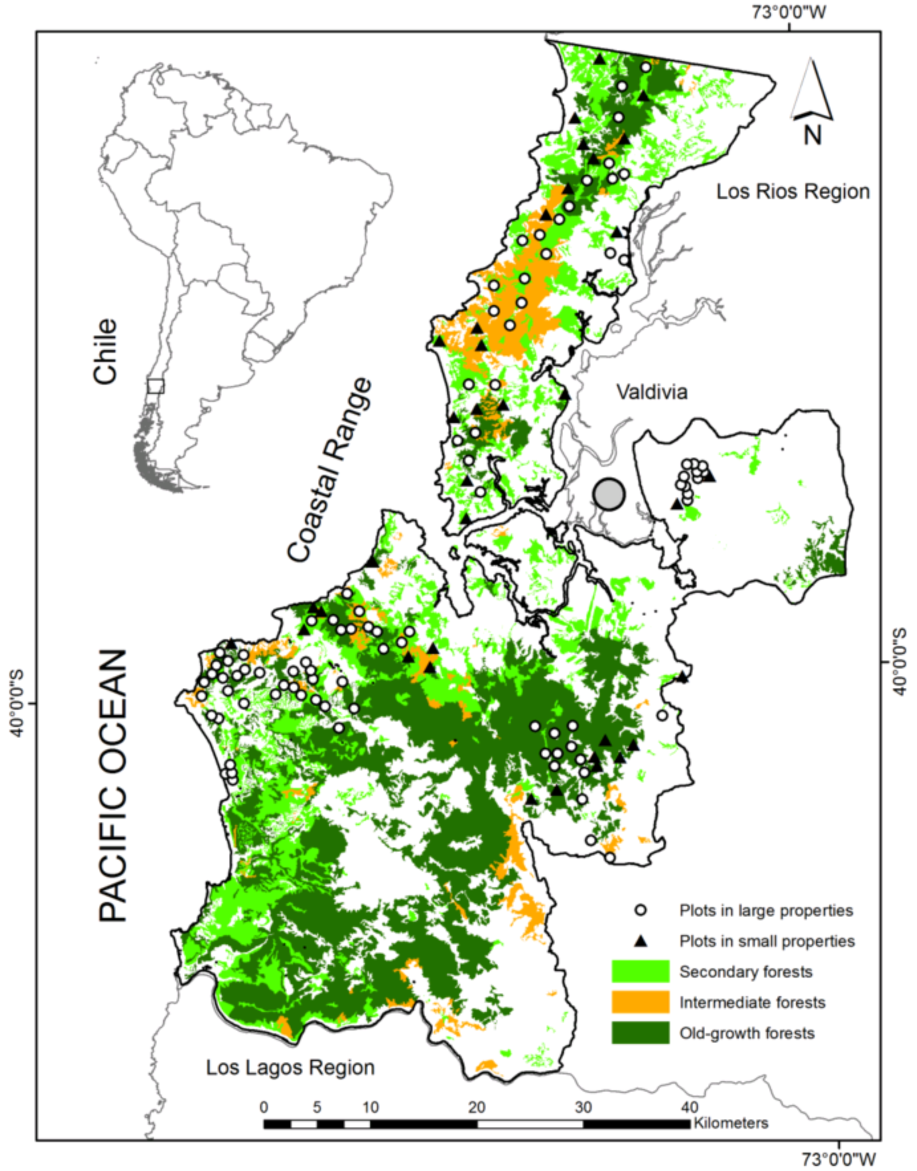

Figure 3.1. Location of the study area and the 129 plots that were surveyed and information related to property size and forest successional stage within the Coastal Range, southern Chile.

Three forest successional stages were defined according to the native forests cadastre, one of the most comprehensive cartographic studies of natural vegetation developed in Chile (CONAF et al*.* 1999): old-growth, old-growth/secondary and secondary forests. Old-growth forests correspond to uneven-aged stands dominated by broad-leaved evergreen tree species, with at least 50% canopy cover, high vertical heterogeneity, structural variability, higher tree species richness, and presence of large canopy emergents (>80 cm dbh, >25 m tall). Secondary forest corresponds to even-aged stands composed mainly of young trees, originating after large-scale disturbance, whether natural or anthropogenic. Old-growth/secondary forest corresponds to intermediate conditions of stand structure, and henceforth we will refer to these forests as 'intermediate forests'.

Sampling methods and forest species selection

A geographical information system (GIS) was used to randomly allocate plots across the study area using a sampling design stratified by land tenure and forest successional stage, and at a minimum distance of 1 km from each other. Once in the field, 129 25 x 20 m plots were established in evergreen forests, at a minimum distance of 200 m from forest edges. The average distance between pairs of plots was 1,600 m. Thirty six plots were located in rural small properties (13 in old-growth forests, 8 in intermediate forest, 15 in secondary forest) and 93 plots in large properties owned by forest companies (31 in old-growth forest, 27 in intermediate forest, and 35 in secondary forest). These properties had an average area of 32 ha. In each plot, a 6 x 6 m subplot located at the plot centre was established, and all seedlings (< 1.3 m tall) and saplings (> 1.3 m tall and < 5 cm in diameter at breast height, DBH) were recorded. Overall, 60% of total regeneration corresponded to seedlings < 0.3 m, which represented recently established regeneration and were ca. < 5 years old, Vita 1977, Uteau 2003, Donoso et al. 2006*a*, *b*, *c*).

Within each plot, two human disturbance-related variables were measured, namely (1) number and basal area of stumps of each tree species without resprouts (a surrogate for selective logging) and (2) number of cattle dung pats (a surrogate for current cattle density and trampling pressure on tree regeneration). Henceforth we will refer to these variables as 'selective logging' and 'cattle intensity index' (CAI, *sensu* Zamorano et al*.* 2012), respectively. Fifty seven percent of stumps were > 10 years old based on qualitative information such as wood decomposition and the stump surface covered by lichens, mosses and fungi. Therefore, it was a reasonable assumption to consider that most of the logging activity pre-dated the establishment of the regeneration trees. In each plot, the number of parent trees of each species, defined as those individuals >5 cm in DBH, was also recorded to account for density-dependent effects. Based on the distribution of tree diameters, we estimated that 45% of the parent trees were > 40 years old (Vita 1977, Uteau 2003, Donoso and Escobar 2006, Donoso et al. 2006*a*, *b*, *c*).

Statistical analyses

The sum of all seedlings and saplings of each species found at each subplot was the response variable in the regeneration models. Not all species were equally present in all plots. In order to fit reliable statistical models, only the most abundant species (i.e. those present in at least 60 plots either in adult or juvenile form) were analyzed in this study. Given the low rate of occurrence of many species, we could only fit statistical models for 10 out of the 33 species recorded in this study. The list of forest species recorded and their occurrence is given in Appendix A.

We followed a 'predict first, assemble later' modelling approach (Ferrier et al*.* 2002) to study the effects of human-induced disturbances on tree regeneration. Regeneration of the 10 selected individual species were modelled one at a time as a function of human disturbances, land tenure, forest successional stage, and number of parent trees. Predictions for each individual model were made under four different scenarios of human disturbance namely, i) undisturbed, ii) high cattle grazing pressure, iii) high selective logging pressure, and iv) both high cattle grazing and selective logging pressures, which were defined based on the range of the observed data. The resulting array of predicted abundances of seedling and saplings (i.e. regeneration) was then assembled and subjected to ordination analyses (non metric multidimensional scaling, NMDS) and permutation multivariate analyses of variance using distance matrices (PERMANOVA), to determine differential effects of human disturbance on regeneration composition in each land tenure regime and forest successional stage.

For the individual species, we used generalised linear models (GLM) with a log-log link function to linearise the observed exponential relationship between the response and the explanatory variables, and a negative binomial error distribution to account for overdispersion. The use of a negative binomial error distribution required estimating an additional parameter, delta, to determine the scaling of the variance to the mean. We used maximum likelihood methods and model selection as an alternative to traditional hypothesis testing (Johnson and Omland 2004). We estimated model parameters that maximized the likelihood of the regeneration measured in the field, given a suite of alternative models. For each species, we examined 85 different nested models including the effects of CAI, selective logging, land tenure regime, successional stage, and number of parent trees, as well as some particular interactions between the explanatory variables. The most complex model takes the form:

$$log(Y_i) = NT_i + log(CAI_i)xLT_ixFS_i + log(SL_i)xLT_ixFS_i + log(CAI_i)xlog(SL_i)$$

where Y_i is the number of seedlings and saplings per hectare in plot *i*; NT_i is the number of parent trees; CAI_i is the cattle intensity index; LT_i is the land tenure regime; FS_i is the forest successional stage; and SL_i refers to selective logging pressure. Interactions among covariates and between covariates and factors would indicate that the effect of a covariate in the response variable changes, depending on the value of the other covariate or factor level.

Alternative models were compared using the Akaike Information Criterion (AIC, Burnham and Anderson 2002). Delta AICc (ΔAICc) was calculated as the difference in AICc between each model and the best model in the set. Models with AICc differences <2 have substantial support (Burnham and Anderson 2002). Therefore, models with ΔAICc >2 were excluded from further calculations. Akaike weights (wi) were calculated for the confidence set of models (ΔAICc <2) to determine the weight of evidence in favour of each model and to estimate the relative importance of each individual parameter in the set of candidate models (w_i). If no single model is clearly superior to the others in a set of models (model with w_i <0.9), a (weighted) model averaging approach should be used (Burnham and Anderson 2002). Hence, we used the entire set of plausible models (ΔAICc <2) to calculate model-averaged estimates for variables included in the confidence set of models and their unconditional standard errors

(SE). This approach reduces model selection bias effects on regression coefficient estimates in all selected subsets (Burnham and Anderson 2002).

Residual plots were explored to assess model assumptions. In addition, spatial correlograms based on Moran's Index were used to explore the autocorrelation of the best-fit model's residuals at different geographical distances (Diniz-Filho et al. 2003). If spatial autocorrelation was detected in a distance class, then one could assume that there were spatially patterned variables not included in the model that contributed to explaining patterns of regeneration for that particular species.

To make model predictions, values for each human-induced variable were set to either zero (no disturbance) or to a value close to the maximum value observed in the field (high disturbance). To account for the stochastic component in each individual model, we added a random error from a negative binomial error distribution using the 'delta' parameter estimated by each individual model. In addition, when there were a set of plausible models for a particular species, we made predictions using only one of the best fit models at a time. This model was randomly selected from the set of best fit models using probabilities proportional to model weights (w_i). Thus, for a single set of values for the explanatory variables, different predictions were possible depending on the random error and the model selected. To account for these sources of variability, we made 250 predictions for the regeneration of each species under each of the four disturbance scenarios in each land tenure regime and forest successional stage.

Next, the resulting array of predicted abundances of regeneration was assembled and subjected to NMDS. Data were square-root transformed and then submitted to Wisconsin double standardization (Legendre and Gallagher 2001). We used the Bray-Curtis dissimilarity distance to compute the resemblance matrix among assembled communities. Finally, differences in regeneration composition of the assembled communities as a function of the two human-induced disturbance variables in each land tenure and forest successional stage were statistically determined by means of PERMANOVA (Anderson 2001). We used the Bray-Curtis distance and 999 permutations for these analyses (permutations of residuals under the reduced model; Anderson and Ter Braak 2003).

All analyses were performed using R (R Development Core Team 2010), including the 'MASS' (Venables and Ripley 2002), 'vegan' (Oksanen et al*.* 2010) and 'MuMIn' (Barton 2013) packages.

Results

Human-induced disturbances on individual species regeneration

The response of forest regeneration to cattle and selective logging was heterogeneous. In seven out of the 10 modeled species there was no single best-fit model (**Table 3.1**). Model assumptions were fulfilled in all cases. Based on the correlograms of model residuals, we only detected significant spatial autocorrelation at some intermediate distance lags for some species, yet the degree of autocorrelation was in general low and followed no particular pattern (detailed results not shown). All species except *Aextoxicon punctatum* and *Amomyrtus meli* were affected by at least one of the two human disturbance-related variables (**Table 3.1**). In some cases, these two variables interacted significantly, further exacerbating (i.e. negative interaction such as for *Gevuina avellana*) or ameliorating (i.e. positive interaction such as *Drimys winteri* and *Laureliopsis philippiana*) the individual effects of these disturbances on species regeneration. Details of parameters for the best fitted models are given in Appendix B

The variable number of parent trees was not included in any of the best-fit models for *Amomyrtus luma* and *L. philippiana* (**Table 3.1**), and had little weight in model-averaged estimates for *D. winteri* and *G. avellana*. For the remaining species, the overall effect of parent trees on regeneration was negative, except for *A. meli* where there was a positive density-dependent effect. As for the effect of land tenure, large properties registered, on average, more regeneration of *A. punctatum*, *D. winteri*, *Eucryphia cordifolia*, *L. philippiana* and *Podocarpus nubigena*, whereas small properties had more regeneration of *A. meli*, *G. avellana* and *Myrceugenia planipes*. Secondary and intermediate forests exhibited low regeneration of *A. meli*, *E. cordifolia* and *M. planipes* as compared to old-growth forests; on the contrary, they exhibited relatively high regeneration of *L. philippiana* (maximum regeneration attained in secondary forests) and *Saxegothaea conspicua* (maximum regeneration attained in intermediate forests).

In some species, the response of regeneration to human disturbance-related variables varied depending on land tenure regime and forest successional stages (**Fig. 3.2**). Such changes occasionally reversed the sign of the effect, from positive to negative or *vice-versa*. Thus, cattle grazing, on one hand, had a negative effect on the regeneration of *D. winteri*, *G. avellana*, *L. philippiana*, *P. nubigena*, and *S. conspicua* in old-growth and secondary forests; and a positive effect on *A. luma* and *E. cordifolia* except in intermediate forests of large properties, and *S. conspicua* in intermediate forests in both small and large properties. Selective logging, on the other hand, had a negative effect on regeneration of *D. winteri*, *P. nubigena*, *G. avellana*, and *M. planipes* in small properties regardless of the forest successional stage. Similar effects were registered for *L. philippiana*, *D. winteri* and *P. nubigena* in large properties. Selective logging had a positive effect on *G. avellana* and *M. planipes* in large properties, and on *L. philippiana* in small properties in intermediate and old-growth forests. Finally, the effect of selective logging on *S. conspicua* was positive in old-growth and secondary forests in both small and large properties.

Table 3.1. Ranking of the generalised linear models of individual species following an AIC-based model selection procedure. ΔAIC and W_i corresponds to AIC differences and Akaike weights (wi). Abbreviation of the model parameters are: NT - the number of parent trees; CAI - the cattle intensity index; LT - the land tenure regime; FS - the forest successional stages; and SL - selective logging pressure. Coefficients in bold indicate the best-fit models for each forest species.

Forest species	Model parameters	AIC	ΔAICc	Wi
Aextoxicon punctatum	NT x LT	**-**	**0.00**	**1.00**
Amomyrtus luma	CAI	**-**	**0.00**	**1.00**
Amomyrtus meli	NT x LT x FS	**-**	**0.00**	**1.00**
Drimys winteri	SL x CAI + SL x LT	**840.04**	**0.00**	**0.53**
	NT + SL x CAI + SL x LT	841.44	1.40	0.26
	SL x LT	841.96	1.92	0.20
Eucryphia cordifolia	NT	**336.75**	**0.00**	**0.32**
	NT x LT	**336.75**	**0.00**	**0.32**
	NT x FS	337.28	0.53	0.24
	NT + CAI	338.72	1.97	0.12
Gevuina avellana	SL	**425.73**	**0.00**	**0.27**
	SL + CAI	425.82	0.09	0.26
	CAI + SL x LT	427.10	1.38	0.14
	SL x LT	427.32	1.59	0.12
	SL x CAI	427.41	1.68	0.12
	NT + SL	427.65	1.92	0.10
Laurelia philippiana	FS x CAI + SL x CAI	**638.46**	**0.00**	**0.32**
	SL x CAI	638.83	0.37	0.26
	FS x SL + SL x CAI	640.01	1.56	0.15
	FS x SL + FS x CAI + SL x CAI	640.08	1.63	0.14
	FS x CAI + SL x CAI + SL x LT	640.21	1.75	0.13
Myrceugenia planipes	SL x LT	**450.72**	**0.00**	**0.38**
	NT + SL x LT	451.64	0.92	0.24
	NT x FS	451.76	1.03	0.23
	NT	452.57	1.84	0.15
Podocarpus nubigena	NT + SL	**409.41**	**0.00**	**0.37**
	NT + SL + CAI	410.36	0.96	0.23
	NT + CAI	410.47	1.07	0.22
	NT x LT	410.89	1.48	0.18
Saxegothaea conspicua	NT + FS x CAI	**408.59**	**0.00**	**0.64**

Changes in regeneration composition of the assembled communities

The regeneration assemblages inferred from individual species model predictions depicted clear patterns in relation to the four disturbance scenarios analyzed, land tenure regimes and forest successional stages (**Fig. 3.3**). The overall variability in predicted species abundances explained by human-induced disturbances ranged from 10.7% to 21.7% (**Table 3.2**). The cattle intensity index explained most of the variability in community composition in all cases, followed by selective logging and the interaction between these two variables (**Table 3.2**). Indeed, there was a clear separation in the ordination diagrams between assemblages with and without cattle disturbance (**Fig. 3.3**). This separation was most clear in old-growth forests and less in secondary forests.

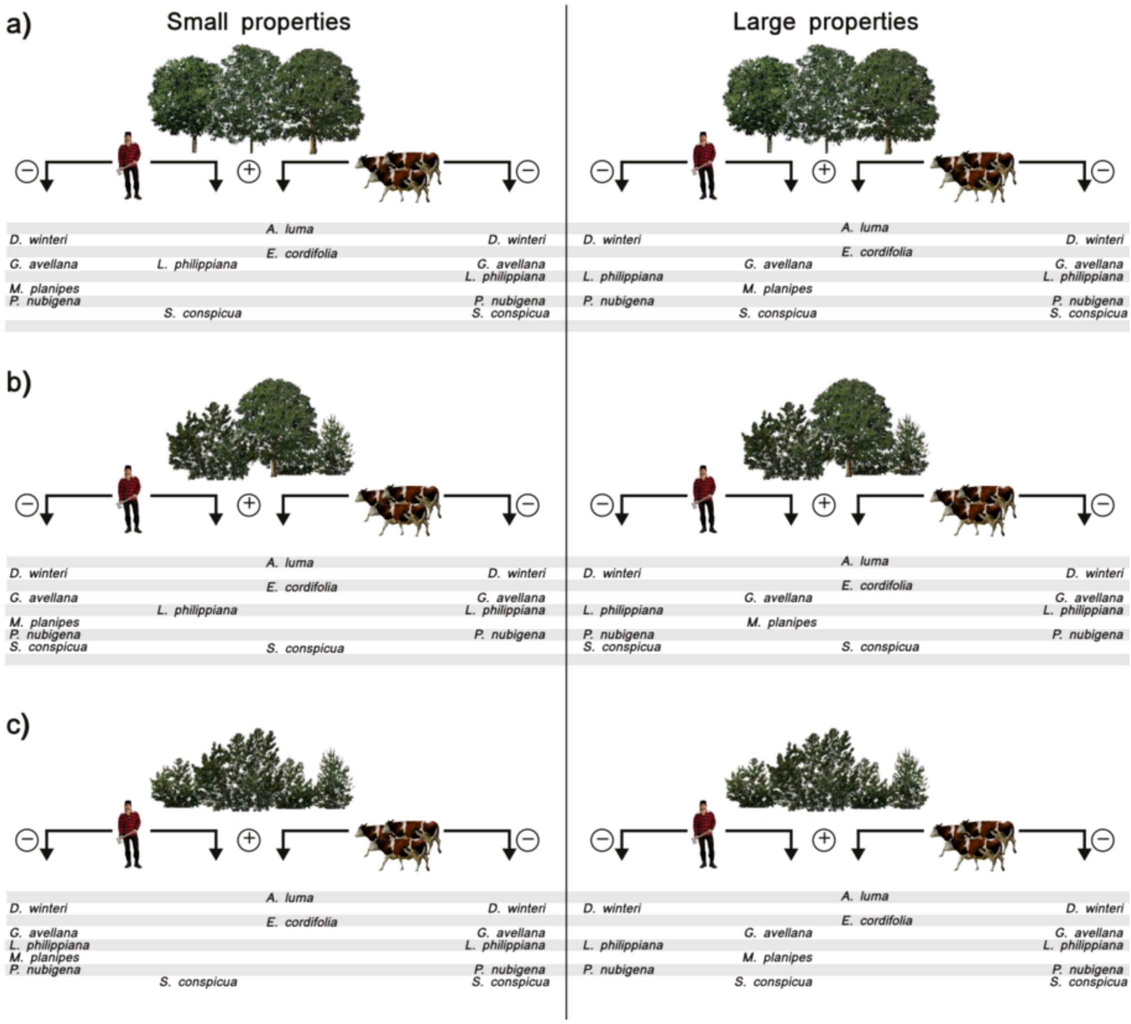

Figure 3.2. Summary of the Cattle Intensity Index and selective logging effects on forest regeneration predicted by land tenure in a) old-growth forests, b) intermediate forests, and c) secondary forests. Symbols represent positive (+) or negative (-) effects on forest species, respectively.

