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“EVALUACIÓN DEL PAISAJE PARA LA CONSERVACIÓN DE LA BIODIVERSIDAD Y SERVICIOS ECOSISTÉMICOS EN EL CENTRO-SUR DE CHILE”

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A mi familia

Teresa, Reinaldo, Martha, Harold e Isabella

"Nunca olvides quien eres, ni de dónde vienes" P.D.

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RESUMEN

El planeta está experimentando importantes cambios ambientales globales, los cuales han sido principalmente generados por actividades antrópicas. Estos cambios han conducido al planeta y los paisajes que lo constituyen a una trayectoria ambientalmente insostenible, lo que ha generado la urgente necesidad de obtener conocimiento clave que permita revertir esta situación.

El cambio de uso del suelo (CUS), proceso de origen antrópico, ha sido identificado como el principal conductor de cambio global, debido a que este puede conducir a la fragmentación y pérdida de hábitat forestal. Lo anterior se evidencia en la alteración de los patrones espaciales del paisaje (composición y configuración), tales como conectividad, tamaño y densidad de parches de hábitat. Estos cambios, a su vez, pueden impactar a las especies más vulnerables y la biodiversidad, la cual está intrínsecamente relacionada con los servicios ecosistémicos (SE). Actualmente, no hay suficientes estudios que permitan entender cómo el CUS impacta en los patrones espaciales de: i) hábitats de especies amenazadas, y ii) de la biodiversidad, la cual influencia la provisión de SE. Tales estudios proveerían importante conocimiento para la comprensión y desarrollo de la sustentabilidad.

El paisaje forestal del sur de Chile, el cual ha sido identificado como un hotspot para la conservación de la biodiversidad en el mundo, ha registrado una progresiva antropización en las últimas décadas debido al intenso y constante CUS. Lo anterior ha conducido a importantes cambios en la biodiversidad a nivel de especie, comunidad y hábitat, y en la provisión de SE, lo que ha destacado la necesidad de implementar acciones que contribuyan en la sustentabilidad de este paisaje. En este contexto, este estudio evaluó el impacto del CUS en los patrones espaciales del paisaje forestal y hábitat de *F. cupressoides*, especie categorizada en Peligro, entre 1999 y 2011. Además, se relacionaron los cambios en los patrones espaciales del hábitat con la composición actual de las poblaciones de *F. cupressoides* y comunidades asociadas. Este estudio también evaluó el impacto del CUS en los patrones espaciales de la diversidad de hábitats de bosque nativo (DHBN), usada como proxy de biodiversidad, y a su vez la influencia de estos cambios en la provisión de los SE control de erosión, acumulación de suelo y provisión de agua entre 1986 y 2011. Finalmente, se evaluó la congruencia especial entre la DHBN y SE durante el mismo periodo de estudio.

Las cuatro poblaciones de *F. cupressoides* estudiadas están localizadas en la Depresión Intermedia, región de Los Lagos, Chile. Mediante imágenes satelitales fue evaluado, a nivel de paisaje, el impacto del CUS en los patrones espaciales del hábitat de *F. cupressoides.* Se establecieron dos parcelas en cada población y se registró la densidad de *F. cupressoides* y riqueza de especies de las comunidades asociadas, lo cual fue analizado junto con los patrones espaciales de los hábitats. La evaluación de la DHBN y SE fue realizada en la cuenca del río Cruces, en la región de Los Ríos, Chile. Mediante imágenes satelitales fue evaluado el impacto del CUS en los patrones espaciales de la DHBN. El mapeo y cuantificación de los SE y su relación con la DHBN fue realizada a través de modelos espacialmente explícitos. La congruencia espacial entre DHBN y SE fue evaluada mediante análisis de superposición.

Los resultados evidencian que la pérdida de hábitat potencial para *F. cupressoides* en el paisaje fue de 46%. La pérdida de hábitat para las cuatro poblaciones varió entre 38% y 100%. La densidad de *F. cupressoides* fue menor a medida que hubo menos hábitat. El número de especies asociadas fue más alto a medida que el tamaño del parche de hábitat fue menor y la matrix fue más antrópica. En 2011 los diferentes tamaños de los hábitats estuvieron relacionados con las diferentes densidades de las poblaciones. En este estudio multiescala, el CUS estuvo asociado con la pérdida de hábitat de *F. cupressoides*. Por otro lado, fue registrada una pérdida del 12% del área de la HDNF, un incremento del 150% en el número de parches con presencia de DHBN, y una pérdida de 0.20 en el índice de diversidad de Shannon. La más grande disminución en la provisión de los SE fue registrada para control de erosión (346%), y la menor para provisión de agua (11%). La pérdida de provisión de SE fue explicada por la interacción de los cambios en los patrones espaciales de la DHNB ($p < 0.001$). 68% de la DHBN registró alta congruencia especial con: 77% de provisión de agua, 69% y 67% con control de erosión y acumulación de suelo, respectivamente. La disminución en la provisión de los SE está relacionada con la pérdida de DHBN, la cual fue causada por el CUS. Se evidencia que la conservación de la DHBN puede asegurar una importante mantención de los SE. Este estudio constituye el más profundo análisis de la relación entre CUS, biodiversidad y SE que se ha realizado en Chile; y provee conocimiento fundamental para el desarrollo de la planificación de la conservación y toma de decisiones.

Finalmente, este estudio contribuye en la comprensión y práctica de la sustentabilidad de los paisajes estudiados, la cual puede asegurar el bienestar humano.

ABSTRACT

The planet is experiencing a significant global environmental impact mainly driven by anthropogenic activities. These changes have led the planet and its constituent landscapes into an unsustainable trajectory, which has highlighted the urgent need for key knowledge to reverse this situation.

Land-use change (LUC), an anthropogenic process, has been identified as the main driver of global change, due to this may lead to habitat fragmentation and loss in forest landscapes. This is evidenced by alterations of landscape spatial patterns (composition and spatial configuration), such as habitat connectivity, habitat patch density and habitat sizes. These changes, in turn, can impact on vulnerable species and biodiversity, which is intrinsically related to ecosystem services (ES). Currently, there is limited studies permitting an understanding how LUC can alter the spatial pattern of: i) unique habitats in which threatened species can become extinct, and ii) biodiversity, which influence the provision of the ES. Such studies would provide crucial knowledge for the understanding and practice of sustainability.