A. luma was always associated with assemblages disturbed by cattle, whereas *L. philippiana* and *D. winteri* were typically associated with undisturbed forests by cattle in all land tenure regimes and forest successional stages. Other species were associated with either disturbed or undisturbed assemblages by cattle depending on the forest successional stage. Such was the case for *S. conspicua*, which was associated with assemblages disturbed by cattle in intermediate forests and undisturbed ones in old-growth forests (**Fig. 3.3**). Associations between specific species and selective logging were not consistent in the different land tenure regimes and forest successional stages, and more difficult to visualise in the ordination diagrams (**Fig. 3.3**).

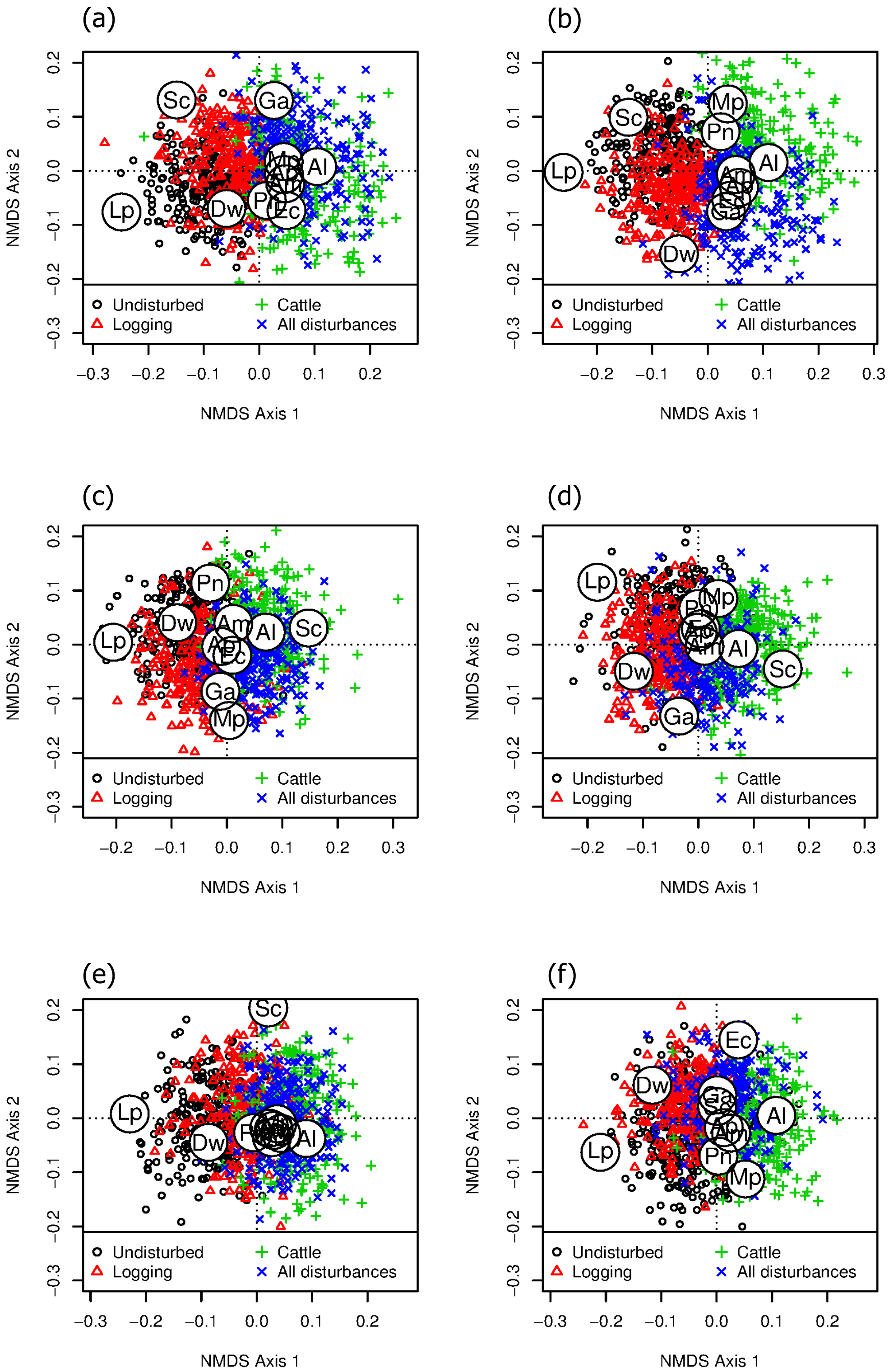

Figure 3.3. Ordination analyses (NMDS) of forest regeneration communities predicted under several scenarios of joint effects of low-intensity disturbances, land tenure and forest successional stage. a) Old-growth forests in large properties; b) old-growth forests in small properties; c) intermediate forest in large properties; d) intermediate forest in small properties; e) secondary forest in large properties and f) secondary forest in small properties. Species abbreviations are the following: Am = *A. Meli*; Al = *A. Luma*; Ap = *A. punctatum*; Dw = *D. Winteri*; Ec = *E. Cordifolia*; Ga = *G. avellana*; Lp = *L. philippiana*; Mp = *M. Planipes*; Pn = *P. nubigena*; Sc = *S. conspicua*. Symbols represent regeneration assemblages in relation to the four disturbance scenarios analysed and main forest species involved.

Table 3.2. Summary of the PERMANOVA to test the effects of the cattle intensity index (CAI), selective logging (SL), and their combined effect (CAI x SL) on the regeneration composition of the assembled communities in each land tenure (property size) and forest successional stage: a) old-growth forests in large properties; b) old-growth forests in small properties; c) intermediate forest in large properties, d) intermediate forest in small properties, e) secondary forest in large properties, f) secondary forest in small properties. Df = Degrees of freedom; SS = Sum of squares; MS = Mean squares; F = F-statistic; R^2 = explained variance.

Parameter	Df	SS	MS	F	R^2	P-value
a) Old-growth forest in large properties						
CAI	1	6.714	6.714	168.218	0.139	0.001***
SL	1	1.011	1.011	25.341	0.021	0.001***
CAI x SL	1	0.872	0.872	21.852	0.018	0.001***
Residuals	996	39.754	0.040		0.822	
Total	999	48.352			1.000	
b) Old-growth forest in small properties						
CAI	1	7.396	7.396	166.961	0.138	0.001***
SL	1	1.583	1.583	35.860	0.030	0.001***
CAI x SL	1	0.354	0.354	8.017	0.006	0.001***
Residuals	996	43.961	0.044		0.825	
Total	999	53.267			1.000	
c) Intermediate forest in large properties						
CAI	1	4.513	4.513	108.875	0.095	0.001***
SL	1	1.111	1.111	26.812	0.023	0.001***
CAI x SL	1	0.703	0.703	16.967	0.015	0.001***
Residuals	996	41.281	0.041		0.867	
Total	999	47.608			1.000	

Discussion

Our study reveals that cattle grazing and selective logging have contrasting effects on the regeneration of different tree species, which ultimately shapes the composition of future communities. These effects were conditioned by land tenure regime and forest successional stages. By adding a random error to our simulated communities under a variety of disturbance scenarios, we predicted a diverse array of future forest composition.

Effects of cattle and selective logging on regeneration communities

Our study suggests that cattle grazing and, to a lesser extent, selective logging, have a major impact on the composition of regeneration communities, and that these impacts are more severe in small than in large properties and in old-growth than in intermediate or secondary forests. Overall, our results suggest negative effects of cattle and selective logging on the regeneration of most of the species studied. Selective logging had a negative influence on more species in small properties. This could be explained by the intensity of tree harvesting necessary to ensure family subsistence among campesinos, mainly through firewood and charcoal production (Burschel et al*.* 2003). On the other hand, cattle grazing had similar negative effects regardless of the land tenure regime and forest successional stage. Cattle raising in the forests studied is mainly by "campesinos" in small properties. However, in large properties, especially in public properties and protected areas, forests are commonly used by neighbouring farmers as a source of fodder and refuge for their cattle, without any control over livestock density or logging intensity, and for firewood production - a pattern followed elsewhere in protected areas (Moorman et al. 2013). Similar conclusions were reported by Zamorano et al*.* (2012) in temperate forests dominated by the monkey puzzle tree *Araucaria araucana*. These results confirmed that cattle grazing can diminish, damage or prevent the recruitment of tree species by browsing and trampling, which can in turn induce changes in species composition (Hobbs 2001, Baraloto et al*.* 2012, Zamorano et al*.* 2012). Selective logging, on the other hand, can cause negative impacts on seedling richness (Farwig et al*.* 2008), tree species diversity (Polyakov et al*.* 2007, Ramírez-Marcial et al*.* 2001), and functional composition of forest communities (Baraloto et al*.* 2012).

Regeneration of shade-tolerant species such as *S. conspicua* and *L. philippiana* was differentially affected by selective logging according to forest successional stage and land tenure. The models showed that selective logging was less intense and the establishment of seedlings was favored in old-growth forests. Seedlings of these species require some shelter to have high survival and growth rates (Donoso et al*.* 2006). In secondary forests of small and large properties and intermediate forests in large properties, regeneration of *L. philippiana* was negatively associated with selective logging, which could be due to higher intensity of logging and cattle grazing. A high proportion of seedlings and saplings of *E. cordifolia* showed damage by grazing (C. Zamorano, *unpublished data*). However, while this is a highly palatable species, its regeneration appears to be positively associated with cattle for all situations evaluated.

Changing microclimatic conditions inside forests might have caused the negative response observed for the pioneer species *D. winteri* in our study area. This species requires permanent soil moisture and abundant organic matter to regenerate (Donoso et al*.* 2007 and references therein). As pointed out by other studies (Farwig et al*.* 2008), disturbed habitat conditions might likewise influence the establishment of late successional species such as *S. conspicua*, *P. nubigena* or *L. philippiana*. In contrast, *A. luma*, a forest species with wider tolerance to several altered conditions, showed a positive response to human-induced disturbances, due mostly to its capacity to reproduce asexually (Donoso and Escobar 2006); this might be one of the reasons why *A. luma* is one of the most common species in temperate evergreen forests of the region (Veblen et al*.* 1981). The competitive ability of this species may be greater under a scenario of permanent human-induced disturbances.

Cattle had a greater influence than selective logging on forest regeneration, especially in old-growth forests, which appear to be more sensitive to human-induced disturbances. Thus, cattle grazing could influence habitat conditions in this forest type, changing their future composition dramatically. Undisturbed old-growth forests or forests associated only with selective logging would be dominated by late-successional species like *S. conspicua*, *A. punctatum* and *L. philippiana*. Instead, the persistent occurrence of either cattle or cattle and selective logging could prevent the establishment of these shade-tolerant and shade-semitolerant species and favor a composition dominated by *A. luma*, *A. meli* and *G. avellana*.

Implications for forest conservation

Understanding the processes shaping biological communities under interacting disturbances is a core challenge in ecology (Moulliot et al*.* 2012). If forests are permanently disturbed by low-intensity disturbances such as cattle grazing and selective logging, their composition will be profoundly altered by loss of biodiversity and changes in the dominance of different species. Effects of altered habitat conditions on forest regeneration could lead to less phenotypic diversity in characteristics such as fruit type, seed mass by area unit and flowering period (Fisher et al*.* 2009). These changes could generate unknown impacts on functional ecosystem properties and on the ecosystem's response to disturbance (Fisher et al*.* 2009). In this context, there is an urgent need to quantify and predict the effects of disturbance on biodiversity patterns to guide conservation efforts and the management of ecological resources (Mouillot et al*.* 2012).

Our analytical approach contributes to the understanding of the differential influences of human-induced disturbances on the tree regeneration community at the landscape scale, with variable land tenure regimes and forest successional stages. Furthermore, it may strongly support conservation policies and actions, which should first focus on addressing the main disturbance factors and on developing strategies to conserve the most sensitive species to such disturbances.

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Appendix 1

List of forest species registered and their occurrence (number of plots where the species was present) for adult, regeneration and all individuals (total occurrence).

Family	Forest species	Occurrence		
		Adults	Regeneration	Total
Myrtaceae	*Amomyrtus meli* (Phil.) D. Legrand & Kausel.	87	112	119
Winteraceae	*Drimys winteri* J.R. Forst & G. Forst.	105	101	116
Monimiaceae	*Laureliopsis philippiana* (Looser) Schodde.	101	91	113
Myrtaceae	*Amomyrtus luma* (Molina) D. Legrand & Kausel.	102	94	110
Proteaceae	*Gevuina avellana* Molina.	86	80	97
Eucryphiaceae	*Eucryphia cordifolia* Cav.	66	51	78
Myrtaceae	*Myrceugenia planipes* O. Berg.	45	70	78
Podocarpaceae	*Saxegothaea conspicua* Lindl.	72	62	78
Aextoxicaceae	*Aextoxicon punctatum* Ruiz. & Pav.	56	69	74
Podocarpaceae	*Podocarpus nubigena* Lindl.	54	62	74
Asteraceae	*Dasyphyllum diacanthoides* (Less.) Cabrera.	41	35	49
Myrtaceae	*Myrceugenia ovata* O. Berg.	41	27	45
Myrtaceae	*Luma apiculata* (DC.) Burret.	33	16	35
Verbenaceae	*Rhaphithamnus spinosus* (Juss.) Moldenke.	21	27	35
Araliaceae	*Pseudopanax laetevirens* (Gay) Franch.	12	18	28
Laureaceae	*Persea lingue* Nees.	19	19	27
Podocarpaceae	*Podocarpus salignus* D. Don.	13	21	24
Fagaceae	*Nothofagus nitida* Krasser.	23	5	23
Cunoniaceae	*Weinmannia trichosperma* Cav.	0	9	23
Thymelaeaceae	*Ovidia pillo-pillo* Hohen. Ex Meisn.	17	11	21
Myrtaceae	*Myrceugenia parvifolia* (DC.) Kausel.	7	18	20
Cunoniaceae	*Caldcluvia paniculata* D. Don.	10	11	17
Proteaceae	*Lomatia dentata* R. Br.	7	12	14
Fagaceae	*Nothofagus dombeyi* (Mirb.) Oerst.	11	4	11
Proteaceae	*Embothrium coccineum* J.R. & G. Forst.	8	6	10
Proteaceae	*Lomatia hirsuta* (Lam.) Diels. Ex J.F. Macbr.	2	4	5
Monimiaceae	*Laurelia sempervirens* (Ruiz et Pav.) Tul.	4	2	4
Myrtaceae	*Tepualia stipularis* Griseb.	4	1	4
Elaeocarpaceae	*Aristotelia chilensis* Stuntz.	1	2	3
Fagaceae	*Nothofagus alpina* (Poepp. & Endl.) Oerst.	1	0	1
Fagaceae	*Nothofagus obliqua* (Mirb.) Oerst.	1	0	1

Appendix 2

Model-averaged estimates, standard errors and relative importance (w_i), for selected variables in the best fitted models of the regeneration of individual tree species. Generalised linear models were used with a log-log link function and a negative binomial error distribution. Estimated coefficients therefore refer to the response variable on a log scale. Abbreviation of the model parameters are: CAI = Cattle intensity index; SL = Selective logging; NT = Number of parent trees. Forest corresponds to old-growth, intermediate and secondary forest. Type of property was defined as large (>200 ha) and small (<200 ha) properties.

Model parameters	Estimate	SE	Relative variable importance (w_i)
Aextoxicon punctatum			
(Intercept)	3.252	0.175	---
NT	-0.335	0.111	1.00
Small properties	-0.684	0.308	1.00
NT x Small properties	0.314	0.112	1.00
Amomyrtus luma			
(Intercept)	2.135	0.163	---
CAI	0.186	0.074	1.00
Amomyrtus meli			
(Intercept)	3.053	0.173	---
NT	0.350	0.223	1.00
Small properties	0.191	0.346	1.00
Intermediate forests	-0.184	0.263	1.00
Secondary forests	-0.086	0.239	1.00
NT x Small properties	-0.366	0.223	1.00
NT x Intermediate forests	0.039	0.015	1.00
NT x Secondary forests	-0.359	0.223	1.00
Small properties x Intermediate forests	0.321	0.505	1.00
Small properties x Secondary forests	0.038	0.481	1.00
NT x Small properties x Intermediate Forests	NA	NA	NA
NT x Small properties x Secondary forests	NA	NA	NA
Drimys winteri			
(Intercept)	3.624	0.233	---
CAI	-0.402	0.154	0.80
SL	-0.234	0.111	1.00
Small properties	-2.463	0.616	1.00
SL x Small properties	0.540	0.228	1.00
SL x CAI	0.151	0.054	0.80
NT	0.005	0.005	0.26
Eucryphia cordifolia			
(Intercept)	2.382	0.264	---
NT	-0.030	0.010	1.00
Small properties	-0.634	0.361	0.32
NT x Small properties	-0.053	0.064	0.32
Intermediate forests	-0.737	0.407	0.24
Secondary forests	-0.629	0.371	0.24
NT x Intermediate forests	NA	NA	NA
NT x Secondary forests	-0.063	0.061	0.24
CAI	0.046	0.077	0.12
Gevuina avellana			
(Intercept)	1.041	0.217	---
SL	0.207	0.091	1.00
CAI	-0.063	0.112	0.51
Small properties	0.518	0.485	0.26
SL x Small properties	-0.297	0.174	0.26
SL x CAI	-0.043	0.056	0.12
NT	-0.002	0.004	0.10

Appendix 2. Continued

Model parameters	Estimate	SE	Relative variable importance (w_i)
Laurelia philippiana			
(Intercept)	2.966	0.317	---
CAI	-1.221	0.426	1.00
Intermediate forests	-0.818	0.454	0.74
Secondary forests	0.329	0.492	0.74
SL	-0.207	0.128	1.00
Intermediate forests x CAI	0.610	0.265	0.59
Secondary forests x CAI	0.485	0.277	0.59
SL x CAI	0.230	0.096	1.00
Intermediate forests x SL	0.001	0.238	0.29
Secondary forests x SL	-0.351	0.226	0.29
Small properties	-1.330	0.645	0.13
SL x Small properties	0.413	0.244	0.13
Myrceugenia planipes			
(Intercept)	1.943	0.392	---
SL	0.256	0.125	0.62
Small properties	0.777	0.578	0.62
SL x Small properties	-0.553	0.225	0.62
NT	-0.039	0.033	0.62
Intermediate forests	-0.738	0.386	0.23
Secondary forests	-0.333	0.332	0.23
NT x Intermediate forests	NA	NA	NA
NT x Secondary forests	0.070	0.035	0.23
Podocarpus nubigena			
(Intercept)	2.467	0.194	---
NT	-0.054	0.017	1.00
SL	-0.182	0.095	0.60
CAI	-0.155	0.100	0.45
Small properties	-0.616	0.332	0.18
NT x Small properties	NA	NA	NA
Saxegothaea conspicua			
(Intercept)	1.595	0.298	---
NT	-0.159	0.104	1.00
CAI	-0.203	0.212	1.00
Intermediate forests	0.927	0.450	1.00
Secondary forests	0.358	0.470	1.00
Intermediate forests x CAI	0.556	0.283	1.00
Secondary forests x CAI	0.149	0.247	1.00
SL	0.032	0.146	0.36
Intermediate forests x SL	-0.331	0.215	0.36
Secondary forests x SL	0.279	0.206	0.36

Plantaciones de *Pinus radiata* y *Eucalyptus* spp, Cordillera de la Costa, Región de Los Ríos.

CAPÍTULO 4

Seeing the glass as half-full or half-empty: a story of native forest replacement by exotic plantations in southern Chile (1985-2011), and how natural regeneration partly compensate these losses

Este capítulo reproduce íntegramente el texto del siguiente manuscrito:

Zamorano-Elgueta, C., Rey Benayas, J. M., Cayuela, L., Armenteras, D., Hantson, S. Seeing the glass as half-full or half-empty: a story of native forest replacement by exotic plantations in southern Chile (1985-2011), and how natural regeneration partly compensate these losses. En revisión en Forest Ecology and Management.

Seeing the glass as half-full or half-empty: a story of native forest replacement by exotic plantations in southern Chile (1985-2011), and how natural regeneration partly compensate these losses

Abstract

In spite a number of studies have reported rates of deforestation and spatial patterns of native forest fragmentation, few have focused on the expansion and spatial configuration of exotic tree plantations and in their role on land cover change. The objective of this study was to analyze the dynamics of land cover change under the hypothesis that exotic tree plantations have caused a major transformation of temperate forest cover in southern Chile during the last three decades. We used three Landsat scenes taken in 1985 (TM), 1999 (ETM+), and 2011 (TM) to quantify land cover change and a set of landscape indicators to describe the spatial land cover configuration. Our results showed that major changes took place mainly as a dynamic conversion among forest, exotic tree plantation and shrubland. During the studied time span, the area covered by exotic tree plantations increased by 168% (20,896 to 56,010 ha), at an annual rate of 3.8%, mostly at the expense of native forest and shrubland. There was a total gross loss of 30% (54,304 ha) of native forest, but a net loss of only 5.1% (9,130 ha) of its initial cover, at an annual net deforestation rate of 0.2%. The difference between gross and net loss of native forest is mostly the result of conversion of shrubland and agricultural and pasture land to secondary forest following natural regeneration. Along the studied years, exotic tree plantations showed a constant increase in patch density, total edge length, nearest-neighbour distance, largest patch index, as well as a highest value for mean patch size in the intermediate year. Native forest exhibited an increase and then a decrease in patch density and total edge length, whereas mean patch size and the largest patch index were lowest in the intermediate year. Overall, the observed trends indicate expansion of exotic tree plantations, and increasing native forest loss and fragmentation, particularly during the 1985-1999 period. Understanding the dynamics of land cover change and the role of exotic tree plantations will help to conservation and restoration strategies of native forest, which is now a binding requirement for forest companies in the region to obtain timber certification.

Keywords: Deforestation, fragmentation, land cover change, temperate forest, spatial patterns.

Introduction

Humans have change the land use and land cover for thousands of years, with important impacts on the environment (e.g. Klein et al. 2011). Human activities and demands are rapidly changing landscapes and ecosystems, with only small areas on the globe not affected (Lambin et al. 2011). Transformation of natural landscapes has eroded ecosystem functions, and habitat loss and fragmentation have increased vulnerability to edge effects and biodiversity loss (Fearnside 2005, Laurance et al. 2006).Therefore, land cover change has been indicated as one of the high priority concerns for research and for the development of strategies for sustainable management (Vitousek 1994). Although important advances have been made, significant gaps in our understanding of the spatial ecology of these changes remain (Iverson et al. 2014).

Notwithstanding large extends of natural forest are lost annually, spontaneous regeneration may counteract deforestation and fragmentation of native forest (Morrison and Lindell 2010, FAO 2011), especially on areas where natural recolonization will be fast due to seed availability, cover of remaining forest, and conserved soils (Prach et al. 2007, Chazdon 2008). However, the recovery from abandoned grassland or shrubland to secondary forest could take a significant time or may not occur (Florentine and Westbrooke 2004). Several factors could act as barriers to forest regeneration, including shortage of tree seeds (Uhl 1987, Bullock et al. 2002), seed and seedling predation (Rey Benayas et al. 2005, Zamorano-Elgueta et al. 2012, 2014), competition with established grasses/weeds (Rey Benayas et al. 2005), and abiotic limitation, such as exhaustion of soil nutrients (Houle et al. 2014), drought and changes in soil physical properties (López-Barrera et al. 2006, Olvera-Vargas et al. 2014).

Land use planning is a complex task with different and sometimes conflicting goals exist regarding management purposes (Iverson et al. 2014). The rapid expansion of tree plantations is one of the most contentious issues in contemporary sustainable development due to the significant environmental and socioeconomic conflicts it is causing (Hartley 2002, Gerber and Veuthey 2011, Kröeger et al 2013). Contrary to the area of native forests, which has declined at an annual rate of 5.2 million hectares per year during the 2000-2010 period, planted forests are expanding and they currently account for ca. 7% of the total forest area worldwide (FAO 2011, Kröger 2013). Three factors have caused the expansion of tree plantations towards increasingly difficult-to-reach areas: depletion of finite resources, particularly timber, natural limits to increasing yields on high-quality land, and development of technical capabilities that make them feasible in cheaper marginal lands (Kröger 2013). Tree plantations are typically established on cleared agricultural land, but they also expand at the expense of native forest, which is an emerging cause of forest loss and fragmentation worldwide (Foley et al. 2005). The two main areas of dramatic, above average plantation expansion increase in the world are South America (67% increase between 1990 and 2010) and Asia and the Pacific (61.6% increase) (Kröger

2013).

Some tree plantations chiefly pursue environmental benefits, including those fostered by the European Community Agrarian Policy (European Commission 2013) and the Chinese Grain-to-Green project (Song et al. 2014). However, most tree plantations are grown primarily for efficient wood production and contribute significantly to the economic growth of many regions, which may also produce substantial changes in natural ecosystems, with impacts on biodiversity and ecosystem services (Hartley 2002). Furthermore, management practices such as periodic clearing of understory vegetation might have more drastic effects than any competitive or allopathic effects of the planted trees (Atauri et al. 2004). The global trend of increasing tree plantation area is likely to continue, especially for the production of biofuels (Kole et al. 2012) and carbon storage (Lindenmayer 2009), while natural forests are in decline and increasingly fragmented (FAO 2010).

In Chile, exotic tree plantations, which are dominated by *Pinus radiata* (D. Don) and Eucalyptus spp., account for 2.3 million ha (INFOR 2013) and spread by 37,000 ha annually (CONAF 2014). The geographic range of southern temperate forest has declined considerably during the last century (Smith-Ramírez 2004), partly as a result of conversion of native forest to other land cover types. These, together with fragmentation of remnant habitats, are two of the main threats to native forest in southern Chile (Echeverría et al. 2006, Lara et al. 2011, Nahuelhual et al. 2012). These temperate rainforests are one of the globally important ecoregions in terms of biodiversity (Myers et al. 2000, Smith-Ramírez 2004, Smith-Ramírez et al. 2007) and the target for urgent conservation efforts by the World Bank, the World Wildlife Fund and other organizations (Dinerstein et al. 1995). In Chile, the last remnants of temperate forest are restricted to the upper elevations in the Andean mountains and the southern section of the Coastal Range, where there are still continuous tracts of forest (Smith-Ramírez 2004). A number of studies have investigated the rate of deforestation and patterns of native forest fragmentation around the world (e.g. Armenteras et al. 2013, Kumar et al. 2014, Dávalos et al. 2014), including Chile (Echeverría et al. 2006, Schulz et al. 2010), but to date very few (e.g. Nahuelhual et al. 2012) have focused on the expansion and spatial configuration of exotic tree plantations and in their role on the dynamics of regional land cover change. Improving our understanding of such dynamics could help mitigate or reverse their impact on forest ecosystems, contribute to land use planning, and design and implement conservation and restoration programs at the landscape scale in southern temperate forests. The objective of this study is to analyze the dynamics of land cover change under the hypothesis that exotic tree plantations have caused a major transformation of temperate forest cover in southern Chile in a recent period of 26 years. To achieve this goal our study attempt to explain (1) the rates and amount of land cover change, (2) the spatial distribution of forest loss and expansion of plantations, and (3) whether natural regeneration compensate the forest loss at the landscape scale.

Methods

Study area

The study area covers ca. 2,700 km^2 of the Chilean Coastal Range (**Fig. 4.1**), including rivers and wetlands, and elevation ranges from 4 to 684 m. It has abundant endemic flora and fauna, which reflect the location of vegetation refuges during the last glacial period (Armesto et al. 1995). Evergreen forests are the dominant vegetation type, and occupy 79% of the total forest cover in the study area (CONAF et al. 1999). The predominant climate is temperate with Mediterranean influence, with a mean annual temperature of 11 ºC and a mean annual precipitation of 2,500 mm. Soils derive from metamorphic material and granitic rocks (IREN-CORFO 1964).

Land tenure is represented by a mosaic of different land cover types, productive activities, and local stakeholders. The dominant types of land tenure correspond to properties owned by forest companies (81,100 ha, 30% of the study area) that concentrate the area covered by exotic tree plantations, private protected areas (52,000 ha, 19.3%), and small properties (46,827 ha, 17%) owned by "campesinos" (Spanish name for rural people with a subsistence economy living in small-sized properties, i.e. <200 ha as defined by Chilean laws). The other major types of land tenure correspond to large properties, i. e. ≥ 200 ha (45,663 ha, 16.97%) and public protected areas (26,000 ha, 9.74%). High frequency and intensity of alterations are typically associated with small properties owned by campesinos due to the need to achieve levels of production to ensure family subsistence (Zamorano-Elgueta et al. 2012).

Image classification

A set of three Landsat images (path 233, row 88) were acquired for the years 1985 (TM), 1999 (ETM+), and 2011 (TM), with a pixel spatial resolution of 30 x 30 m. All images were taken during the dry season (December to February). The images were pre-processed, including geometric, atmospheric and topographic corrections (Hantson and Chuvieco 2011). We defined five classes of land cover: (1) native forest, (2) exotic tree plantation, (3) shrubland, (4) agricultural and pasture land, and (5) bare ground. Exotic tree plantation corresponds to industrial plantations of exotic tree species dominated by *Pinus radiata* (D. Don) and *Eucalyptus* spp. (Reyes and Nelson 2014).

Landscape complexity poses particular challenges for image classification (Cayuela et al. 2006a). There is an inevitably high degree of misclassification, particularly if various categories are interspersed within a small spatial area (Foody 2002) or some of the land cover categories have overlapping spectral signatures (Pedroni 2003). For example, vegetation stages following successional gradients such as shrubland, arboreous-shrubland and forest categories (Echeverría et al. 2006) or forest successional stages such as young, intermediate and old forest (Liu et al. 2014) are usually very similar in their spectral signatures. For this reason, we included within the native forest category the three main forest successional stages defined by the Chilean native forest cadastre (CONAF et al. 1999), i.e. old-growth, old-growth/secondary, and secondary forest.

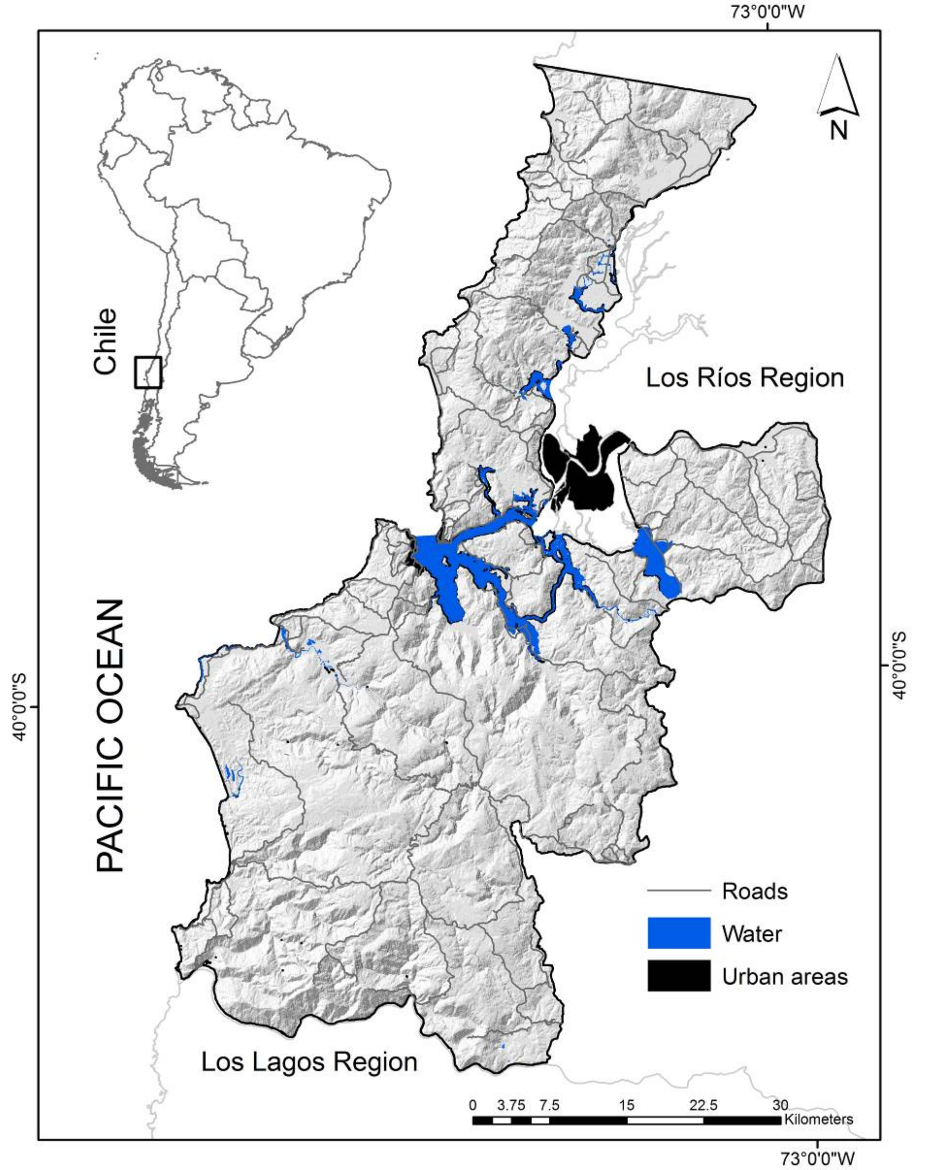

Figure 4.1. Location of the study area within the Coastal Range of southern Chile.

To classify the scenes we used a supervised classification method. Training sites were selected to represent the spectral variability of each land cover classes and were extracted from colour composite images and based on local knowledge (Chuvieco 2010). The signature separability was assessed by the Bhattacharyya distance (Kailath 1967), which was used to analyse the spectral separability of the land cover class signatures before performing the classification. The maximum likelihood algorithm was used to assign probabilities of membership to each class and each pixel was finally assigned to the class of maximum probability (Richards and Jia 2006).

Accuracy assessment

Accuracy assessment of the 1999 and 2011 scenes were conducted using ALOS scenes from 2010 with a pixel spatial resolution of 10 x 10 m, and the most updated version of the cadastre for the Región de los Ríos (CONAF-CONAMA 2008). The cadastre was developed at the 1:50,000 scale, and was derived from aerial photographs and satellite imagery. For the classification of the 1985 scene, we used forest cover maps generated from aerial photograph of 1985 at the 1:60,000 scale (Lara et al., unpublished data). Additionally, we used two maps of land cover for 1985 and 1999 derived from 1986 and 1999 Landsat scenes, respectively (Gonzalez et al. 2005). Sets of 173 and 199 control points were used for the 1985 and 1999 scenes, respectively. The points were overlain on the reference land cover maps and assigned to the respective class. For accuracy assessment of the 2011 image, 198 ground control points were visited in the field during the dry season in 2012. Confusion matrices were constructed to cross-validate the land covers derived from the satellite scenes. Three accuracy measures, namely producer's accuracy, user's accuracy, and overall accuracy, were calculated. Most of the processing work was performed using PCI 7.0 (PCI 2001) and ArcGis version 10 (ESRI 2011).

Cover change and spatial configuration of tree plantations and native forest

To compare the change in cover of exotic tree plantations and native forest for each time interval, we used the compound-interest-rate formula proposed by Puyravaud (2003):

$$Change\ rate = 100/(t_2-t_1) \times \ln (A_2/A_1)$$

where A_1 and A_2 are the cover of tree plantations or native forest at time t_1 and t_2. Both net and gross changes were calculated. Net change represents the difference between the gains and losses in a cover type between two periods. On the other hand, gross change represents the total area modified between two periods. Quantification and comparison of the spatial configuration of fragments was conducted based on the following set of landscape metrics: (a) patch density (number of patches/100 ha), (b) total edge length (km), (c) mean patch size (ha), (d) Euclidean nearest-neighbor distance (m), and (e) largest patch index (%). Landscape spatial indices were computed using FRAGSTATS (version 4.2, Mcgarigal et al. 2012).

Results

Accuracy assessment

Overall accuracy for classification was 73.2%, 83.9%, and 82.9% for the 1985 TM, 1999 ETM+, and 2011 TM scenes, respectively. The lowest values of producer's accuracy were attained in exotic tree plantations for the 1985 and 1999 scenes, and in shrublands for the 2011 scene, whereas the lowest values of user's accuracy corresponded to exotic tree plantations for the 1985 scene, and to shrublands for the 1999 and 2011 scenes (**Appendix 1**).

Changes of land cover

Changes were characterized by a net loss for all land cover classes but exotic tree plantations (**Table 4.1**). Net losses were highest for shrubland (18,867 ha, -6.9% of the study area), followed by native forest (9,117 ha, -3.4%) and agricultural and pasture land (7,161 ha, -2.7%), whereas exotic tree plantations gained 35,093 ha or 13%. These changes were more intense between 1985 and 1999 than between 1999 and 2011 (**Fig. 4.2, Appendix 2**).

Table 4.1. Net change (gain minus loss) for land cover classes in hectare and percentage of the study area.

Cover type	1985-1999						1999-2011					
	Gains		Losses		Net change		Gains		Losses		Net change	
	ha	%	ha	%	Ha	%	ha	%	ha	%	ha	%
Native forest	21,987	8.1	30,908	-11.5	-8,921	-3.3	23,188	8.6	23,397	-8.7	-209	-0.1
Exotic tree plantation	30,245	11.2	6,219	-2.3	24,025	8.9	21,125	7.8	10,036	-3.7	11,089	4.1
Shrubland	12,346	4.6	23,790	-8.8	-11,444	-4.2	7,261	2.7	14,723	-5.5	-7,462	-2.8
Agricultural and pasture land	7,712	2.9	13,710	-5.1	-6,000	-2.2	7,122	2.6	8,385	-3.1	-1,263	-0.5
Bare ground	3,114	1.2	776	0.3	2,338	0.9	414	0.2	2,570	-0.9	-2,155	-0.8

Over the time period analyzed, the main land cover class was dominated by native forest, which decreased from 66.5% of the study area (179,663 ha) in 1985 to 63.2% (170,534 ha) in 2011, and it was 63.2% (170,743 ha) in 1999. Between 1999 and 2011, total native forest cover remained relatively stable (**Fig. 4.2, Appendix 2**). The annual net deforestation rate was 0.2% (351 ha per year) along the whole study period, and higher in 1985-1999 (0.36% or 637 ha) than in 1999-2011 (0.01% or 17 ha per year). Native forest was distributed across the entire study area but concentrated in the southern parts, where it showed a continuous distribution **(Fig. 4.2)**. In the central and eastern parts of the study area, native forest was represented by a higher number of smaller patches. Exotic tree plantations represented 7.7% of the study area (20,896 ha) in 1985, 16.6% (44,921 ha) in 1999, and 20.7% (56,010 ha) in 2011 (**Fig. 4.2, Appendix 2**). In other words, exotic tree plantations increased by 168% from 1985 to 2011, with an annual net gain of 3.8% equivalent to 1,351 ha per year. This rate was highest in 1985-1999 (1,717 ha, 5.5%) than in 1999-2011 (924 ha, 1.8%). In 1985, exotic tree plantations were concentrated in the eastern and central parts of the study area, whereas in 1999 they had spread all over the study area, especially in the northern and central parts (**Fig. 4.2**). This expansion continued in 2011, but at a lower rate.

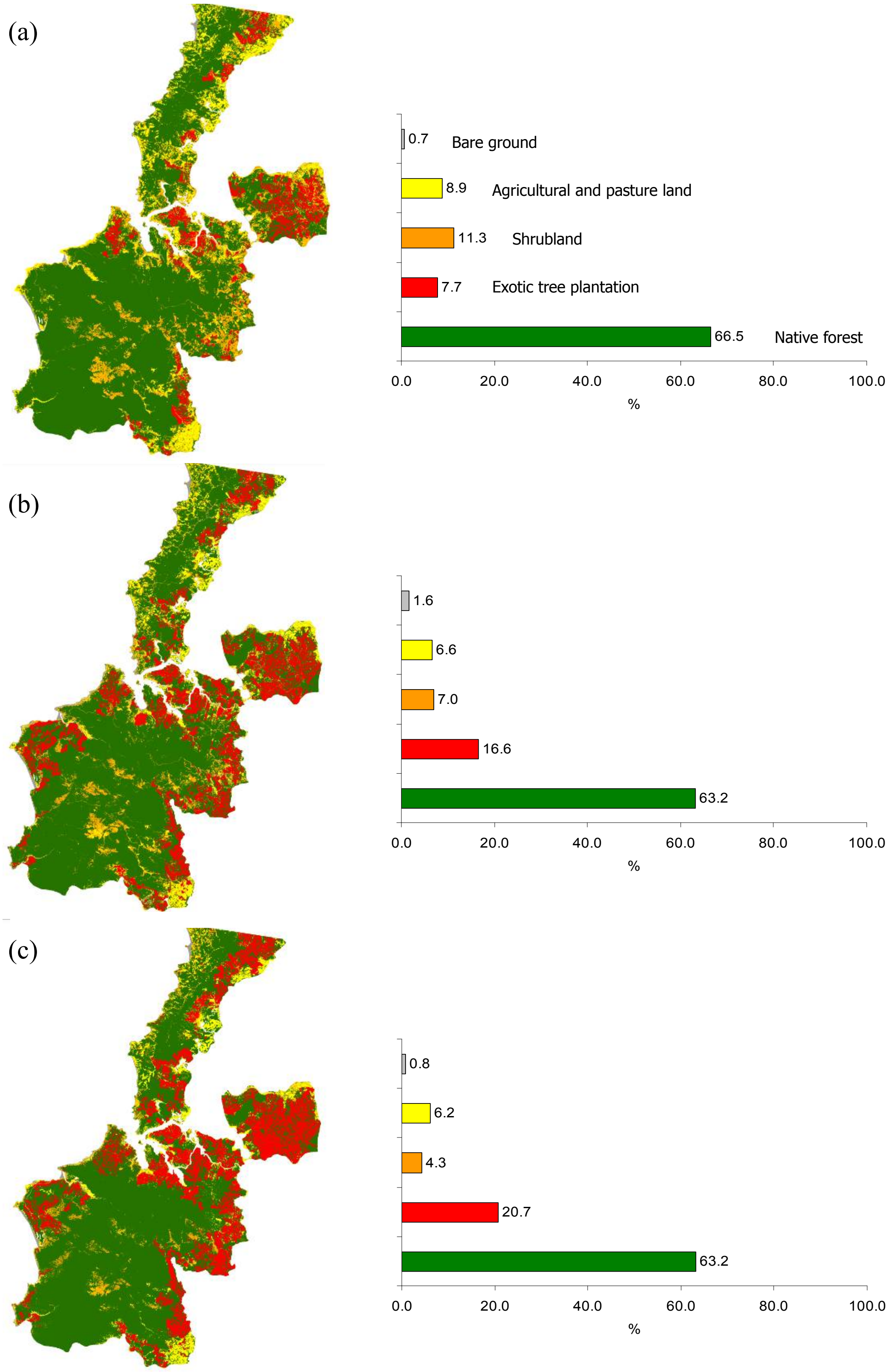

Figure 4.2. Land cover maps based on the classification of TM and ETM+ Landsat images for the years (a) 1985, (b) 1999, and (c) 2011, and comparison of the extents of land cover classes by percentage of the study area.