The forest landscape of southern Chile, which has been identified as a hotspot for biodiversity conservation in the world, has undergone a progressive anthropization in recent decades due to intense and permanent LUC. The foregoing have led to important changes in biodiversity at the species level, habitat and the provision of the ES, which have highlighted the need to implement effective conservation strategies. In this context, this study assessed the impact of LUC on the spatial patterns of the forest landscape and habitat of *F. cupressoides*, species which has been categorised as Endangered, between 1999 and 2011. Additionally, this study related these changes in spatial patterns to the current composition of the populations of *F. cupressoides* and associated communities. This study also assessed the impact of LUC on the spatial patterns of the habitat diversity of the native forest (HDNF), which was used as a proxy of biodiversity, and in turn, the influence of these changes on the provision of the ES water supply, control of erosion, and soil accumulation between 1986 y 2011. Finally, this study assessed the spatial congruence between HDNF and ES in the same study period.

The populations studied of *F. cupressoides* are located in the Central Depression, Los Lagos Region, Chile. By means of satellite images, the impact of LUC on the habitat spatial

patterns of *F. cupressoides* was assessed at the landscape level. Eight plots were established in four remaining populations of the species to assess the current status of these. In each plot, the *F. cupressoides* density and species richness of the associated communities were recorded and analysed together with the spatial patterns at the population and community level. On the other hand, the assessment of the HDNF and ES was carried out in Río Cruces watershed, Los Ríos region, Chile. By means of satellite images was assessed the impact of LUC on the habitat spatial patterns of the HDNF. The mapping and quantification of the ES and its relationship with HDNF was carried out by spatially explicit models. The spatial congruence between HDNF and ecosystem services was assessed using overlap analysis.

The results of this study evidence that the loss of *F. cupressoides* potential habitat in the landscape was 46%. The loss of habitat for the four populations ranged from 38% to 100%. The density of *F. cupressoides* was lowest where there was a lower habitat. The number of species was higher as the size of the habitat patches was lower and the matrix was more dominated by human-related land uses. In 2011, the different sizes of habitat were related with the different densities of the populations. In this multiscale study, LUC was associated with a loss of *F. cupressoides* habitat. On the other hand, 12% of HDNF area, more than 150% increase in the number of patches with presence of HDNF, and loss of 0.20 in the Shannon diversity index were recorded. The greatest decrease in the provision of ES was recorded for erosion control (346%), and the lowest for water supply (11%). The loss of provision of the ES was explained by the interaction of changes in the spatial patterns HDNF ($p < 0.001$). 68% of biodiversity registered high spatial congruence with: 77% of water supply, 69% and 67% of erosion control and soil accumulation, respectively. The decrease in provision of ES is related to the loss of HDNF, which was caused by the intensification of land use. Accordingly, the conservation of HDNF may ensure an important maintenance of the ES. This study constitutes the largest analysis of the relationship between impacts of LUC on HDNF and ES that has been done in Chile. It provides fundamental information to the development of alternatives for conservation planning and decision-making.

Finally, this study contributes to the understanding and practice of sustainability landscapes studied, which can ensure human well-being.

Capítulo I
Introducción General

Sustentabilidad del paisaje

El planeta está experimentando un ritmo sin precedentes de cambios ambientales globales, los cuales han sido generados principalmente por actividades antrópicas (Steffen et al. 2011; Vince 2011). Desde el advenimiento de la revolución industrial hace aproximadamente dos siglos, los avances tecnológicos se han multiplicado y la población mundial ha aumentado de manera exponencial, lo que ha traído consigo un incremento de las demandas socioeconómicas y la intensificación de sus actividades (Wu 2013). Este desarrollo ha generado importantes cambios ambientales globales en el clima (Vitousek et al. 1997; IPCC 2007), el uso del suelo (Tilman et al. 2001) e interacciones biológicas (Walther et al. 2009).

Los diferentes cambios ambientales globales han generado una reducción en la biodiversidad a nivel mundial (Sala et al. 2000a), la cual en su más amplio sentido es definida como la riqueza y abundancia de genes, especies y ecosistemas (Balmford and Bond 2005); y en los servicios ecosistémicos (SE) (Walker et al. 2006), los cuales son los beneficios que los ecosistemas proveen a los seres humanos (MA 2005). En consecuencia el bienestar humano global ha experimentado una creciente amenaza en los últimos tiempos (Díaz et al. 2006; Wu 2013).

En este contexto, la trayectoria ambientalmente insostenible en la que se encuentra el planeta (NRC 1999; Kates et al. 2001; Clark and Dickson 2003; Bettencourt and Kaur 2011; Kates 2011) evidencia la necesidad de urgentes acciones que permitan revertirla (Wu 2006; Wu 2013). En la actualidad, un desarrollo ambientalmente sustentable es una necesidad, más no una elección (Wu 2006; Forman 2008). Es decir, es necesario un desarrollo en el que las actividades económicas sean parte del ámbito social, y que tanto las acciones económicas y sociales estén limitadas por el medio ambiente (Musacchio 2009; Wu 2013), de modo que se alcance un balance entre las necesidades humanas y la integridad ambiental (Wu 2006; Musacchio 2009).

De acuerdo a lo anterior, es ampliamente reconocido que la sustentabilidad es el tema de nuestros tiempos y representa uno de los mayores desafíos (Forman 1990; Wu 2006; Forman 2008; Wu 2012). Para afrontar este reto, que requiere una mejor comprensión de la dinámica relación entre la sociedad y la naturaleza a escala local, de paisaje y global, en las últimas décadas se ha establecido la ciencia de la sustentabilidad (NRC 1999; Kates et al. 2001; Forman 2008; Wu 2012).