Shrubland and agricultural and pasture land represented 11.3% (30,461 ha) and 8.9% (23,911 ha) of the study area, respectively, in 1985, 7.0% (19,017 ha) and 6.6% (17,913 ha) in 1999, and 4.3% (11,555 ha) and 6.2% (16,650 ha) in 2011 (**Fig. 4.2, Appendix 2**).

Change trajectories among land cover classes

The major changes occurred among native forest, shrubland, and exotic tree plantation and, to a lesser extent, between these three types of land cover classes and agricultural and pasture land (**Fig. 4.3**). Between 1985 and 1999, changes mainly consisted in the conversion of native forest to exotic tree plantations (7.0% of the study area) and to shrubland (3.2%). Also remarkable in this period was the conversion of shrubland to native forest (4.2% of the study area), exotic tree plantations (2.5%), and agricultural and pasture land (1.7%), and of agricultural and pasture land to native forest (1.9%) and exotic tree plantations (1.5%) (**Fig. 4.3**). Between 1999 and 2011, the major changes were the conversion of native forest to exotic tree plantation (6.2% of the study area), shrubland (1.3%), and agricultural and pasture land (1.1%). Agricultural and pasture land changed mainly to native forest (1.5%), whereas shrubland changed to native forest (3.7%), agricultural and pasture land (0.9%), and exotic tree plantation (0.8%) (**Fig. 4.3**).

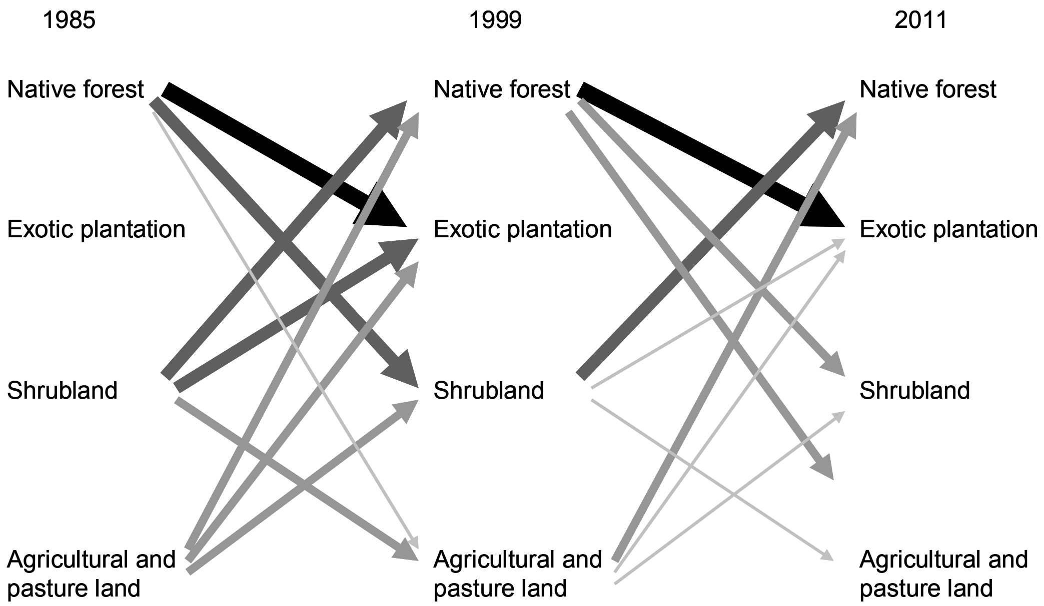

Figure 4.3. Major change trajectories and their contributions to net change among land cover classes in the study area. Thick arrows correspond to net change >6% of the study area, intermediate arrows to net changes between 2.5% and 4.2%, thin arrows to net change between 1% and 1.9%. The thinner arrows correspond to a marginal net change < 1%.

Changes in spatial configuration

Exotic tree plantations showed a constant increase in patch density (**Fig. 4.4a)**, total edge length (**Fig. 4.4b)**, nearest-neighbour distance (**Fig. 4.4d)**, and largest patch index (**Fig. 4.4e)** along the studied years, and a highest value for mean patch size in the intermediate year

(**Fig. 4.4c**). Native forest exhibited an increase and then a decrease in patch density (**Fig. 4.4a**) and total edge length (**Fig. 4.4b**), whereas mean patch size (**Fig. 4.4c**) and the largest patch index (**Fig. 4.4e)** were lowest in the intermediate year and the nearest-neighbour distance hardly changed in the studied period (**Fig. 4.4d)**.

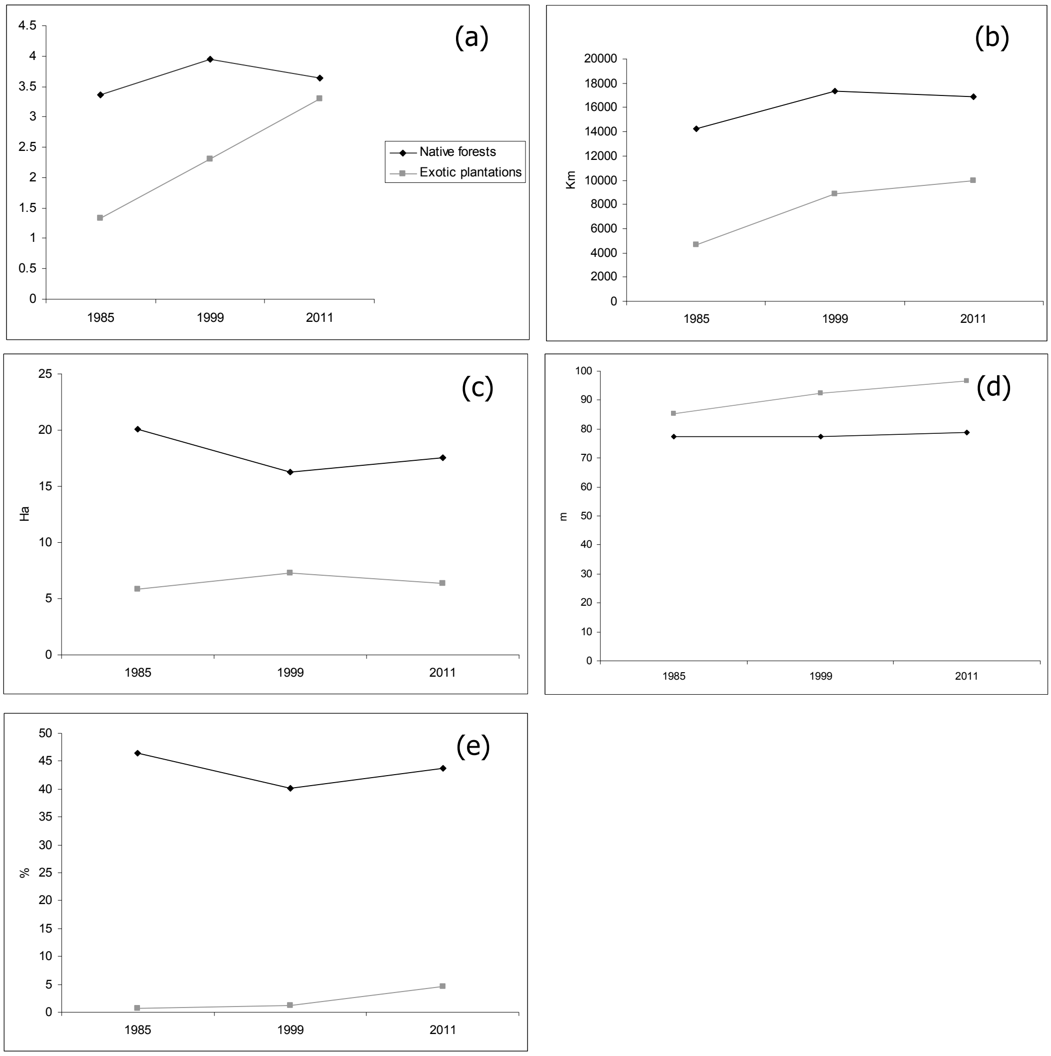

Figure 4.4. Landscape metrics variation of exotic tree plantation and native forest fragments for the years 1985-1999-2011: (a) patch density, (b) total edge, (c) mean patch size, (d) nearest neighbour distance, and (e) largest patch index.

Discussion

Land cover changes

Our study points to three major changes in southern Chilean temperate forests in the recent decades, namely (1) an intense expansion and compactness of exotic tree plantations coupled with (2) a reduction and fragmentation of native forest and shrubland and (3) the natural regeneration of native forest at the expense of shrubland and agricultural and pasture land. Similarly to Echeverría et al. (2006), the rate of increase of exotic tree plantations was the highest of all land cover classes. Exotic tree plantations have increased over two-fold between 1985 and 2011, from 8% of the study area by 1985 to over 20% in 2011. On the contrary, native forest has resulted in a net loss of only 5.1% of its initial cover, at an annual deforestation rate of 0.2%. This discrepancy between gross and net forest loss is mostly the result of conversion of shrubland and agriculture and pasture land to secondary forest. The rate of increment in exotic plantations in our study area are in line with previous results from adjacent region in the Coastal Range of South-central Chile: Echeverría et al. (2006) registered an annual deforestation rate of 4.5% in the period 1975-2000, and Nahuelhual et al. (2012) highlighted the rapid expansion of tree plantations from 5.5% to 42.4% of the landscape, at annual rates of 7.9% and 5.1% in the periods 1975-1990 and 1990-2007, respectively.

In our study region, native forest has been replaced by exotic tree plantations to a large extent and has been degraded by logging for firewood and clearance for livestock and cultivation (Smith-Ramírez 2004, Zamorano-Elgueta et al. 2014). Historically, logging and clearance have been more intense in central-south and central Chile since the European colonization, especially in the Coastal Range (Camus 2006). Here, anthropogenic disturbance represents a serious threat for biodiversity conservation, mainly due to the concentration of human population and the characteristics of the coastal mountains, which, unlike the Andes Range, are outspread in the east-west direction, with altitudes not generally exceeding 1,500 m, thus providing greater accessibility and, consequently, greater vulnerability to the forests (Armesto et al. 1995). These features have triggered a major expansion of exotic tree plantations in the region (Echeverría et al. 2006, Nahuelhual et al. 2012).

In the Coastal Range of Southern Chile, our results showed that land cover change took place as a progressive conversion among forest, exotic tree plantation, shrubland and agricultural and pasture land cover. These patterns are similar to those reported in dryland forest of Central Chile, especially in the coastal zone, where frequent exchanges between pasture and shrubland as well as among pasture, bare ground and agricultural areas were reported (Schulz et al. 2010). However, in Central Chile the expansion of exotic tree plantations did not result in major conversion of native forest. Forest loss was triggered by the conversion of forest to shrubland, mostly driven by a continuous degradation due to permanent grazing pressure, firewood extraction and charcoal production (Armesto et al. 2007). In addition, successional recovery of dry forests is largely constrained by factors such as water availability, soil erosion, human-induced fires and limited regeneration capacities of forest species as compared to shrubland species (Armesto et al. 2007). On the contrary, in our study area these conditions are less restrictive and do not impede natural regeneration (Albornoz et al. 2013). This is evidenced by the large proportion of shrubland converted to native forest over the whole study period, in contrast to deforestation patterns in other regions (Vogelmann et al. 2012, Armenteras et al. 2013). Natural forest regeneration could have been facilitated by the creation in 2002 of a 50,000-ha private protected area in the region, which was established in an area historically altered by firewood production, intensive logging of high-value species such as the conifer *Fitzroya cupressoides* (González 2004) and plantation of exotic trees. In addition, the two major Chilean forest companies have been certified recently by the Forest Stewardship Council (FSC), and have started several initiatives of forest conservation and restoration to achieve the goals underpinned by this certification. Restoration activities in the properties of these companies may spread native forest in the future. However, replacement of native forests by tree plantations is still a common practice as indicated in this study and in line with previous works (Echeverría et al. 2006, Schulz et al. 2010, Nahuelhual et al. 2012).

Native forest and shrubland were found to be highly dynamic, with large gains and losses over the study period, as compared to other land cover classes. We registered exchanges between native forest and shrubland as well as between shrubland and agricultural and pasture land, which took place in small patches scattered throughout the study area. Such exchanges resulted in a net loss of native forest and shrubland due to the expansion of exotic tree plantations. This pattern of change is still motivated by the afforestation policies developed by the Chilean government to promote fast-growth tree plantations since the 70s, including important subsidies funding between 75% and 90% of the afforestation costs (Reyes and Nelson 2014).

Spatial configuration of changes

Our analysis showed a constant increase in patch density, largest patch index, total edge length and nearest-neighbour of exotic tree plantations along the studied period of time, whereas mean patch size exhibited an increase in the first period and then a decrease. These results indicate their expansion as both continuous and non-continuous patches, i.e.that have been established as isolated patches and close to previously existing plantations, distributed along the study area. On the other side, these metrics increased for native forests in the first assessed period but decreased during the second one. The greatest decline in the largest native forest patch size coincided with the time period with highest annual forest loss, as registered elsewhere (Cayuela et al. 2006b, Echeverría et al. 2006, Schulz et al. 2010). Echeverría et al. (2006) suggested that the constant action of deforestation led to a decline in patch density of native forest in southern Chile. Decline in patch density and other metrics in our study area could be explained by the effect of passive conversion of shrubland to native forest and the decreasing annual rate of exotic tree plantation.

Trends in the spatial configuration of exotic tree plantations and native forest are useful to explain some changes in the pattern of these cover classes that have occurred since the 1980s. While exotic tree plantations increased in the first studied period, native forest became gradually more fragmented. In 1999, exotic tree plantations became the second dominant land cover class, and replaced the dominance of shrubland and agricultural and pasture land registered in 1985. Native forest was primarily surrounded by shrubland in the beginning of the study period, whereas exotic tree plantations dominate the neighboring areas of native forest patches at present.

Application to sustainable land use planning

Deforestation and native forest fragmentation in the Coastal Range of Región de Los Ríos was found to be less intensive than in other regions of Chile (Echeverría et al. 2006, Schulz et al. 2010, Nahuelhual et al. 2012). In addition, we found a relatively high rate of passive conversion of shrubland to native forest. These results could be attributed to the better conservation status of the studied landscape as compared to other regions of Chile. However, the still occurring expansion of exotic tree plantations and loss and fragmentation of native forest could lead to microclimatic changes at the forest edges that might facilitate the spread of exotic species towards the interior of the forest fragments (Murcia 1995). Whereas forest loss include both old-growth and secondary forests, native forest established after secondary succession exhibit several differences in diversity, structure, and functionality respect to old-growth and old growth/secondary forests (Lu et al. 2003), and in the study area, they have been shown to respond differently to human impacts (Zamorano-Elgueta et al. 2014). Different successional stages also provide different levels of ecosystem services including those related to soil (Moran et al. 2000) and water (Lara et al. 2009). These alterations will affect humans in ways that go beyond the immediate land-use situation (Turner et al. 1993).

Our study is a first step to understanding ecological processes underpinning forest changes in southern Chile, and not as an end itself (Li and Wu 2004). Although forest successional stages represent dynamic changes between old-growth, old-growth/secondary and secondary forest, with similar spectral signature, identifying these native forest categories would complement the results of our study. This could improve our knowledge on the ecological influence of land cover change and exotic tree plantations in landscape ecology, including areas where less intense but constant increase of this type of cover is present. These results could support conservation or restoration strategies, including the definition of priority areas for such restoration or conservation actions. Identifying priority areas would increase the efficiency and impact of available resources to design, plan and establish forest restoration programs, where interventions will produce the greatest benefits for e.g. maintaining and enhancing biodiversity and provision of ecosystem services. Future research should rather focus on the management of external influences on forests, for example the expansion and ecological impacts of exotic tree plantations.

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Appendix 1

Confusion matrices for Dempster-Shafer classifications of (A) 1985, (B) 1999, and (C) 2011 Landsat scenes. Abbreviations are the following: NF = Native forests; EP = Exotic tree plantation; SHR = Shrubland; APL = Agricultural and pasture land; BG = Bare ground.

(A)

Land cover map	Ground verification points						
	1985 TM						
	NF	EP	SHR	APL	BG	Total	User's accuracy
NF	73	8	9	8	0	98	74.5
EP	8	15	0	0	0	23	65.2
SHR	7	0	20	1	0	28	71.4
APL	3	1	2	18	0	24	75.0
BG	0	0	0	0	0	0	0.0
Total	91	24	31	27	0	173	
Producer's accuracy	80.2	62.5	64.5	66.7	0.0		
Overall classification accuracy: 73.2%							

(B)

Land cover map	Ground verification points						
	1999 ETM+						
	NF	EP	SHR	APL	BG	Total	User's accuracy
NF	91	13	0	2	1	107	85.1
EP	8	38	1	0	0	47	80.8
SHR	1	1	18	2	1	23	78.3
APL	1	0	2	16	0	19	84.2
BG	0	0	0	0	3	3	100.0
Total	101	52	21	20	5	199	
Producer's accuracy	90.1	73.1	85.7	80.0	60.0		
Overall classification accuracy: 83.9%							

(C)

Land cover map	Ground verification points						
	2011 TM						
	NF	EP	SHR	APL	BG	Total	User's accuracy
NF	89	14	4	1	0	108	82.4
EP	11	50	0	0	0	61	81.9
SHR	1	1	10	1	0	13	76.9
APL	2	0	0	12	1	15	80.0
BG	0	0	0	0	1	1	100.0
Total	103	65	14	14	2	198	
Producer's accuracy	86.4	76.9	71.4	85.7	50		
Overall classification accuracy: 82.9%							

Appendix 2

Transition matrices for different land cover changes in southern Chile for the periods (A) 1985-1999 and (B) 1999-2011. Abbreviations are the following: NF = Native forest; EP = Exotic tree plantation; SHR = Shrubland; APL = Agricultural and pasture land; BG = Bare ground.

(A)

	NF		EP		SHR		APL		BG		Total 1985	
	Ha	%	Ha	%	Ha	%	Ha	%	Ha	%	Ha	%
NF	148,756	57.9	19,035	7.4	8,562	3.3	2,648	1.0	662	0.3	179,663	66.5
EP	5,175	2.0	14,676	5.7	711	0.3	250	0.1	83	0.03	20,896	7.7
SHR	11,250	4.4	6,880	2.7	6,671	2.6	4,652	1.8	1,008	0.4	30,461	11.3
APL	5,335	2.1	4,187	1.6	2,826	1.1	10,201	3.9	1,361	0.5	23,911	8.9
BG	227	0.1	142	0.06	246	0.1	161	0.06	1,153	0.4	1,930	0.7
Total 1999	170,743	66.5	44,921	17.5	19,017	7.4	17,913	6.9	4,268	1.7	256,861	100.0

(B)

	NF		EP		SHR		APL		BG		Total 1999	
	Ha	%	Ha	%	Ha	%	Ha	%	Ha	%	Ha	%
NF	147,346	57.36	16,712	6.51	3,538	1.38	3,073	1.20	74	0.03	170,743	63.2
EP	8,352	3.25	34,885	13.58	872	0.34	762	0.30	50	0.02	44,921	16.6
SHR	10,024	3.90	2,119	0.83	4,294	1.67	2,440	0.95	140	0.05	19,017	7.0
APL	4,106	1.60	1,860	0.72	2,268	0.88	9,528	3.71	151	0.06	17,913	6.6
BG	706	0.27	434	0.17	583	0.23	847	0.33	1,698	0.66	4,268	1.6
Total 2011	170,534	66.39	56,010	21.81	11,555	4.5	16,650	6.48	2,113	0.82	256,861	100.0

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CAPÍTULO 5

Restoring forests for biodiversity and ecosystem services: A spatial multicriteria approach to identify priority areas

Este capítulo reproduce íntegramente el texto del siguiente manuscrito:

Zamorano-Elgueta, C., Geneletti, D., Orsi, F., Cayuela, L., Rey Benayas, J. M. Restoring forests for biodiversity and ecosystem services: A spatial multicriteria approach to identify priority areas. En revisión en Landscape and Urban Planning.

Restoring forests for biodiversity and ecosystem services: A spatial multicriteria approach to identify priority areas

Abstract

Forest restoration can play an increasingly important role in enhancing biodiversity and ecosystem services, as 0.6% of the world's forest is lost annually. In this study, we focus on identifying priority areas for forest restoration for maintaining and enhancing biodiversity, and the provision of ecosystem services. We used a multicriteria approach to assess the ecological suitability and socioeconomic feasibility of forest restoration. The method was structured into four main steps aimed at: (i) defining potential areas for restoration; (ii) assessing the suitability of restoration based on ecological criteria (namely biodiversity, water provision, and soil protection); (iii) assessing the feasibility of restoration based on socioeconomic criteria (namely accessibility, pressure on forest, and land tenure); and (iv) combining suitability and feasibility maps to identify priority areas. This methodology was applied to a case study in Chilean temperate forests, a global biodiversity hotspot. Forest degradation was defined based on empirical evidence of alterations in forest regeneration by cattle grazing and selective logging. The area to be restored was defined according to the best suitability and feasibility areas for forest restoration according to different perspectives, e.g. a biodiversity oriented, a land tenure oriented, or a balanced perspective. This approach will not only allow practitioners to understand where to restore, in order to enhance the ecological values of a region, but also to define the feasibility of restoration activities in the medium and long term.

Keywords: Deforestation; feasibility; forest degradation; land tenure; suitability; value functions.

Introduction

Deforestation and forest degradation are recognized among the major causes of biodiversity loss and global environmental change (Laestadius et al., 2011; Sasaki & Putz, 2009). Forest cover is lost at a global rate of 0.6% per year (Hansen et al., 2010). About 130 million hectares of forest were lost during the last decade, of which 40 million hectares were primary forests (FAO, 2010). In contrast to deforestation, which is defined as forest conversion to other ecosystem type, forest degradation implies the existence of some forest cover having a reduced function of the original one (MEA, 2005). For instance, forest degradation by low-intensity chronic disturbances (e.g. grazing, selective logging, invasion of exotic species and fires) change species diversity and composition, with major influence on community composition and functioning (Chapin III et al., 1998; Cadotte et al., 2011; Baraloto et al., 2012). Deforestation and forest degradation adversely affect millions of people who depend, fully or in part, on forest goods and services at a local, regional and global scale (FAO, 2011). These services include carbon storage, climate and water flow regulation, clean water provision, and maintenance of soil fertility (MEA, 2005).

To mitigate or reverse these processes, various global agreements, commitments, and initiatives in the recent decades have established an imperative for ecosystem restoration, which is now recognized as a global priority for countries and communities alike (Aronson & Alexander, 2013). For instance, one of the priority outcomes of the 2012 UN Rio +20 Conference on Sustainable Development was a target to restore 150 million ha of degraded forest by 2020 (IUCN, 2012). A major goal of ecological restoration is the reestablishment of the characteristics of an ecosystem, such as biodiversity and ecological function that were prevalent before degradation. A meta-analysis of restoration assessments indicated that ecological restoration increased provision of biodiversity and ecosystem services across a wide variety of ecosystems including forests (Rey Benayas et al., 2009). It has been estimated that, on a global scale, more than two billion hectares of forest land that has been either cleared or degraded offers opportunities for restoration (Laestadius et al., 2011).

Identifying priority areas would increase the efficiency and impact of available resources to design, plan and establish forest restoration programs, where interventions will produce the greatest benefits. The prioritisation problem has been addressed in a variety of ways (Mittermeier et al., 1998). To identify priority areas for forest restoration, conservation scientists and planners face the challenge of developing ecologically sound methods that can be applied at different spatial scales so that the results are relevant to land use planning (Geneletti, 2008; Moilanen et al., 2011).

Multicriteria Analysis (MCA) methods have been developed to support decision-making processes that involve the comparison of possible alternatives against a set of different objectives and criteria (Belton & Steward, 2002). They are typically used for dealing with planning situations in which one needs to holistically evaluate different decision alternatives, especially by the multiplicity of decision criteria and conflicting interests affecting decision-making (French et al., 2009). Thus, many of the challenges of today's complex forest management planning can be addressed through MCA methods (Mills & Clark, 2001; Kangas & Kangas, 2005). On the one hand, MCA-based approaches have been proposed to identify the most suitable and feasible areas for reforestation at the landscape scale (Uribe et al., 2014), and for restoration of ecosystem services at the watershed scale (Trabucchi et al., 2014), among others. On the other hand, previous studies have defined potential restoration areas based in deforestation processes (Orsi & Geneletti, 2010), or local perception of forest degradation identified through interviews (Ianni & Geneletti, 2010). To date, no empirical approach has been developed to define degraded forests as a restoration target. In addition, the problem of developing methods to identify priority areas for maintaining and enhancing biodiversity and ecosystem services through forest restoration at the landscape-scale level is less frequently addressed in the scientific literature (Trabucchi et al., 2014).

The present study develops a MCA approach to identify priority areas for forest restoration interventions targeted at maintaining the provision of biodiversity and relevant ecosystem services, namely soil conservation and water provision. The approach considers both the suitability and the feasibility of restoration interventions. Suitability of land is defined in terms of their ability for maintaining the provision of biodiversity and ecosystem services. Feasibility areas are defined according to the socioeconomic context in order to increase the success of restoration activities. The developed approach is illustrated with a case study in the Coastal Range of the Región de los Ríos, southern Chile.

Study area

The area covers ca. 2,700 km^2 (**Fig. 5.1**) and elevation ranges from 4 to 684 m. The Coastal Range has abundant endemic flora and fauna (Smith-Ramirez et al. 2007), which reflect the location of vegetation refuges during the last glacial period (Armesto et al., 1995). Evergreen forests are the dominant vegetation type, and occupy 79% of the total forest cover in the study area (CONAF-CONAMA, 2008). The predominant climate is temperate with Mediterranean influence. Soils derive from metamorphic material and granitic rocks (IREN-CORFO, 1964). Land tenure is represented by a mosaic of different land use types, productive activities and local stakeholders. The dominant types of land tenure correspond to properties owned by forest companies (81,100 ha, 30% of the study area) that concentrate the area covered by exotic tree plantations, private protected areas (52,000 ha, 19.3%), small properties (46,827 ha, 17%) owned by "campesinos" (Spanish name for rural people living in small-sized properties, i.e. < 200 ha as defined by Chilean laws, with a subsistence economy), large properties, i. e. ≥ 200 ha (45,663 ha, 16.97%), and public protected areas (26,000 ha, 9.74%).

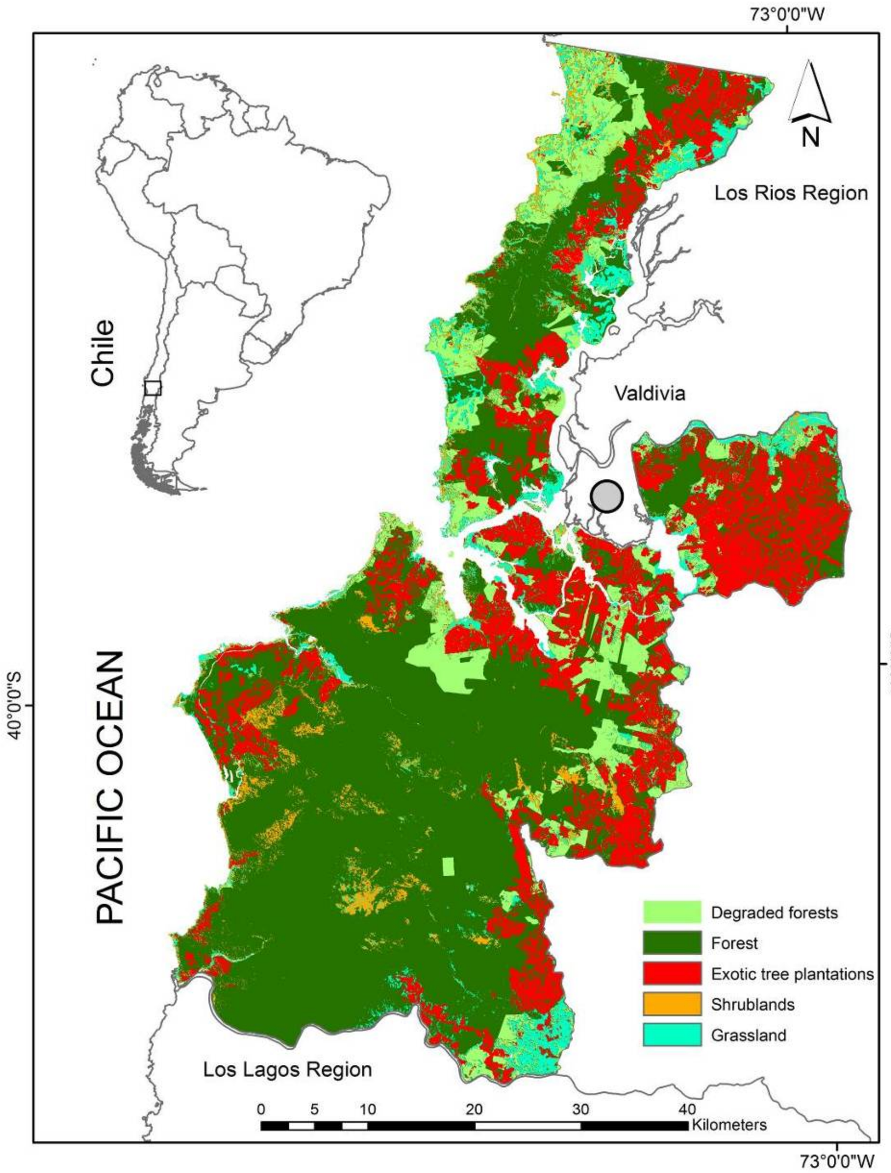

Figure 5.1. Location of the study area, major land covers types and degraded forest within the Coastal Range, southern Chile.

Methods

The method was sequentially structured as following: (i) defining potential areas for restoration; (ii) assessing the suitability of restoration based on ecological criteria (namely biodiversity, water provision, and soil protection); (iii) assessing the feasibility of restoration based on socioeconomic criteria (namely accessibility, pressure on forest, and land tenure); and (iv) combining suitability and feasibility maps to identify priority areas for forest restoration (**Fig. 5.2**).

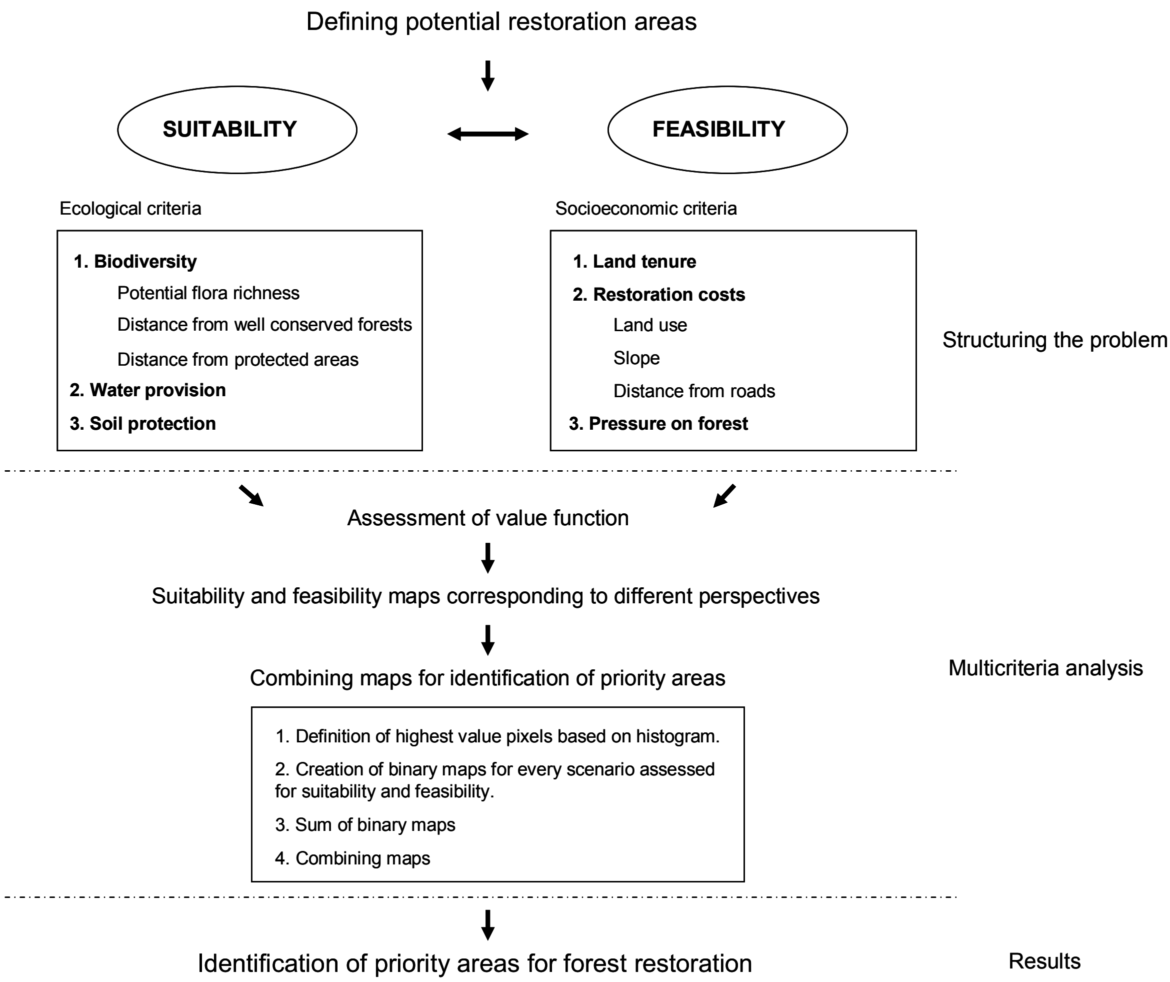

Figure 5.2. Flow chart of the methodological approach proposed in this study. The method was structured into the following main steps: (i) defining potential areas for restoration; (ii) assessing the suitability of restoration based on ecological criteria; (iii) assessing the feasibility of restoration based on socioeconomic criteria; and (iv) combining suitability and feasibility maps to identify priority areas for forest restoration.

Defining potential restoration areas

We considered both deforested areas and degraded forests as potential areas for forest restoration. The native forest along the Coastal Range of central and southern Chile, has been historically replaced by agricultural land, grasslands, shrubland, and exotic tree plantations (Echeverría et al., 2006; Schulz et al., 2010), with profound impacts on the ecosystem service provisions (Little et al., 2009; Lara et al., 2009; 2013). In this study, we considered all these land cover classes as deforested areas. Areas were defined using a Landsat image (path 233, row 88) for the year 2011 (TM) with a pixel resolution of 30 x 30 m. The image was pre-processed, including geometric, atmospheric and topographic corrections, and then classified using supervised maximum likelihood into the following land cover types: native forest, exotic-forest plantations, shrubland, and agricultural/grassland (**Fig. 5.1**). More details related to image processing can be found in **Appendix 1**. We considered as degraded forest all native forests found on small properties of "campesinos", as cattle grazing and selective logging have been reported here to have major impacts on forest regeneration (Zamorano-Elgueta et al., 2014). The suitability and feasibility of restoration was assessed in the deforested areas and degraded forests.

Assessing the suitability of restoration

To assess the suitability of restoration, we identified those areas that play a major role for biodiversity conservation, soil conservation and water provision.

(a) Biodiversity

Three subcriteria were used to assess the suitability for forest restoration in terms of biodiversity conservation: potential flora richness, distance to well conserved forests, and distance to protected areas. These subcriteria were selected based on literature review (Ianni & Geneletti, 2010; Orsi & Geneletti 2010; Trabucchi et al., 2014) and data availability for the study area, as described below.

To estimate the potential flora richness, we assembled single predicted species ranges, following the "predict first, assemble latter" principle proposed by Ferrier and Guisan (2006). For this purpose, 129 randomly allocated plots of 25 x 20 m were sampled in the evergreen forests (the dominant native forests) in 2012 (see Zamorano-Elgueta et al. 2014 for a further description of the sampling design). Average distance between plots was 1,600 m. All flora species were recorded in each plot. Based on presence data from field surveys, we produced probability models of species distribution in the study area. We selected the following environmental covariates to model species distributions: i) climate: temperature (average temperature of the coldest month), and rainfall (average rainfall of the driest month); ii) topography: elevation, slope, and aspect; iii) soil properties: soil pH, soil depth, and drainage; and iv) land cover. Most of these data were obtained from the Sistema de Información Territorial de la Región de Los Ríos (http://www.idelosrios.cl/clienteDescarga_idegore/ 02 feb 2014). In order to generate reliable statistical models, only the most abundant species (i.e. those present in at least 10 plots) were analyzed. The analyses were performed at the species level using the maximum entropy (Maxent) algorithm, a machine learning approach that has been applied widely in recent years to model species niches and distributions based on presence-only data (Phillips et al., 2006; Phillips & Dudik, 2008; Wisz et al., 2008). Each single model was converted from a continuous suitability map to a presence/absence binary map using a threshold criteria (Jiménez-Valverde and Lobo 2007; Freeman and Moisen 2008). These maps were summed to obtain a richness model, and elaboration of a presence/absence matrix to represent species composition (Benito et al. 2013).

Distance from forests was considered as another subcriterion because areas around existing forests are a restoration priority due to proximity to reservoirs of native species (WCMC, 2000).

Likewise, distance from protected areas was used to identify areas that play a major role in biodiversity conservation, because protected areas are a representative sample of the regional biodiversity and provide protection from external threats (Margules & Pressey, 2000).

(b) Soil protection

Soil protection was estimated through a potential erosion map developed by the *Centro de Información de Recursos Naturales of Chile* (CIREN), which is based on a model that integrates soil physical characteristics, topography and climate variables, and has a spatial resolution of 30 m. This map reflects the potential soil erosion, which was categorized in five classes based on land-use intensity: low, moderate, severe, and very severe, other uses. According to this map, the main classes of potential soil erosion in the study area correspond to very severe (46%), severe (34%) and moderate (12%). Further information about this procedure can be found in CIREN (2010).

(c) Water provision

Water provision was estimated using the relationship between mean annual direct runoff coefficient and native forest cover (%) in each watershed, as proposed by Lara et al. (2009) in southern Chile. This relationship follows the equation:

$$Y_i = 0.2271 \times FC_i + 0.2725$$

where Y_i is the mean annual direct runoff coefficient (quickflow/precipitation), and FC_i is the native forest cover (%) in a watershed. Watersheds were defined from high-resolution digital elevation data (ASTER GDEM, available at http://gdem.ersdac.jspacesystems.or.jp).

Assessing the feasibility of restoration

The feasibility of restoration was assessed by considering three criteria: land tenure, accessibility, and pressure on forest. The rationale for selecting these criteria is that restoration interventions are in general more feasible in areas where the land tenure is stable (i.e., property is not likely to change), where access is easier and restoration is less expensive, and where the pressure on forest resources is less (Sunderlin et al. 2005). For the purposes of this study, the land tenure was classified in terms of property size and ownership, features that could influence the success of restoration activities. Forest companies certified by the Forest Stewardship Council (FSC) have started several initiatives of forest restoration as a way to achieve the goals underpinned by this certification. Restoration activities in these properties could increase their medium and long-term success, unlike non-certified companies. In addition, we considered small properties (< 200 ha), and large properties (≥ 200 ha) as separated classes of land tenure. High frequency and intensity of forest alterations are typically associated with small properties owned by "campesinos" due to the need to achieve levels of production to ensure family subsistence (Zamorano-Elgueta et al. 2012, 2014). On public properties, neighbouring farmers commonly used forests as a source of fodder and refuge for cattle, and for firewood production, without any control over livestock density or logging intensity. Based on this features, we propose the following classification of tenure types in the study area: land owned by forest companies certified by the FSC; protected areas (public and private); large properties (> 200 ha); other land owned by forest companies; sustainable small properties; public properties; unsustainable small properties; and land with uncertain property rights (**Table 5.1**). Sustainable small properties were classified based on the sustainability threshold proposed by Reyes (2004), which considered a standard family of five members and a minimum sustainable area of 55 ha. This threshold was estimated integrating economic (e.g. balance between intra and extra-property incomes, estimation of the potential sustainable productivity), social (e.g. local organization, social cohesion), and environmental features (e.g. forest and grassland productivity, size of the property and other land uses). Forest companies certified by the FSC and land with uncertain property rights represent the highest and lowest feasibility classes for forest restoration, respectively.