En este contexto, surgió la sustentabilidad del paisaje, la cual es definida como la capacidad del paisaje para proveer, a largo plazo, SE esenciales que permitan mantener y mejorar el bienestar humano a pesar de los cambios ambientales y socioculturales (Wu 2012; Wu 2013). La comprensión y desarrollo de la sustentabilidad a escala de paisaje permite alcanzar un balance entre las actividades humanas y la integridad ambiental (Wu 2013), debido a que el paisaje representa la escala más pequeña y operativa en la cual las interacciones entre sociedad, biodiversidad y SE pueden ser estudiadas y entendidas (Wu 2006; Wu 2012). De tal modo que es necesario desarrollar estudios que contribuyan en la sustentabilidad del paisaje, con los cuales se pueda revertir la problemática ambiental actual.

Actualmente, está documentada que la pérdida de biodiversidad (e.g. riqueza y abundancia de especies de plantas) y SE (e.g. red alimentaria, provisión de agua, medicinas, entre otras) son generadas por los diferentes conductores de cambio global (NRC 1999; Clark and Dickson 2003; Baillie et al. 2004; MA 2005; Díaz et al. 2006; IUCN 2013). Sin embargo, existen pocos estudios que evalúen y analicen la manera en que los conductores de cambio global impactan la biodiversidad y la influencia de estos impactos en la provisión de los SE a escala de paisaje. Este tipo de estudios brindarían información fundamental sobre cómo, cuándo y dónde los conductores de cambio global impactan el bienestar humano, lo que aportaría información fundamental para comprender y llevar a cabo acciones que contribuyan en el desarrollo de la sustentabilidad (Forman 2008; Musacchio 2009; Wu 2013). Así, se hace necesario contribuir en la sustentabilidad del paisaje desde la evaluación y comprensión de la relación entre el cambio de uso del suelo (CUS), biodiversidad y SE en paisajes antropizados cambiantes.

Antes de considerar las evidencias teóricas y prácticas del CUS, biodiversidad y SE es necesario examinar el paisaje como unidad espacialmente heterogénea y fundamental para la comprensión de la sustentabilidad.

El paisaje: unidad espacialmente heterogénea

El paisaje es definido como un área espacialmente heterogénea, la cual está compuesta por múltiples elementos o parches que corresponden a diferentes hábitats, tipos de vegetación o usos de suelo (Forman and Godron 1986). El paisaje se puede caracterizar según sus patrones espaciales, procesos y cambios (Turner 1989). Los patrones espaciales del paisaje corresponden a la composición y configuración, los cuales se refieren a los tipos de elementos o parches y al arreglo espacial de estos en el paisaje, respectivamente. Por otro lado, los procesos del paisaje tienen relación con el flujo de energía, materia y organismos, y disturbios (Turner 1989; Wiens 2002). Los procesos del paisaje son espacialmente dependientes de los patrones espaciales (e.g. el movimiento de nutrientes y sólidos en suspensión depende de las características de la red hidrológica en los cuerpos de agua), relación que a su vez determina diversos procesos ecológicos tales como ciclo de nutrientes y dinámica de poblaciones. Además, existe una retroalimentación entre estos procesos, los cuales afectan los patrones espaciales del paisaje (Fig. 1) (Forman and Godron 1986). El cambio del paisaje se refiere a que los patrones espaciales y procesos del paisaje cambian en el tiempo y espacio, debido principalmente a acciones antrópicas como el uso del suelo (Fig. 1) (Rindfuss et al. 2004). En este contexto, el paisaje es la unidad espacialmente heterogénea a través de la cual se puede evaluar y analizar la influencia de los cambios de los patrones espaciales en los diversos procesos. Este tipo de estudios brindaría información fundamental para: i) comprender la manera como las acciones antrópicas impactan la biodiversidad y SE, y ii) identificar áreas de importancia para la conservación de estos recursos. El desarrollo de este tipo de estudios se sustenta en el uso de diferentes índices o métricas de paisaje, las cuales se describen a continuación.

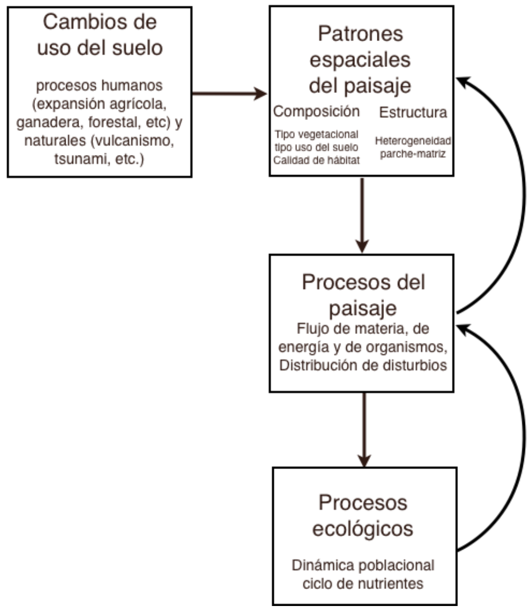

Figura 1.1 Relación entre los patrones espaciales, procesos del paisaje y procesos ecológicos y su interacción con cambios de uso del suelo antrópico. Fuente: Echeverría et al. 2014.

Índices del paisaje

Los índices o métricas de paisaje son ampliamente usados para analizar en profundidad los patrones espaciales de los diferentes tipos de hábitats, formaciones vegetacionales y usos del suelo, los cuales pueden afectar los procesos ecológicos y de paisaje (Franklin 2001). El desarrollo de estos índices de paisaje se ha sustentado en la aplicación de los Sistemas de Información Geográficos (SIG) y el uso de mapas temáticos derivados de imágenes satelitales (Newton et al. 2009).

Los patrones espaciales del paisaje se pueden cuantificar a través de índices que exploran: i) la configuración de los elementos o parches del paisaje, tales como los índices de área y densidad de parches; y ii) la composición de los múltiples elementos del paisaje, tales como los índices de riqueza de parches y diversidad de Shannon (Mcgarigal et al. 2002). Los índices evidencian de forma explícita los patrones del paisaje, los procesos que pueden ser responsables de esos patrones y como estos se relacionan con los procesos que se estudia (Mcgarigal et al. 2002). Diversos estudios han usado los índices en la evaluación de los cambios de paisaje producidos principalmente por el uso de suelo antrópico, lo que ha permitido conocer la manera cómo han cambiado los paisajes, los impactos producidos en la biodiversidad y SE, y las necesidades de manejo y conservación (Zeng and Wu 2005; Sano et al. 2009; Peng et al. 2010).

Cambio de uso del suelo

La necesidad de proporcionar alimento, fibra, agua y abrigo a más de seis mil millones de personas ha generado un intenso y constante uso de suelo antrópico a nivel global (Vitousek et al. 1997). Las áreas agrícolas y ganaderas, las plantaciones forestales, y las áreas urbanas se han expandido en las últimas décadas acompañadas de un gran aumento en el consumo de energía, agua y agroquímicos (Foley et al. 2005). Tales cambios han permitido a los seres humanos apropiarse de una parte importante de los recursos del planeta (Vitousek et al. 1997), afectando la capacidad para sostener la producción de alimentos, mantener la calidad y cantidad de agua, regular las condiciones climáticas y calidad del aire, y controlar las enfermedades infecciosas (Foley et al. 2005).

Las severas alteraciones ambientales causadas por el CUS en el último siglo han sido tan drásticas como las que ocurrieron durante los períodos glaciales (NCR 2001). De tal manera que, debido a la velocidad, magnitud y alcance con la que ocurre el CUS, éste ha sido identificado como el principal conductor de cambio global (NCR 2001). Según Lambin et al. (2001), los principales causantes de este cambio global no sólo han sido el crecimiento poblacional y la pobreza, como suele argumentarse. La respuesta a oportunidades económicas mediatizadas por factores institucionales, y fuerzas globales, amplificadas o atenuadas por factores locales, se han convertido en los principales "factores forzantes" (Lambin et al. 2001).

Entre los principales impactos del CUS se encuentran la pérdida y fragmentación de hábitat de paisajes forestales (Pimm and Raven 2000; Sala et al. 2000b). Se estima que durante la primera mitad del siglo XX la cobertura forestal en el planeta declinó en 13 millones de hectáreas por año, esto debido principalmente a la actividad humana (FAO 2010). Sin embargo, Hansen et al. (2010) reportó una pérdida mayor de cobertura forestal entre el 2000 y 2005, la cual fue de 20 millones de hectáreas por año. La Organización de las Naciones Unidas para la Alimentación y la Agricultura (FAO 2010) también reportó un aumento en la perturbación de la cobertura forestal en el planeta durante la década del 2000 – 2010, la cual fue de 4.2 millones de hectáreas por año. Lo anterior, debido a la tala selectiva y otras formas de perturbación antrópica (FAO 2010). En este contexto, la pérdida y fragmentación de hábitat forestal puede generar diversos impactos a nivel de paisaje y especie, los cuales causan serias consecuencias ecológicas que influyen en el decline de la biodiversidad (Sala et al. 2000a; Baillie et al. 2004).

Antes de considerar las evidencias teóricas y prácticas de los impactos de la fragmentación a nivel de paisaje y especie, y las consecuencias ecológicas asociadas, es necesario describir los procesos de deforestación y fragmentación.

Pérdida y fragmentación de hábitat de paisajes forestales

La deforestación ha sido definida como la pérdida de bosque natural (Allen and Barnes 1985). Esta está asociada con importantes cambios en el clima, balance hidrológico, almacenamiento de carbono, entre otros (Laurance 1999). Por otro lado, la fragmentación de bosques ha sido definida como la división de grandes y continuos parches de bosque en otros más pequeños (Forman and Godron 1986). Esta ocurre cuando un parche de bosque se subdivide debido a un disturbio natural o por actividades antrópicas, tales como el desarrollo urbano e industrial, expansión agrícola, explotación maderera, entre otras (Lindenmayer and Fischer 2006).

Impactos de la fragmentación de hábitat a nivel de paisaje y especie

El proceso de fragmentación de paisajes forestales se reconoce a través de cuatro componentes principales. El primero es la pérdida completa de hábitat forestal debido a que una porción del paisaje es transformada a otro tipo de uso del suelo. La pérdida de hábitat comienza con una perturbación, la cual genera claros que producen la división de continuos y homogéneos hábitats forestales (Lindenmayer and Fischer 2006). A medida que avanzan las perturbaciones, se incrementa el número y tamaño de los claros, lo que disminuye el área total de hábitat y aumenta el aislamiento de los hábitats remanentes (Harper et al. 2005; Newton and Echeverría 2014). En las fases iniciales del proceso de fragmentación, la pérdida de hábitat es la principal causa del decline de la biodiversidad, mientras que en fases avanzadas lo es el aislamiento de los individuos (Newton and Echeverría 2014). El segundo componente es la reducción del tamaño de parches de hábitat. A medida que se genera la pérdida completa de hábitat, se crean parches de hábitat remanentes que poseen un menor tamaño, lo que hace más vulnerable a las especies de los bordes a las condiciones ambientales adversas, lo que aumenta la probabilidad de extinción (Lindenmayer and Fischer 2006). A su vez, los efectos y tensiones de la matriz inciden en el interior de los hábitats, de tal modo que las especies de interior pueden llegar a ser impactadas (Manu et al. 2007). Sin embargo, es probable que el interior de los hábitats mantenga sus condiciones biofísicas y los efectos de la matriz solo impacten los bordes de estos (Fletcher 2005). El tercer componente es el aumento del aislamiento de parches de hábitats a medida que nuevos usos del suelo ocupan el área intervenida. Diversos procesos ecológicos que influyen en la biodiversidad y que dependen de un vector para su transmisión y (e.g. dispersión de semillas, polinización, relación depredador-presa, entre otros) son los más impactados por el aislamiento de hábitats (Newton and Echeverría 2014). El cuarto y último componente es el aumento del efecto borde. El proceso de fragmentación genera parches de hábitat que poseen diversos bordes o límites, los cuales implican cambios microclimáticos de luminosidad, temperatura, viento, humedad e incidencia de incendios (Lindenmayer and Fischer 2006). Estos cambios pueden tener una importante influencia en procesos ecológicos que están relacionados con la biodiversidad, tales como la dispersión, establecimiento, crecimiento y sobrevivencia de semillas (Harper et al. 2005).