Table 5.1. Values assigned to convert into quantitative range qualitative criteria. Lower values correspond to maximum feasibility and the higher ones to minimum feasibility.

Land tenure classes	Value	Land use classes	Value
Land owned by forest companies certified by the Forest Stewardship Council (FSC)	1	Degraded forests	1
Protected areas (public and private)	2	Shrubland	2
Large properties (> 200 ha)	3	Agricultural/grassland	3
Other land owned by forest companies	4	Exotic tree plantations	4
Sustainable small properties (> 55 ha and < 200 ha)	5		
Public properties	6		
Unsustainable small properties (< 55 ha)	7		
Land with uncertain property rights	8		

Accessibility was assessed as a combination of land use, slope and distance from roads. Land use corresponds to the land cover classes defined as potential restoration areas, i.e. degraded forests, shrubland, agricultural/grassland, and exotic tree plantations, and was assessed considering if the land use conversion is easier and less expensive according to the type of cover. For example, the lowest and highest restoration costs are represented by degraded forest and exotic plantations, respectively. Thus, degraded forest or shrubland with low slopes and close to roads are more feasible in terms of implementing successful restoration activities.

Pressure on forest was defined in terms of people access as a combination of distance from urban areas, population density, and slope. Areas in close proximity to cities and towns show a high concentration of human activities that demand resources from the surroundings. Thus larger settlements put more pressure on forests. Access of people represents a source of disturbance and can undermine the success of a restoration plan.

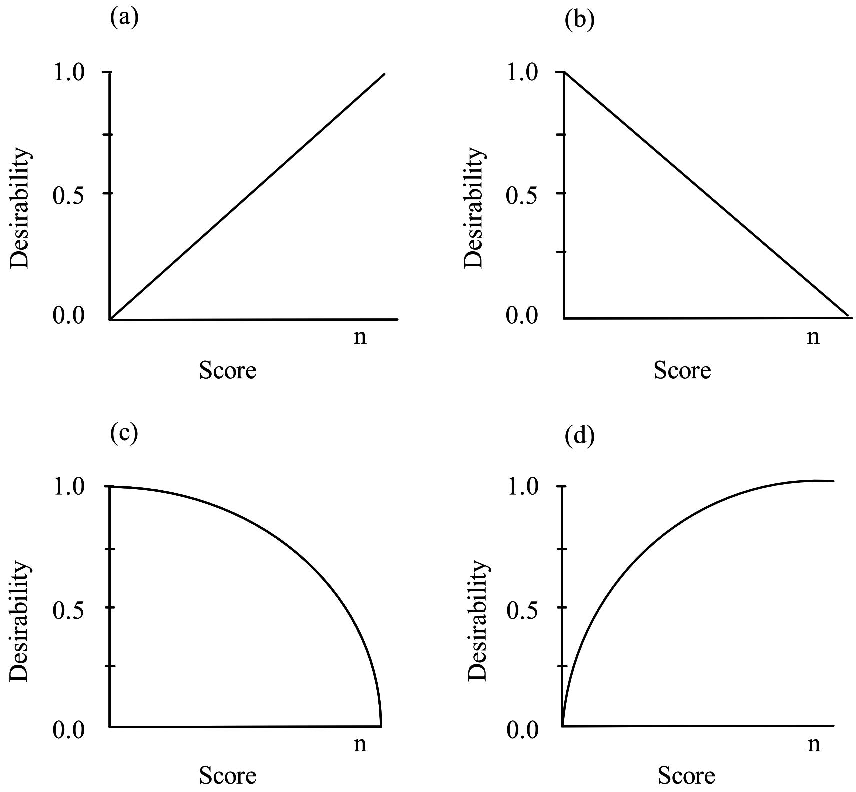

Figure 5.3. Value functions used for the different ecological and socioeconomic criteria: (a) a positive linear relationship (soil protection, potential biodiversity); (b) a negative linear relationship (land use, slope, water provision, distance from forests, distance from protected areas, land tenure); (c) a cuadratic negative relationship (distance from roads); and (d) a cuadratic positive relationship (pressure on forest).

Generating suitability and feasibility maps

Generating suitability and feasibility maps required the combination of the previously generated maps, whose value range and measurement units were different. In order to overcome this issue, all maps were converted to a 0-1 range through the assessment of value functions. A value function was assessed for each criterion in order to make all maps comparable. Value functions transform the original score of a given criterion into dimensionless values that range between 0 and 1, where 0 corresponds to minimum desirability and 1 to maximum desirability (Geneletti, 2005). These value functions selected for this study belong to four main categories, depending on whether the relationship between desirability and the original score was linear (**Fig. 5.3a, b**) or cuadratic (**Fig. 5.3c, d**), and whether maximum desirability was attained at high (**Fig. 5.3a, d**) or low original scores (**Fig. 5.3b, c**). Linear relationship reflects that value score and the corresponding indicator measurement change in constant terms (e.g. soil protection, slope). Cuadratic relationship represents a non constant variation between the variables, e.g. pressure on forests, where cuadratic negative relationship reflects that smaller values are more desirable than larger ones. Criteria that are not expressed through a quantitative scale, but through classes (land tenure, land use) have been converted into the 0-1 range by using look-up tables (**Table 5.1**).

After having converted each criterion map into the 0-1 range, overall suitability and feasibility maps were generated by means of weighted summation of the relevant criterion maps. Given that the assignment of weights is a subjective procedure and that this study did not rely on consultations with stakeholders or decision-makers to derive weights, we generated four scenarios, both for suitability and feasibility, corresponding to different perspectives (**Table 5.2**). Each perspective weighted higher one particular criteria (0.5 weight) than the others (0.25 weight), but we also included one perspective where all criteria were assigned the same weight, i.e. balanced scenario (0.33 weight) (**Table 5.2**).

Identifying priority areas for forest restoration

Individual criterion maps were combined by means of multicriteria analysis (MCA). First, binary maps were created for each perspective by assigning a value of 1 to the 10% most suitable/feasible pixels and a value of 0 to all other pixels. Second, the four binary maps for suitability and feasibility were summed, generating maps with values ranging between 0 (i.e. pixels never identified a priorities) and 4 (i.e. pixels identified as priorities under all perspective). Third, priority areas for restoration were assumed those that had been selected as priorities under all perspective in both suitability and feasibility (i.e. pixel value equal to 4). The basic assumption that guided the identification of priority areas for forest restoration was that restoration on a site should only be attempted if the site is sufficiently suitable and likely to let restoration succeed.

Results

Potential restoration areas spread on 115,500 ha (42.78% of the study area), of which 86,563 ha (32%) were deforested areas, and 26,894 ha (10%) were degraded forests. Deforested areas and degraded forest were concentrated in the north, central and eastern parts of the study area (**Fig. 5.1**).

In general, cells of the suitability map presented a broad range of values. These ranged between 0 and 0.94 for the water provision and biodiversity perspectives, and between 0 and 0.93 for the soil protection and balanced perspectives **(Fig 5.4)**. Major peaks were observed for biodiversity in the 0.5-0.7 interval, for water provision in the 0.3-0.75 interval, and for soil protection in the 0.47-0.65 interval. The balanced perspective presented a slight concentration of cells between 0.5 and 0.6.

Table 5.2. Assignment of weights to the different criteria under different perspectives of (a) suitability and (b) feasibility.

a)

	Perspective				
	Criteria	Biodiversity	Water provision	Soil protection	Balanced
Suitability	Biodiversity	0.5	0.25	0.25	0.333
	Soil protection	0.25	0.25	0.5	0.333
	Water provision	0.25	0.5	0.25	0.333

b)

	Perspective				
	Criteria	Restoration cost	Pressure on forest	Land tenure	Balanced
Feasibility	Restoration cost	0.5	0.25	0.25	0.333
	Pressure on forest	0.25	0.5	0.25	0.333
	Land tenure	0.25	0.25	0.5	0.333

Feasibility values ranged between 0 and 0.82 for the land tenure and restoration cost perspectives, and between 0 and 0.65 for the pressure on forest perspective **(Fig 5.5)**. Cells values were concentrated in the intermediate class for all perspectives that were evaluated. Major peaks were found around 0.35-0.40 and 0.6 for land tenure, 0.55 for restoration cost, 0.4 for pressure on forest, and 0.5 for the balanced perspective. Land tenure and the balanced perspective showed cells distributed in a broader class values with respect to pressure on forest and restoration cost. The summed area with highest values of suitability in all the evaluated perspectives was 53,193 ha (19.7%), whereas it was 59,136 ha (21.9%) for high feasibility values **(Fig 5.6)**. A large part of the territory showed 0 values in all perspectives, which mostly corresponded to well conserved forests **(Fig 5.6)**. These areas correspond mostly to old-growth forests, which are concentrated on the southern region of the study area.

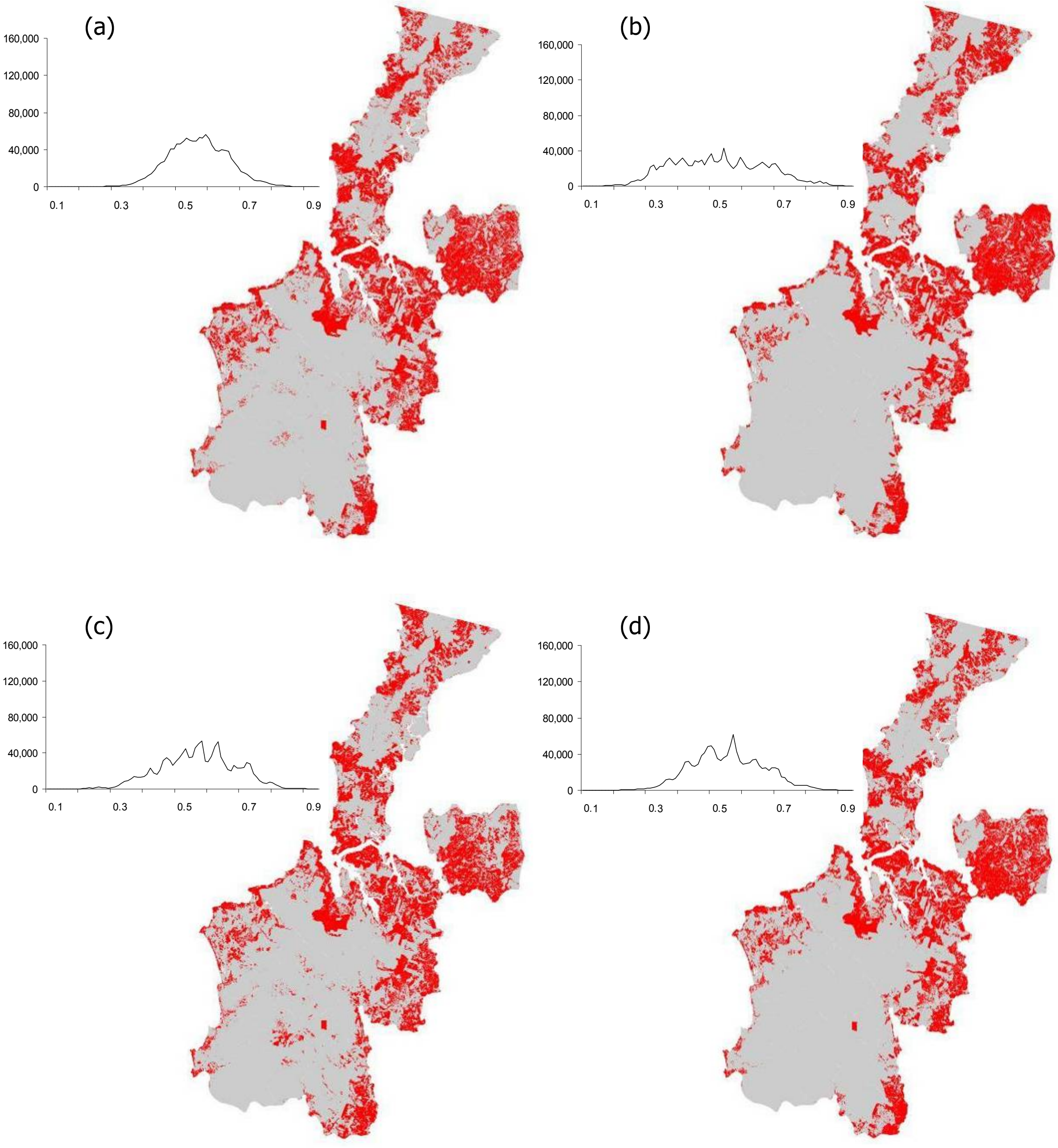

Figure 5.4. Suitability maps and cell value range for the following perspectives: (a) biodiversity, (b) water provision, (c) soil conservation, and (d) balanced scenario, where 0 corresponds to minimum desirability and 1 to maximum desirability.

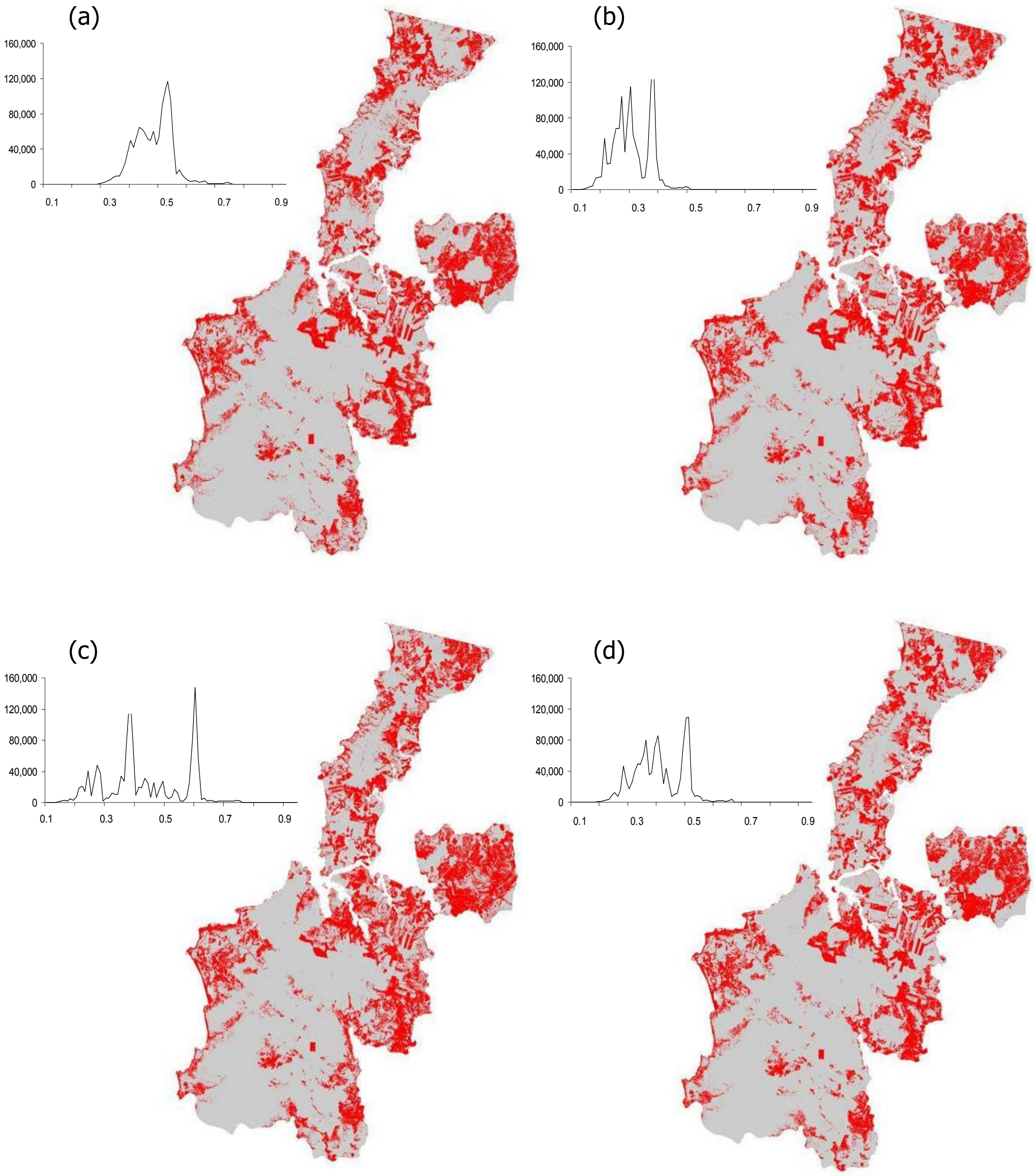

Figure 5.5. Feasibility maps and cell value range for the following perspectives: (a) restoration cost, (b) pressure on forest, (c) land tenure, and (d) balanced scenario, where 0 corresponds to minimum desirability and 1 to maximum desirability.

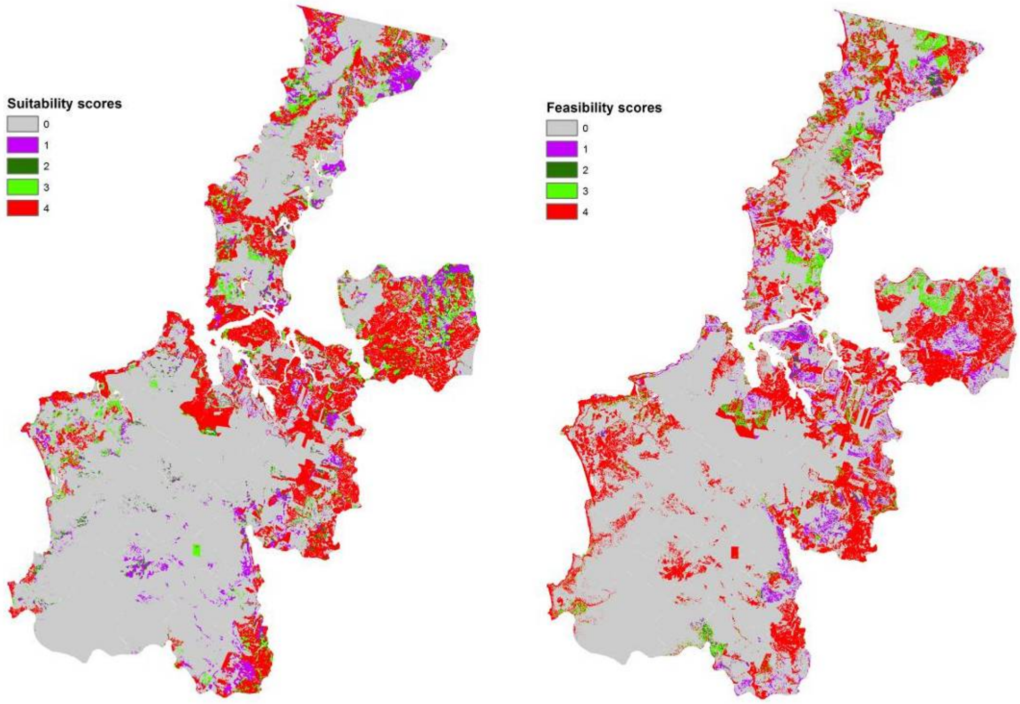

Figure 5.6. Suitability and feasibility scores for forest restoration after summing the four suitability (Fig. 5.4) and feasibility maps (Fig. 5.5), respectively, for the different perspectives, where 0 indicates minimum, and 4 maximum suitability/feasibility.

The priority areas for forest restoration were distributed across the entire study area, but concentrated in the central and eastern parts of the region **(Fig 5.7)**, where they show a continuous distribution. In the northern and southern parts, priority areas were represented by several smaller patches of land. The total amount of priority areas for restoration accounted for 28,985 ha (10.7% of the study area). The priority areas correspond to deforested areas (20,052 ha, 7.4%) and degraded forests (8,933 ha, 3.3%) near well-conserved forests, in land owned by forest companies certified by the FSC (15,817 ha, 5.85% of the study area), sustainable small properties (8,781 ha, 3.25%), and protected areas (2,345 ha, 0.87%), with high biodiversity potential, located in watersheds characterized by low runoff coefficients, with severe potential erosion, in more accessible areas, exposed to low pressure on forest, and with low cost of restoration activities.