También se han identificado las tres principales consecuencias ecológicas de la fragmentación forestal que influyen en la pérdida de biodiversidad. La primera es la pérdida de especies en parches de hábitat debido a la pérdida y reducción de estos (Bennett 2003). La segunda consecuencia son los cambios en la composición de ensambles de especies debido a diferentes respuestas a la fragmentación. Las especies más sensibles a la fragmentación son aquellas cuya presencia es en bajas densidades o tienen alguna dependencia a hábitats de interior (Laurance et al. 2010). Estas especies pueden incluir animales de gran tamaño que requieren grandes áreas, depredadores que están al final de la cadena alimenticia y especies especialistas en alimentación o hábitat. La tercer consecuencia son los cambios en los procesos ecológicos que forman parte del funcionamiento de los ecosistemas tales como interacciones por competencia, dispersiones de semillas, polinización entre otros (Lindenmayer and Fischer 2006). Estos cambios, debido a la pérdida de especies que tienen un papel clave en estos procesos, pueden tener efectos negativos en la persistencia de un importante número de especies, generar el decline poblacional y aumentar la probabilidad de extinción (Lindenmayer and Fischer 2006; Laurance et al. 2010).

Debido a que la fragmentación de bosques puede generar serias consecuencias en los procesos ecológicos los cuales implican alteraciones negativas en la biodiversidad a diferentes niveles de organización ecológica (Baillie et al. 2004; Mace et al. 2005; MA 2005; Díaz et al. 2006), se hace necesario desarrollar investigaciones que evalúen y analicen cómo ocurre esta relación. Este tipo de investigaciones brindaría información valiosa que permitiría desarrollar acciones de manejo que garanticen la mantención de los procesos ecológicos y por ende la de la biodiversidad.

Diversos estudios han evidenciado que las inesperadas e irreversibles alteraciones negativas en la biodiversidad, causadas por la fragmentación y deforestación del hábitat forestal, afectan la provisión de los SE (Baillie et al. 2004; MA 2005; Díaz et al. 2006). A continuación se discuten las evidencias teóricas y prácticas de la relación entre la biodiversidad y servicios ecosistémicos.

Biodiversidad y Servicios Ecosistémicos

La biodiversidad composicional, estructural y funcional (Fig. 2) regulan la magnitud y variabilidad de los procesos ecosistémicos (e.g. descomposición, ciclo de nutrientes, evapotranspiración, entre otros) (Wallace 2007), los cuales son los encargados de mantener la integridad de los ecosistemas (Díaz et al. 2006). Estos, a su vez, proveen servicios (SE) a los seres humanos, los cuales proporcionan salud, seguridad, materiales básicos para la vida, entre otros (MA 2005). De tal modo que los SE se obtienen sólo si los ecosistemas mantienen la biodiversidad que garantiza los procesos funcionales necesarios para proporcionarlos (MA 2005). En este sentido, la biodiversidad constituye la base para la provisión de los SE, los cuales permiten el desarrollo social, cultural y económico de la humanidad (MA 2005).

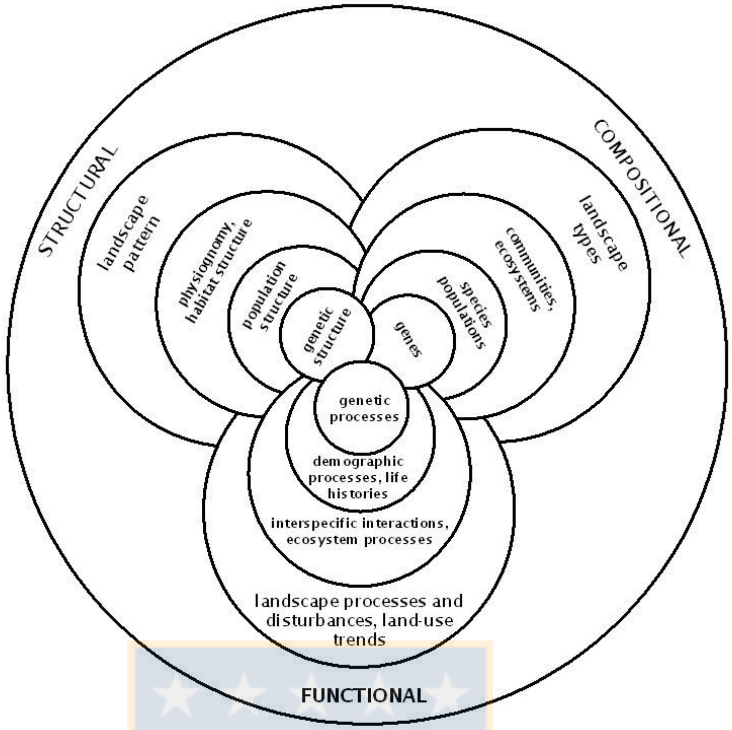

Figura 1.2. Biodiversidad composicional, estructural y funcional representadas como esferas interconectadas, incluyendo cada una diversos niveles de organización. Fuente: Rozzi et al. 1994.

Irónicamente, la principal amenaza para la biodiversidad y SE es la acción humana a través de diferentes conductores de cambio global, principalmente el CUS (Swift et al. 2004; MA 2005). La pérdida de biodiversidad por lo general implica, inesperadas e irreversibles alteraciones negativas de los procesos ecosistémicos, con repercusiones a nivel de los SE, con grandes pérdidas ambientales, económicas y culturales (MA 2005). En las últimas décadas, los ejemplos más dramáticos de los efectos de cambios en la biodiversidad sobre los ecosistemas se han producido a nivel de paisaje y han involucrado diversas alteraciones en la red alimentaria a través de interacciones indirectas y cascadas tróficas (Díaz et al. 2006).

Debido a lo anterior, en 1992 fue promulgado por parte de la Organización de la Naciones Unidas (ONU) el Convenio sobre la Diversidad Biológica, el cual plantea entre sus objetivos lograr una reducción significativa del ritmo de pérdida de biodiversidad. Posteriormente, Myers et al. (2000) identificaron 35 "ecorregiones críticas" o hotspot de biodiversidad en el planeta, los cuales son áreas que presentan las más altas prioridades de conservación debido a la alta biodiversidad, endemismo y vulnerabilidad (Myers et al. 2000). En el 2005 fue emitida por parte de Millennium Ecosystem Assessment (MA) una importante declaración sobre: i) la intrínseca relación entre biodiversidad y SE, y ii) la perspectiva de la protección de los SE como justificación de la conservación de la biodiversidad (MA 2005). Aunque esto ha sido un avance positivo para generar acciones que reduzcan la pérdida de biodiversidad y cambios en la provisión de los SE (Díaz et al. 2006), a la fecha hay pocos estudios que permitan una comprensión más amplia sobre cómo es la relación biodiversidad-SE (MA 2005). Este tipo de estudios brindaría conocimiento fundamental para la planificación de las estrategias de conservación (MA 2005; Onaindia et al. 2013).

En cuanto a la relación entre la biodiversidad y SE, la teoría existente establece que el número y la intensidad de las conexiones entre estos justifica la protección de la integridad biótica en los ecosistemas existentes y restaurados (Díaz et al. 2006). Se espera que una mayor provisión de SE requiera de una mayor biodiversidad, lo que justificaría la protección de los ecosistemas (Swift et al. 2004). Lo anterior, ha evidenciado la necesidad de conocer cómo es la relación biodiversidad-SE en los diferentes ecosistemas del mundo, en especial a escala de paisaje donde se ha reportado el mayor “stress” ambiental (Díaz et al. 2006). Actualmente, la relación biodiversidad-SE no ha sido ampliamente estudiada (Chan et al. 2007; Costanza et al. 2007; Naidoo et al. 2008; Schneiders et al. 2012). Algunos estudios han evidenciado una directa relación entre riqueza de especies, usada como proxy de biodiversidad, y producción primaria, ciclo de nutrientes, provisión de medicinas y control de plagas (Pfisterer and Schmid 2002; Hooper et al. 2005; Díaz et al. 2006; Costanza et al. 2007; Hector and Bagchi 2007; Thomas et al. 2008; Elmqvist et al. 2010). A su vez, varios autores han argumentado que en la provisión de los SE pueden incidir, a parte de la diversidad de especies, los diferentes componentes de la biodiversidad, (Díaz et al. 2006; Ridder 2008; Srivastava and Vellend 2010). De acuerdo a lo anterior, se necesitan estudios que provean una comprensión más amplia sobre cómo los diferentes componentes de la biodiversidad están relacionados con la provisión de los SE, en especial aquellos en los que se han registrado las mayores amenazas ambientales y, por ende, requieren prontas acciones de conservación (Costanza et al. 2007; Schneiders et al. 2012). En este sentido, en las últimas décadas los mayores impactos sobre la biodiversidad y SE en el mundo se han reportado a nivel de hábitat, requiriéndose prontas acciones para revertir la situación (Swift et al. 2004; MA 2005; Díaz et al. 2006). De tal modo que es evidente la urgente necesidad de conocer cómo la diversidad de hábitat, usada como proxy de biodiversidad, está relacionada con la provisión de los SE; cómo, dónde y cuándo la pérdida de biodiversidad afecta el bienestar humano. Lo anterior brindarían información relevante sobre la relación biodiversidad – SE y para el manejo y conservación de los ecosistemas existentes, restaurados y degradados (Turner 1989, Díaz et al. 2006, Sutherland et al. 2009).

Por otro lado, la protección de los SE ha sido usada para justificar las acciones de conservación de la biodiversidad (IUCN 2009). Perspectiva de conservación que puede contribuir en la optimización de las estrategias de conservación (MA 2005). Sin embargo, varios autores han destacado la necesidad de una comprensión más amplia sobre cómo los SE se relacionan espacialmente con la biodiversidad (Turner et al. 2007; Onaindia et al. 2013), y en qué medida la conservación de la biodiversidad asegura la provisión de múltiples servicios (MA 2005; Díaz et al. 2006). Actualmente, estas relaciones no han sido ampliamente estudiadas (Costanza et al. 2007; Schneiders et al. 2012.). Algunos estudios han reportado una baja correlación y moderada congruencia espacial entre la biodiversidad y SE (Chan et al. 2006; Schneiders et al. 2012.). Sin embargo, otros estudios han registrado una alta congruencia espacial entre la conservación de la biodiversidad y la provisión de SE (Turner et al. 2007; Egoh et al. 2009). La ambigüedad de estos resultados sugiere que es necesario realizar estudios en nuevas regiones del mundo y escalas espaciales que no hayan sido ampliamente investigadas (Egoh et al. 2009), estos brindarían una comprensión más amplia de esta relación (Onaindia et al. 2013) y la oportunidad de realizar eficientes planificaciones de las toma de decisiones (Turner et al. 2007). En este sentido, se hace necesario desarrollar estudios espacialmente explícitos a través de los cuales se identifique y cuantifique la biodiversidad y SE, y se puedan realizar profundos análisis sobre las congruencias espaciales entre estos dos recursos. Estos estudios contribuirían en la búsqueda de alternativas viables que permitan optimizar los esfuerzos de conservación. Es decir, proveerían información de gran relevancia para diseñar, gestionar e implementar acciones simultáneas de conservación, las cuales contribuirían en el desarrollo de acciones que conlleven a la sustentabilidad del paisaje.