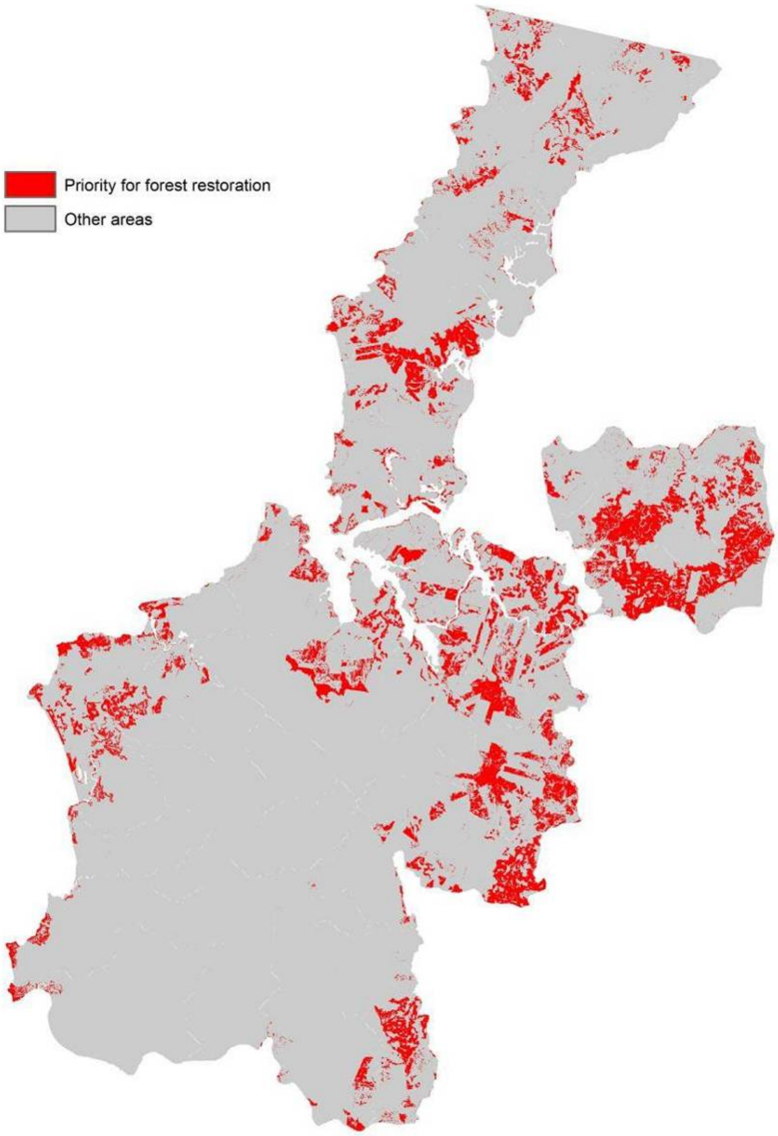

Figure 5.7. Priority areas for forest restoration combining the highest value pixels from both suitability and feasibility maps (Fig. 5.6). Selected areas correspond to pixels that presented suitability or feasibility values under the four scenarios assessed, provided that they are deforested areas and degraded forests.

Discussion

The proposed method involves several innovations: an empirical definition of degraded forests as potential restoration areas, assessment of suitability in terms of biodiversity and provision of ecosystem services, and an explicit evaluation of feasibility according to socioeconomic criteria, including land tenure and pressure on forest. Previous studies at the landscape-scale level have, to our knowledge, not developed methods of prioritization for maintaining and enhancing ecosystem services (e.g. Ianni & Geneletti, 2010; Uribe et al., 2014). Furthermore, less attention has been paid to assess potential restoration areas based on empirical definition of degraded forests. The definition of forest degradation, based on alterations of forest regeneration, would help to mitigate or reverse early degradation processes, and thus increase the efficiency of restoration activities. Restoring degraded forests can be faster and imply lower costs than restoration of deforested areas, especially for passive restoration on areas where natural recolonization will be fast due to seed availability, cover of remaining forest, and conserved soils (Prach et al., 2007; Chazdon, 2008).

The assessment of the ecological suitability of the territory in terms of biodiversity and key ecosystem services and of the feasibility according to socioeconomic criteria, will aid to increase the success of restoration actions. The identification of the most suitable and feasible areas for forest restoration using higher value pixels has been proposed by previous studies (Hirzel et al., 2006; Orsi & Geneletti, 2010). As is defined, these values represent the minimum ecological and socioeconomic requirement for an area to become a forest restoration priority, provided that they are deforested areas and degraded forests. In a similar approach to identify restoration priorities, Orsi and Geneletti (2010) addressed spatial multicriteria analysis in subsequent steps, considering ecological objectives and measures as the basis for the analysis, whereas socioeconomic objectives and measures allowed for the refinement of the outcome. However, the authors defined priority for reforestation in terms of potential for biodiversity conservation, and considered aspect and elevation heterogeneity as proxy variables to estimate tree species richness. Our approach used stacked species distribution models for estimating potential flora richness, generated from an extensive field dataset. This played an important role in assessing the need for restoration according to ecological variables. A significant number of studies have revealed the role of biodiversity for the supply of ecosystem services (Rey Benayas et al., 2009; Isbell et al., 2011). Biodiversity represents the foundation of ecosystems that, through the services they provide, affect human well-being.

Socioeconomic context limits the implementation of restoration activities. Firewood demand, land tenure and plantation of exotic trees are major socioeconomic conditions that limit restoration feasibility. Urban areas are highly demanding of firewood production because it represents the main energy source in southern Chile, both for residential and industrial use (Burschel et al., 2003). In addition, disturbances of higher intensity on forests are associated with the dominant subsistence farming system, in which cattle grazing increases forest degradation by reducing the regeneration potential of some tree species (Zamorano-Elgueta et al., 2012; 2014), especially on properties with smaller surfaces (Reyes 2004). On the other hand, land tenure represents a key aspect to assess feasibility of restoration, due to the heterogeneous needs and socioeconomic conditions associated. A large part of the study area is characterized by the presence of exotic tree plantations, which are associated to severe impacts on water provision, in particular during the dry summer season (Lara et al., 2009). The management of exotic plantations, mainly through extensive clear cutting, leaves the soil unprotected for several years until the new plantation is established. It is essential that sustainable forest management ensures the retention of significant canopy cover to prevent soil erosion and foster nutrient retention and recycling (Pérez, 1999). This is particularly critical as restoration priorities are concentrated in land owned by forest companies. Moreover, outside protected areas, exotic tree plantations surround most of native forest remnants.

Our approach integrates aggregation techniques for the different criteria in order to develop a complete analysis to define the best areas for forest restoration, based on biodiversity and ecosystem services provision. This could support the implementation of restoration initiatives, considering the different contexts, stakeholders, and needs present in a territory. Moreover, the method provides a useful framework that could be modified through a more participatory process at all levels of the modeling steps and through the integration of other ecological, social or economic features.

Conclusions

The proposed methodology in the present study provides an integral and innovative approach to identify priority areas for forest restoration at the landscape scale and to assess the ecological suitability and the socioeconomic feasibility of forest restoration. However, the method proposed is not static. Even though priority areas were defined following different perspectives, this methodology could be adapted to achieve particular goals, and/or modified through the involvement of stakeholders and expert judgement. This could be done, for example, in the definition of the highest value pixels for the best suitability and feasibility areas for forest restoration, in the selection of ecological or socioeconomic criteria, and/or the assignment of weights to reflect different needs. This approach will allow practitioners understanding where to restore according to ecological variables, but also define the feasibility of restoration activities in the medium and long term, including deforested areas and degraded forests.

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Appendix

Image processing

A TM Landsat image (path 233, row 88) from 2011, with a resolution of 30 x 30 m, was acquired. The image was taken during the dry season (February), and was pre-processed, including geometric, atmospheric and topographic corrections. We defined five classes of land cover: (1) forest, (2) exotic-forest plantations, (3) shrubland, (4) agricultural/grassland, and (5) bare ground. For the classification of the scene, we used the "Catastro" developed by the Chilean forest service in 1999 (CONAF et al. 1999), and the updating for the Región de los Rios in 2006 (CONAF-CONAMA 2008). This dataset was developed at 1:50,000 scale and was derived from aerial photographs and satellite imagery. A second dataset corresponded to high resolution ALOS scenes taking in the dry season of 2010.

To classify the scene we used the supervised classification method. Training sites was selected considering representation of all digital categories of radiance according to the spectral signature and colour composites (Chuvieco 2010). Signature separability was assessed by the Bhattacharrya distance, which is used to analyse the quality of training sites and class signatures before performing the classification. The maximum likelihood algorithm was used to assign probabilities of membership to each class. Each pixel was finally assigned to the class of maximum probability. Accuracy assessment of the 2011 (TM) image was conducted using 140 ground control points visited in the field in the dry season of 2012. Most of the processing work was performed using PCI 7.0 (PCI 2001) and ArcGis (ESRI).

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Scelorchilus rubecula (chucao), ave endémica de los bosques templados sudamericanos.

CAPÍTULO 6

Discusión general

Discusión general

Los resultados generados por esta Tesis Doctoral abordan diferentes aspectos de gran relevancia para la restauración de los bosques nativos y contribuyen a la comprensión de los procesos que configuran las comunidades biológicas sujetas a la interacción de diversas alteraciones, la cual ha sido definida como uno de los principales desafíos de la ecología (Mouillot et al. 2012). En concreto, esta Tesis evalúa (1) la influencia de la ganadería y la tala selectiva en la regeneración forestal, tanto a nivel de especie como de comunidad de regeneración (**Capítulos 2 y 3**); (2) la variación del efecto de estas dos perturbaciones según el estado sucesional de los bosques y del régimen de propiedad, incorporando así un factor socioeconómico importante para la conservación de bosques a nivel global, en especial en los países menos industrializados (**Capítulo 2 y 3**); (3) los procesos de cambio en la configuración espacial y temporal del paisaje, determinando la influencia de la expansión de las plantaciones forestales de especies exóticas y de la regeneración natural de los bosques nativos (**Capítulo 4**); y, además, (4) propone un enfoque multicriterio que integra tanto variables ecológicas como socioeconómicas para identificar las áreas de mayor prioridad para la restauración forestal, con el objeto de maximizar los beneficios de las acciones de restauración para la conservación de la biodiversidad y la provisión de servicios ecosistémicos (**Capítulo 5**).

Impactos de la ganadería y de la tala selectiva en los ecosistemas forestales.

A pesar de ser una especie protegida, la conífera *Araucaria araucana* (Molina) K. Koch (Araucariaceae) o araucaria es continuamente expuesta a la influencia directa e indirecta de disturbios de origen antrópico (**Capítulo 2, Figura 6.1**). Nuestros resultados revelan que la regeneración de araucaria está muy afectada por la ganadería, la cual favorece la regeneración de origen asexual, sobre todo en pequeñas propiedades de campesinos. La regeneración asexual es principalmente por rebrote vegetativo, la cual tiene diversas ventajas competitivas respecto a la regeneración de origen sexual. Entre éstas destaca un sistema radicular ya establecido y de mayor superficie para captar agua y nutrientes desde el suelo (Simões & Marques 2007, Miller & Kauffman 1998). Los resultados del **Capítulo 3** demuestran que el impacto de la ganadería no se circunscribe sólo a araucaria, sino que también afecta a la comunidad de regeneración de los bosques siempreverdes, los bosques templados del sur de Chile y Argentina más extensos. En este estudio, los resultados revelan que la ganadería y, en menor grado, la tala selectiva tienen impactos negativos en la regeneración forestal a nivel de especie y de comunidad, y que estos impactos son mayores en pequeñas propiedades y en bosques adultos. Los efectos de las alteraciones en la regeneración forestal podrían tene diversas consecuencias en los ecosistemas, como una menor diversidad fenotípica en características tales como el tipo de fruto, la producción de semillas y el período de floración (Fisher et al. 2009). Estos cambios podrían generar impactos poco conocidos en las propiedades funcionales de los ecosistemas, así como también en su capacidad de respuesta a las alteraciones. En este contexto, existe una urgente necesidad de cuantificar y predecir los efectos de los disturbios en la biodiversidad, con el objeto de guiar los esfuerzos para la conservación y el manejo sustentable de los recursos naturales (Mouillot et al. 2012).

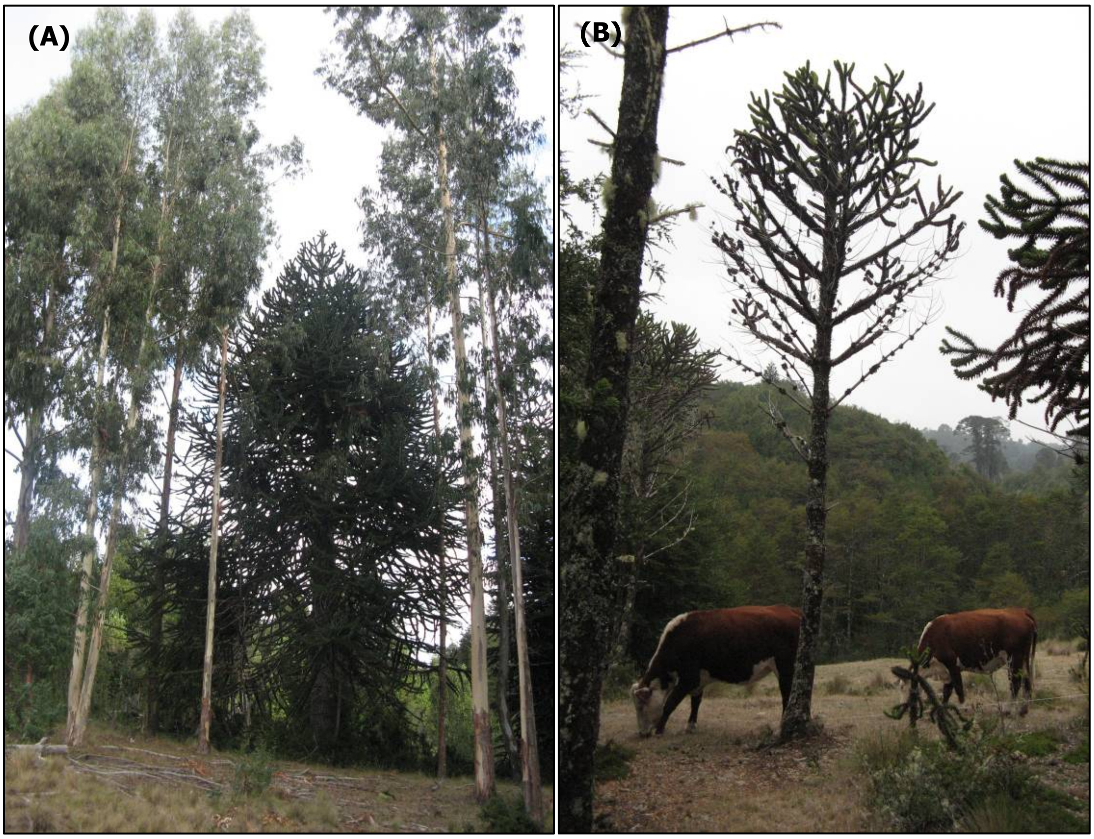

Figura 6.1. De izquierda a derecha: (A) plantación de *Eucalyptus* spp establecida en un remanente de araucaria en áreas colindantes al Parque Nacional Nahuelbuta; (B) ganadería en bosques de araucaria de la Cordillera de Nahuelbuta.

En las grandes propiedades de empresas forestales y en las áreas protegidas, los bosques nativos son utilizados por los campesinos de localidades cercanas como fuente de forraje y refugio para el ganado y la producción de leña, como ha sido reportado en otros estudios (Moorman et al. 2013). En algunos casos, las empresas permiten estas prácticas a través de un "arriendo de talaje", pero ello no incluye control alguno sobre la intensidad ganadera o los bosques expuestos a estas actividades. El menor impacto de la ganadería en la regeneración forestal en grandes propiedades se puede explicar por dos razones. Por una parte, los bosques de estas propiedades son utilizados por el ganado principalmente en invierno, mientras que en las pequeñas propiedades de campesinos la actividad ganadera se desarrolla durante todo el año. De este modo, y aunque no fue analizado directamente en este estudio, la frecuencia de un factor de disturbio puede ser tan determinante como su intensidad en su influencia sobre la regeneración forestal. Por otro lado, la producción de leña y la extracción de madera representan una de las principales fuentes de ingreso para las pequeñas propiedades (Burschel et al. 2003, Zamorano-Elgueta et al. 2008), lo que puede explicar los efectos negativos más severos de la tala selectiva registrados en estas propiedades (**Capítulo 3**). Si bien la tala de araucaria está prohibida por la legislación chilena, la extracción de especies forestales asociadas a esta conífera, en especial *Nothofagus* spp., se realiza sin restricciones. Esta práctica, menos intensa en los bosques nativos de empresas forestales, puede actuar sinérgicamente con la ganadería (Hobbs 2001, Laurance & Useche 2009), aumentando su impacto negativo en la regeneración de araucaria (**Capítulo 2**).

Los resultados de esta Tesis confirman que la ganadería puede dañar, disminuir o limitar el reclutamiento de especies forestales por el pisoteo y ramoneo de los animales domésticos, lo que a su vez puede inducir cambios en la composición de especies (**Capítulo 2 y 3**, Hobbs 2001, Baraloto et al. 2012, Floyd et al. 2003). La tala selectiva, por otro lado, puede causar efectos negativos en la diversidad de la regeneración forestal (Farwig et al. 2008) y de árboles (Polyakov et al. 2007, Ramírez-Marcial et al. 2001), así como en la composición funcional de los ecosistemas forestales (Baraloto et al. 2012). De este modo, los cambios en las condiciones microclimáticas dentro de los bosques podrían causar las respuestas negativas observadas para especies pioneras como *Drimys winteri* J.R. Forst & G. Forst. (**Capítulo 3**), la cual requiere de una humedad permanente del suelo y abundante materia orgánica para regenerarse (Donoso et al. 2007). Como ha sido señalado por otros estudios (Farwig et al. 2008), las condiciones generadas en hábitats alterados pueden igualmente influir en el establecimiento de especies sucesionales tardías como *Saxegothaea conspicua* Lindl., *Podocarpus nubigena* Lindl. o *Laureliopsis philippiana* (Looser) Schoode. En contraste, especies con una gran tolerancia a alteraciones severas, como *Amomyrtus luma* (Molina) D. Legrand & Kausel., presentaron una respuesta positiva a los factores de disturbio evaluados, principalmente por su capacidad de reproducción a partir de rebrotes vegetativos (Donoso & Escobar 2006). Ello puede ser una de las razones que explica que *A. luma* sea una de las especies más abundantes en los bosques templados siempreverdes en la Región estudiada (Veblen et al. 1981). Los bosques inalterados o expuestos únicamente a una tala selectiva podrían ser dominados por especies sucesionales tardías como *S. conspicua*, *Aextoxicon punctatum* Ruiz & Pav. y *L. philippiana*. En cambio, las alteraciones por ganadería o por tala selectiva y ganadería podrían limitar el establecimiento de estas especies y favorecer una composición dominada por *A. luma*, *A. meli* y *Gevuina avellana* Molina (**Figura 6.2**).

Figura 6.2. Regeneración natural y semillas de *G. avellana*.

El papel de las plantaciones de especies forestales exóticas y de la regeneración natural de los bosques en la dinámica del paisaje.

En los **Capítulos 2** y 3 se han analizado los distintos efectos que los disturbios antrópicos tienen en la regeneración de especies leñosas. Un impacto aún más evidente en los ecosistemas forestales, pero no siempre estudiado, es el causado por la deforestación y la fragmentación de los bosques nativos y el papel que en estos procesos tiene la expansión de las plantaciones forestales de especies exóticas. En el **Capítulo 4** analizamos los principales factores de cambio en los bosques templados del sur de Chile en las últimas décadas (1985-2011). Estos corresponden a: (1) una elevada expansión y compactación de las plantaciones de especies forestales exóticas, en especial durante el primer período de análisis (1985-1999); (2) una disminución y fragmentación de la superficie de bosque nativo; y (3) la regeneración natural de los bosques nativos a partir de los matorrales y áreas agrícolas y pastizales. La tasa de incremento en la superficie de las plantaciones de especies forestales exóticas fue la mayor de todas las clases de uso del suelo analizadas, coincidiendo con otros estudios realizados en Chile (Echeverría et al. 2006, Nahuelhual et al. 2012). Por otro lado, se registró una tasa bruta de deforestación del 30% (54,304 ha), y una tasa neta del 5.1% (9,130 ha) de la superficie inicial de los bosques. Esta diferencia entre la tasa bruta y neta de pérdida de los bosques se debe a la conversión de matorrales y áreas agrícolas y pastizales a bosques secundarios. En la Cordillera de la Costa del centro-sur de Chile, Echeverría et al. (2006) registraron una tasa anual de deforestación del 4.5% en el período 1975-2000, mientras que Nahuelhual et al.

(2012) destacaron la rápida expansión de las plantaciones forestales que pasaron a ocupar del 5.5% al 42.4% del territorio, con una tasa anual del 7.9% y 5.1% para el período 1975-1990 y 1990-2007, respectivamente. Históricamente, la tala de bosques y la habilitación de áreas para uso agrícola han sido más intensas en la zona central y centro-sur de Chile, en especial en la Cordillera de la Costa (Camus 2006). Ello se debe principalmente a la mayor accesibilidad de este cordón montañoso en comparación con la Cordillera de los Andes, aumentando con ello su vulnerabilidad frente a las presiones productivas (Armesto et al. 1995).