Se han elaborado diversas clasificaciones de los SE (Costanza et al. 1997; Daily et al. 1997; de Groot et al. 2002; MA 2005; Wallace 2007; Haines-Young and Potschin 2010; TEEB

2010), siendo el esquema más aceptado el que entrega MA (2005). Según ese esquema, los SE se clasifican en servicios de provisión (e.g. agua, comida, fibras, medicina), servicios de regulación (e.g. control de erosión, regulación del clima, inundaciones y enfermedades, polinización), servicios de soporte (e.g. acumulación de suelo, fotosíntesis, ciclo de nutrientes), y servicios culturales (e.g. plenitud espiritual, recreación, educación, diversidad cultural).

Entre los diferentes SE esenciales para el desarrollo de la vida humana se ha identificado que: i) la provisión de agua, que suple las necesidades de consumo diario, ayuda a mantener las condiciones óptimas de aseo y riego agrícola, entre otros (de Groot et al. 2010); ii) control de erosión, que evita los daños del suelo y mantienen la productividad agrícola (Egoh et al. 2008); y iii) acumulación de suelo, que ayuda a mantener la integridad y funcionamiento del suelo y de los ecosistemas (de Groot et al. 2010); han registrado una disminución en su provisión en diferentes partes del mundo en las últimas décadas (MA 2005; Egoh et al. 2008; Bai et al. 2011; Onaindia et al. 2013). A la fecha, las investigaciones realizadas sobre estos SE se han enfocado en identificar y cuantificar tanto la producción como las principales áreas de provisión (Troy and Wilson 2006), en evaluar la oferta y demanda (Naidoo and Ricketts 2006), en realizar la valoración económica de estos (Nuñez et al. 2006), y en evaluar el impacto de la implementación de políticas de uso del suelo en la provisión de los servicios (Geneletti 2013). Sin embargo, se necesitan desarrollar investigaciones que evalúen y analicen de manera espacial y multitemporal cómo la provisión de estos SE está relacionada con la biodiversidad. Este tipo de investigaciones brindaría información de gran relevancia para conocer dicha relación y para realizar acciones de conservación que contribuyan en la mantención del bienestar humano.

Implicancias de la tesis

El paisaje forestal del sur de Chile, el cual ha sido identificado como un hotspot para la conservación de la biodiversidad en el mundo (Myers et al. 2000), no se escapa a esta tendencia mundial. A partir de la promulgación del Decreto Legislativo Nº 741 de 1974, el cual incentivo las plantaciones de especies comerciales, este paisaje ha exhibido una conversión cada vez mayor de hábitat forestal a plantaciones comerciales (CONAF 2006). A esto se le ha sumado el incremento de otras actividades antrópicas como la ganadería y cultivos agrícolas en las últimas décadas, las cuales han implicado la habilitación de grandes extensiones de suelo del paisaje forestal (Wilson et al. 2005). De tal modo que el paisaje forestal del sur de Chile ha experimentado una progresiva antropización en las debido a los intensos y permanentes CUS (Echeverría et al. 2006; Echeverría et al. 2012), lo que ha dado lugar a importantes cambios en la biodiversidad a nivel de especie (Bustamante and Grez 1995; Douglas 2000; Kelt 2000; Vergara and Simonetti 2004; Tomasevic and Estades 2008) y en la provisión de SE (Little et al. 2008; Little et al. 2009; Lara et al. 2009; Oyarzún et al. 2005; Oyarzún et al. 2011; Nahuelhual et al. 2014). De tal modo que lo anterior ha sido objeto de discusión entre las entidades gubernamentales ambientales, empresas forestales, y la comunidad en general, lo cual ha evidenciado la urgente necesidad de desarrollar acciones que conduzcan a la sustentabilidad del paisaje basadas en la comprensión de la relación entre CUS, biodiversidad y SE (Di Marzio and McInnes 2005; Conaf 2006).

En este contexto, el presente estudio aportará conocimiento importante sobre: i) la relación CUS, biodiversidad y SE, ii) cómo la biodiversidad se relaciona con la provisión de los SE, iii) cómo, dónde y cuándo la pérdida de biodiversidad impacta el bienestar humano, iv) en qué medida la conservación de la biodiversidad asegura la provisión de múltiples SE, v) en la identificación de áreas con alto valor para la conservación de la biodiversidad, v) en el conocimiento y comprensión del estado actual de especies arbóreas amenazadas, biodiversidad, SE y de los atributos del paisaje que los proveen, lo cual es información valiosa desde la que se deben sustentar la planificación de la conservación y toma de decisiones que contribuyan en la sustentabilidad del paisaje.

El presente estudio buscó evaluar: i) los impactos del CUS en los patrones espaciales del paisaje forestal y de la biodiversidad, y a su vez la influencia de estos impactos en la provisión de SE; ii) las sinergias entre SE y la congruencia espacial entre biodiversidad y SE; y iii) los impactos del CUS en los patrones espaciales del hábitat de especies arbóreas amenazadas y la influencia de estos cambios en la composición y estructura de sus poblaciones. De este modo,

Las principales hipótesis que guían esta investigación se resumen en:

- El CUS modifica los patrones espaciales del paisaje lo que genera pérdida de hábitat, biodiversidad y provisión de SE.

- El CUS antrópico conduce a una fragmentación sustancial del hábitat de especies arbóreas amenazadas, lo que a su vez causa una reducción del tamaño poblacional de la especie y variación en la composición de la comunidades vegetales asociadas.
- La pérdida de biodiversidad asociada al CUS, está relacionada, a su vez, con una sustancial pérdida en la provisión de SE.
- Las áreas cordilleranas aportan las mayores sinergias entre SE y congruencias espaciales entre biodiversidad y SE, por lo que deben ser consideradas áreas prioritarias para la conservación de estos dos recursos.

Esta tesis aborda las anteriores hipótesis en tres capítulos:

1) En el capítulo 2, se evalúan y analizan los impactos del CUS en los patrones espaciales del paisaje y del hábitat de poblaciones remanentes de *Fitzroya cupressoides*, especie arbórea amenazada categorizada en Peligro (IUCN 2013). Adicionalmente, se relacionan los cambios en los patrones espaciales del hábitat con la composición de las poblaciones y comunidades asociadas.
2) En el capítulo 3, se evalúa y analiza el impacto del CUS en los patrones espaciales de la diversidad de hábitats de bosque nativo, usada como proxy de biodiversidad, y la influencia de estos cambios en la producción de los SE provisión de agua, control de erosión y acumulación de suelo.
3) En el capítulo 4, se evalúa y analizan las sinergias entre SE y las congruencias espaciales entre la biodiversidad y provisión de los SE estudiados, en orden a contribuir con información de gran relevancia para la planificación y toma de decisiones en conservación en común para ambos recursos.