En la Cordillera de la Costa de la Región de Los Ríos, los cambios en la configuración del paisaje corresponden principalmente a una conversión dinámica entre bosques nativos, plantaciones de especies forestales exóticas, matorrales y áreas agrícolas y pastizales (**Capítulo 4**). Estos patrones son similares a los reportados en los bosques secos de la zona central de Chile (Schulz et al. 2010). Sin embargo, estos últimos se caracterizan por una regeneración natural muy restringida, principalmente por la disponibilidad de agua, la erosión del suelo, los incendios y la limitada capacidad de regeneración de las especies forestales en comparación a las especies de matorrales (Armesto et al. 2007). En nuestra área de estudio, las condiciones de hábitat son menos restrictivas y presentan menores limitaciones para la regeneración natural de los bosques (Albornoz et al. 2013) que las descritas para la zona central. Además, la regeneración natural podría ser favorecida por la creación en el año 2002 de un área protegida privada de 50,000 ha de superficie, establecida en un área históricamente alterada por la producción de leña, la tala intensiva de especies forestales de alto valor como la conífera *Fitzroya cupressoides* (González 2004) y el establecimiento de plantaciones forestales de especies exóticas. Además, las dos mayores empresas forestales del país han sido recientemente certificadas por el *Forest Stewardship Council* (FSC), habiendo iniciado diversas iniciativas de restauración y conservación de bosques nativos para cumplir con los objetivos definidos por esta certificación. Sin embargo, la sustitución de los bosques por plantaciones de especies exóticas es aún una práctica común (Reyes & Nelson 2014), por lo que es importante considerar el efecto que las tendencias actuales pudieran tener sobre el bosque nativo a largo plazo.

Priorización para la restauración forestal

Diversos estudios han propuesto enfoques metodológicos para la identificación de áreas prioritarias para la restauración forestal a escala de paisaje (Ianni & Geneletti 2010, Uribe et al. 2014). Sin embargo, hasta donde sabemos, ninguno de estos métodos considera la restauración de bosques como una medida para mantener la provisión de servicios ecosistémicos. Más aún, en la actualidad se ha dado una menor atención a la evaluación de áreas potenciales de restauración basadas en una definición empírica de bosque degradado. La evaluación de procesos de degradación es más difícil que la medición de la deforestación (Lamb & Gilmour 2003). Mediciones y evaluaciones simples como la cantidad de bosque perdida no consideran los efectos más complejos de la degradación, como son las alteraciones de los ecosistemas por cambios en las condiciones de hábitat. Lamb & Gilmour (2003) propusieron tres categorías generales de ecosistemas degradados:

- Bosques con una degradación muy severa que han perdido la mayor parte de la biodiversidad original, su estructura, biomasa o productividad (**Figura 6.3, punto B**).
- Áreas con una regeneración natural del bosque que se ha establecido tras la acción de intensos disturbios como áreas para uso agrícola abandonadas. Estas áreas han recuperado parte de su biodiversidad, aunque son dominadas por especies de sucesión temprana con baja diversidad de flora y fauna representativa de bosques maduros (**Figura 6.3, punto C**).
- Bosques maduros expuestos a alteraciones severas, como una intensa tala selectiva, que han modificado drásticamente su estructura y reducido su biomasa y productividad. Sin embargo, parte del bosque residual permanece y muchas de las especies forestales del bosque primario están presentes, aunque representadas por plántulas y brinzales (**Figura 6.3, punto D**).

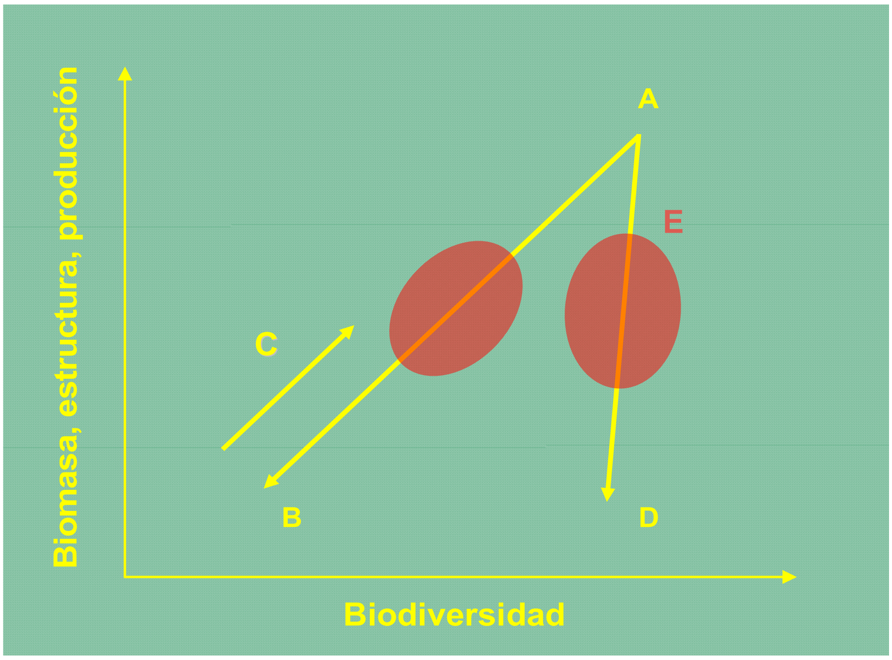

Figura 6.3. Un bosque inalterado **(A)**, puede perder gran parte de su biodiversidad, biomasa, estructura o productividad por intervenciones intensivas **(B)**. El ecosistema puede o no recuperarse **(C)** por ej. por regeneración natural. Algunos métodos de tala intensiva pueden causar la degradación del bosque mientras éste aún presenta una significativa biodiversidad **(D)**. Los procesos tempranos de degradación por factores de alteración de baja intensidad como la ganadería y la tala selectiva afectan la regeneración forestal **(E)**, siendo necesaria su detección para facilitar la restauración de los ecosistemas. Fuente: Modificado de Lamb & Gilmour (2003).

Las categorías de degradación forestal propuestas por Lamb & Gilmour (2003) no consideran como bosques degradados aquellos expuestos a procesos tempranos de alteración **(Figura 6.3, punto E**), al definir esta condición a partir de un estado evidente de pérdida de estructura, biodiversidad y productividad, entre otros (**Figura 6.3, puntos B y D**). Los procesos tempranos de degradación podrían ser detectados, por ejemplo, a través del análisis del efecto de las alteraciones en la regeneración forestal (**Capítulos 2 y 3).** Estos análisis podrían apoyar iniciativas y políticas para revertir estos procesos, aumentando la eficiencia de las acciones de restauración. La detección temprana de procesos de degradación podría facilitar una restauración más rápida y de menor costo en comparación con áreas deforestadas o intensamente degradadas (**Figura 6.3, puntos B y D**), en especial por la disponibilidad de semillas, la protección de árboles remanentes y suelos mejor conservados (Prach et al. 2007, Chazdon 2008).

Por otra parte, la evaluación de la idoneidad ecológica del territorio en términos de biodiversidad y de servicios ecosistémicos y de la factibilidad según criterios socioeconómicos para la identificación de áreas prioritarias permitiría aumentar el éxito en el medio y largo plazo de las acciones de restauración (**Capítulo 5**). La identificación de las áreas más idóneas y factibles para la restauración forestal basada en requerimientos ecológicos y socioeconómicos mínimos de un área ha sido propuesta por otros trabajos (Hirzel et al. 2006, Orsi & Geneletti 2010). Utilizando un enfoque similar al nuestro, Orsi & Geneletti (2010) propusieron un análisis multicriterio que consideró criterios ecológicos como la base del análisis, mientras que los criterios socioeconómicos fueron utilizados para precisar mejor la identificación de áreas prioritarias para la restauración. Sin embargo, estos autores definieron esta priorización a partir del potencial del área de estudio para la conservación de la biodiversidad, la cual además es estimada mediante la riqueza de especies arbóreas según pendiente y altitud. Nuestro enfoque utilizó modelos de distribución de especies para estimar la riqueza potencial de flora, incluyendo especies forestales, trepadoras, rastreras, musgos, helechos, arbustos y epifitas. Esta información fue generada a través de una intensiva campaña de campo realizada en toda el área de estudio, y por tanto aporta una información mucho más precisa que la obtenida por Orsi & Geneletti (2010) de cara a definir áreas potenciales para la restauración forestal.

La demanda de leña, el régimen de propiedad y el establecimiento de plantaciones de especies forestales exóticas son las mayores restricciones que pueden influir en la factibilidad de restauración de un área (**Capítulo 5**). Así, los contextos socioeconómicos pueden limitar o favorecer la implementación de acciones de restauración. Las áreas urbanas demandan una creciente producción de leña en el sur de Chile, la cual representa la principal fuente de energía tanto para uso residencial como industrial (Burschel et al. 2003). Además, una mayor intensidad de disturbios de origen antrópico en los bosques se asocia con el sistema de producción de subsistencia de las pequeñas propiedades campesinas. En estas propiedades la ganadería y la tala selectiva aumentan la degradación de los bosques al influir, por ejemplo, en la regeneración de algunas especies forestales (**Capítulo 2 y 3**). Por otro lado, el régimen de propiedad representa un criterio de gran relevancia para evaluar la factibilidad de la restauración debido a la heterogeneidad de las necesidades y condiciones socioeconómicas asociadas a éste. Una gran parte del área de estudio se caracteriza por la presencia de plantaciones de especies forestales exóticas, las cuales influyen en la provisión de agua, en particular en la temporada de verano (Lara et al. 2009). El manejo de las plantaciones forestales se limita a la tala extensiva de las mismas, exponiendo el suelo a procesos de erosión, a veces irreversibles, durante varios años hasta que la nueva plantación se establece y desarrolla. Es fundamental que este tipo de manejo se modifique hacia intervenciones más sustentables que aseguren el mantenimiento de una adecuada cobertura de árboles para prevenir la erosión del suelo y aumentar la retención y el reciclado de nutrientes (Pérez 1999).

El enfoque multicriterio propuesto en el **Capítulo 5** de esta Tesis Doctoral integra técnicas de agregación para los diferentes criterios utilizados para definir las mejores áreas para la restauración forestal, basadas en la biodiversidad y la provisión de servicios ecosistémicos. Este enfoque podría apoyar el diseño e implementación de políticas e iniciativas de restauración, considerando los diferentes contextos ecológicos y socioeconómicos presentes en un territorio. Por otra parte, proporciona un marco de análisis que podría ser modificado a través de un proceso más participativo en todas las etapas del análisis multicriterio, y a través de la integración de otros objetivos y/o criterios ecológicos, sociales o económicos.

Perspectivas de investigación

La degradación forestal es una problemática de creciente preocupación a nivel global, pero su evaluación y cuantificación es en extremo compleja, por lo que se han planteado numerosas definiciones siguiendo diferentes propósitos e intereses (Lund 2009). El análisis del efecto de la ganadería y de la tala selectiva en la regeneración forestal representa un primer paso en la comprensión del efecto de los factores de disturbio en los ecosistemas forestales. Es necesario desarrollar nuevas investigaciones que permitan avanzar tanto en la detección temprana de estos procesos como en propuestas efectivas para revertir sus impactos. El estudio de la relación de los factores de disturbio con la funcionalidad ecológica de los ecosistemas podría generar nuevos antecedentes para identificar procesos de degradación, por ejemplo analizando el impacto de la ganadería y de la tala selectiva en la provisión de servicios ecosistémicos.

Como lo demuestran los resultados de esta Tesis Doctoral, los factores de disturbio de origen antrópico como la ganadería y la tala selectiva generan impactos negativos en la regeneración forestal, los cuales varían según el régimen de propiedad y estado sucesional del bosque. Sin embargo, es necesario analizar si estos efectos son a su vez influenciados por los procesos de cambio a escala de paisaje como la fragmentación de bosques. Por ejemplo, evaluar si el tamaño y la forma de los fragmentos de bosques se relacionan con el impacto de la ganadería y la tala selectiva en la regeneración forestal y en otros componentes de los ecosistemas, como la diversidad y riqueza de especies de flora.

Nuevos antecedentes permitirían complementar el análisis multicriterio propuesto en esta Tesis mediante (1) la definición de áreas potenciales de restauración a partir de los patrones temporales y espaciales de cambio del uso del suelo. Por ejemplo, el análisis de la dinámica del paisaje podría servir de base para la identificación de zonas potenciales de restauración, al considerar su menor o mayor capacidad de regenerar naturalmente; (2) la identificación de áreas prioritarias en función de las medidas de restauración a implementar, es decir, activa en aquellas áreas con lenta, limitada o sin regeneración natural de la cobertura forestal, o pasiva en aquellas áreas con mayor potencial de regeneración natural; (3) la selección de diferentes criterios ecológicos o socioeconómicos para la identificación de las áreas de mayor idoneidad y factibilidad de restauración; y (4) la asignación de pesos para cada uno de los criterios que reflejen de la mejor manera posible los diferentes intereses y necesidades de los diferentes actores sociales y tomadores de decisiones.

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Amomyrtus meli (meli), una mirtácea endémica de los bosques templados del sur de Chile.

CAPÍTULO 7

Conclusiones generales

Conclusiones generales

A continuación se enuncian las principales conclusiones de esta Tesis Doctoral. La primera y la última son las más generales y transversales.

1. El enfoque de análisis utilizado en esta Tesis permitió revelar el efecto de alteraciones de baja intensidad, en particular la ganadería y la tala selectiva, en la regeneración forestal como una aproximación para la detección de procesos tempranos de degradación de los bosques (**Capítulos 2 y 3**). De igual modo, se determinó el papel de las plantaciones de especies forestales exóticas en la dinámica de cambios en la cobertura del suelo (**Capítulo 4**). Finalmente, a través del método multicriterio propuesto se identificaron las áreas prioritarias para la restauración forestal con el fin de optimizar los beneficios de las acciones de restauración para la conservación de la biodiversidad y la provisión de servicios ecosistémicos en el área estudiada (**Capítulo 5**). El enfoque adoptado representa un primer paso para diseñar e implantar políticas e iniciativas de restauración para revertir o mitigar la degradación forestal a escala de paisaje.

2. La ganadería tuvo un efecto cuantitativo y cualitativo negativo en la regeneración de la araucaria. Así, la actividad ganadera se relacionó con una menor densidad de plántulas y brinzales y con un aumento en la proporción de regeneración de origen sexual, lo que podría afectar la conservación de la diversidad genética de la especie en el largo plazo. Estos efectos fueron mayores en las propiedades de los pequeños propietarios forestales (**Capítulo 2**).

3. Los impactos que la ganadería genera en una especie protegida como la araucaria demuestran que la protección de una sola especie no necesariamente asegura su conservación en el largo plazo. Es necesaria la protección del ecosistema en el cual ésta se desarrolla e involucrar a los actores locales en el diseño de estrategias para reducir el impacto de la ganadería en los relictos de araucaria (**Capítulo 2**).

4. En los bosques siempreverdes estudiados, la ganadería y la tala selectiva influyeron en la regeneración forestal tanto a nivel de especie como de comunidad, aunque la ganadería tuvo un efecto negativo mayor que la tala selectiva. Mientras los efectos de la tala selectiva fueron variables según el régimen de propiedad, la ganadería siempre presentó efectos negativos en la regeneración, en especial en los bosques adultos y en pequeñas propiedades de campesinos (< 200 ha). Las estrategias de restauración y conservación deberían primero concentrarse en limitar o eliminar los principales factores de disturbio y luego en proteger y

recuperar las especies más sensibles a estas alteraciones (**Capítulo 3**).

5. Los bosques siempreverdes sin alteraciones o sólo expuestos a tala selectiva podrían ser dominados por especies de sucesión tardía como *Saxegothaea conspicua*, *Aextoxicon punctatum* y *Laureliopsis philippiana*. En cambio, la continua presión por la ganadería y/o por la tala selectiva podría impedir el establecimiento de especies tolerantes y semitolerantes a la sombra, y favorecer una composición dominada por *Amomyrtus luma*, *Amomyrtus meli* y *Gevuina avellana*. Estos resultados confirman que los disturbios de origen antrópico evaluados, en especial la ganadería, pueden disminuir, limitar o impedir el reclutamiento de especies leñosas, con impactos desconocidos en las propiedades funcionales de los ecosistemas forestales (**Capítulo 3**).

6. La deforestación y la fragmentación de los bosques en el área de estudio fueron mayores en el período 1985-1999, si bien su intensidad ha disminuido recientemente (período 1999-2011) y, además, son menos intensas que en otras regiones de Chile. Sin embargo, ambos procesos continúan alterando los ecosistemas forestales en el área de estudio, en especial por la expansión de las plantaciones forestales de especies exóticas (**Capítulo 4**).

7. Se estimó una pérdida bruta de bosques (30%) muy superior a la neta (5.1%), básicamente como consecuencia de una alta regeneración natural a partir de los matorrales y las áreas agrícolas y pastizales abandonados. Así, mientras la deforestación afecta tanto a los bosques adultos como a los secundarios, los bosques regenerados corresponden exclusivamente a estos últimos. Ello puede influir en la capacidad de los bosques para proveer servicios ecosistémicos, incluyendo aquellos relacionados con la conservación de los suelos y la producción y regulación del agua, con impactos que van más allá del uso inmediato del suelo (**Capítulo 4**).

8. Las áreas prioritarias de restauración forestal se distribuyeron por toda el área de estudio, si bien se concentraron en las regiones Central y Este del territorio estudiado, en donde presentaron una distribución continua. En el Norte y Sur del territorio las áreas prioritarias fueron representadas por pequeños parches aislados. En el área Norte ello se debe a la mayor presión productiva y la menor factibilidad económica, mientras que en el Sur se explica por la existencia de bosques mejor conservados (**Capítulo 5**).

9. Las áreas prioritarias para la restauración forestal tienen una alta biodiversidad y una severa erosión potenciales, y están localizadas en cuencas hidrológicas caracterizadas por bajos coeficientes de escorrentía y áreas de elevada accesibilidad pero al mismo tiempo expuestas a una menor presión productiva del bosque. Su superficie total se estimó en un

10.7% del área de estudio, de la cual el 7.4% correspondieron a áreas deforestadas y el 3.3% a bosques degradados en zonas cercanas a bosques bien conservados, en propiedades de empresas forestales certificadas por el *Forest Stewardship Council*, en pequeñas propiedades sustentables y en áreas protegidas (**Capítulo 5**).

10. La evaluación de la idoneidad ecológica y la factibilidad socioeconómica para la conservación de la biodiversidad y la provisión de servicios ecosistémicos representa un enfoque innovador propuesto por esta Tesis Doctoral, que permitió identificar las mejores áreas para la restauración forestal en el área de estudio (**Capítulo 5**). Ello proporcionará a los expertos y planificadores nuevas herramientas y antecedentes para identificar aquellas áreas ecológicamente más idóneas para la restauración forestal, pero también para definir la factibilidad de estrategias de restauración en el medio y largo plazo, incluyendo tanto los bosques degradados (**Capítulos 2 y 3**) como las áreas deforestadas (**Capítulo 4**).

Bosque de araucaria, Parque Nacional Nahuelbuta.

APÉNDICE

CURRICULUM VITAE

Curriculum vitae

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EDUCACIÓN

2014 Ph.D. en Ecología (candidato). Universidad de Alcalá de Henares, Madrid, España. Tesis: Definición de áreas prioritarias para la restauración forestal en la Cordillera de la Costa de la Región de Los Ríos, Chile.

2008 Maestría en Ciencias en recursos naturales y desarrollo rural. El Colegio de la Frontera Sur, San Cristóbal de las Casas, Chiapas, México. Tesis: Factores de disturbio, actores sociales y estados de conservación en bosques de *Araucaria araucana* (Molina) K. Koch en la Cordillera de Nahuelbuta, Chile.

2002 Licenciatura en Ciencias Forestales, especialización en manejo de bosques naturales. Universidad Austral de Chile, Valdivia, Chile. Tesis: Caracterización del sistema productivo rural y análisis de sustentabilidad.

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6. 2003 - 2005. An integrated conservation programme for threatened endemic forest species in Chile. Universidad Austral de Chile. **Colaborador.**

7. 2003 - 2004. Conservation and restoration of Monkey Puzzle (*Araucaria araucana*) forests in Chile. Universidad Austral de Chile. **Colaborador.**

8. 2003 - 2005. Biodiversity conservation, restoration and sustainable use of fragmented forests landscapes (BIOCORES). Universidad Austral de Chile. **Colaborador.**

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2. 2009. Beca Chile, granted by Comisión Nacional de Investigación Científica y Tecnológica – CONICYT, Ministerio de Educación, Gobierno de Chile. Estudios de doctorado.

3. 2007. Red Latinoamericana de Botánica. Financiamiento para tesis de Maestría.

4. 2007. WWF Prince Bernhard Scholarship for Nature Conservation. Financiamiento para tesis de Maestría.

5. 2005. Catherine Olver Scholarship, otorgada por el Royal Botanic Garden Edinburgh. Estudios de Maestría.

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(www.revista.bosquenativo.cl)

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(http://www.revistaecosistemas.net/index.php/ecosistemas)

MEMBRESÍAS

Miembro de la Asociación Española de Ecología Terrestre, desde 2010.
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Miembro fundador de la Fundación Internacional para la Restauración de Ecosistemas. (www.fire.org)

Miembro fundador de la Sociedad Iberoamericana y del Caribe de Restauración Ecológica.

PASATIEMPOS E INTERESES

Artes marciales (Qwan Ki Do)

Trekking

Fotografía

Terapia de parrilla