Finalmente, en el capítulo 6 se presentarán conclusiones generales que integrarán los resultados obtenidos de los demás capítulos.

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Capítulo II

Impacts of Anthropogenic Land Use Change on Populations of the Endangered *Fitzroya cupressoides* in Southern Chile. Implications for its Conservation.

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Abstract

Land-use change may have negative effects on endangered species by modifying the habitat and population dynamics. The habitat of *Fitzroya cupressoide*s (Mol.) Johnst (Cupressaceae), an endemic endangered conifer from temperate forests of southern Chile and Argentina, has been severely transformed as a result of land-use change and overexploitation. By means of satellite images, the impact of land-use change on the spatial patterns of *F. cupressoides* habitat between 1999 and 2011 was evaluated at the landscape level. Eight plots of 20 x 25 m were established in four remaining populations of the species to assess their current status. In each plot, the *F. cupressoides* density and species richness of the associated communities were recorded and analysed together with the spatial patterns at the population and community level. The loss of *F. cupressoides* potential habitat in the landscape was 46%. The loss of habitat for the four populations ranged from 38% to 100%. The density of *F. cupressoides* was the lowest where the size of the habitat was smaller. The number of species was higher as the size of the habitat patches was smaller and the matrix was more dominated by human-related land uses. In this multiscale study, land-use change was associated with a loss of *F. cupressoides* potential habitat, resulting in differences in the habitat spatial patterns of the four *F. cupressoides* populations in 2011. These differences influenced the composition of remaining populations and communities. A landscape approach is suggested as a strategy for the planning of *F. cupressoides* conservation.

Keywords: anthropogenic processes, Chilean larch tree, habitat assessment, landscape change, spatial patterns.

Introduction

Land-use change may lead to fragmentation and loss habitat in forest landscapes, both of which have been recognised as two of the greatest threats for forest ecosystems worldwide (Noss, 2001; Baillie et al., 2004). Land-use change modifies habitat spatial patterns, such as habitat connectivity, density of habitat patches, and interior habitat sizes in native forest patches (Lindenmayer & Fischer, 2006). These changes may in turn influence diverse ecological attributes of biodiversity at the species and community levels, such as species richness and individual density (Laurance et al., 2000; Bustamante et al., 2003; Simonetti et al., 2006). In this sense, fragmentation and loss habitat can have negative effects on species survival by directly affecting habitat quality and quantity (Tominatsu & Ohara, 2003), both of which may alter the dynamics of the species population. This alteration could lead to a local and total extinction of the most vulnerable species (Bennett, 2003).

Several authors agree on the need to investigate the impact of land-use change on endangered species (Turner, 1989; Lindenmayer & Fischer, 2006). Such studies can provide useful information to aid understanding of i) the way in which land-use change can alter the landscape spatial pattern (composition and spatial configuration) of unique habitats on which endangered species depend, ii) the influence of these alterations on species composition, and iii) the influence of the matrix and the minimum buffer zone distance needed for management of the remaining interior habitats (Oliveira et al., 2004). Multiscale studies (landscape, habitat and species) could provide valuable information for the conservation of endangered species (Vergara & Simonetti, 2004).

Chilean Temperate forest, which has been classified as a hotspot for biodiversity conservation due to its high endemism level (Myers et al., 2000), has experienced high levels of degradation, fragmentation and loss in the past few decades as a result of land-use change (Echeverría et al., 2012). These impacts have led to significant changes in species composition (Echeverría et al., 2007), richness of bird species (Vergara & Simonetti, 2004) and spatial distribution of endangered species (Altamirano et al., 2007).

Fitzroya cupressoides, an endemic and monotypic conifer of the temperate forests of southern Chile and Argentina, has been categorised as endangered on the IUCN Red List of threatened species (IUCN, 2013). Individuals of this species can live for more than 3,600 years, becoming the world’s second longest lived species, which gives it a high and global scientific value (Lara & Villalba, 1993). Since the arrival of Europeans to the south of Chile in 1850, *F. cupressoides* has suffered overexploitation due to its beautiful and decay-resistant wood. In Chile, *F. cupressoides* is found in the Coastal Range, Andean Range and in the Central Depression (Fraver et al., 1999). In the Central Depression, 13 sites are known; five consist of small remaining populations, and eight support scattered small trees and saplings that are embedded in a landscape severely transformed by human activity (forest logging for firewood and pasture expansion for cattle grazing) (Fraver et al., 1999). Genetic studies indicate that the *F. cupressoides* populations in the Central Depression have the greatest genetic differences relative to the remaining populations in Chile and Argentina. Therefore, these highly endangered populations represent ancient populations from the glacial era from which the rest of the populations are derived (Premoli et al., 2003).

While several studies have been carried out in reproductive biology, ecology and distribution of the species (Armesto et al., 1992; Donoso et al., 1993; Fraver et al., 1999), presently, there are no studies that not only quantify the impact of land-use change on the spatial patterns of *F. cupressoides* habitat but also understand the effect of habitat fragmentation and loss on the composition of *F. cupressoides* populations. This type of information is crucial for the assessment of the current status of *F. cupressoides* populations that have been most severely altered in Chile.

In this study, we assessed the impact of land-use change on the spatial patterns of the forest landscape and *F. cupressoides* habitat in the Central Depression in southern Chile. Additionally, we related changes in spatial patterns to the current composition of four *F. cupressoides* remaining populations. We hypothesise that human-induced land-use change has substantially fragmented the remaining *F. cupressoides* habitat, which in turn caused a reduction in the density of the species and variation in community composition.

Study area

The study area was defined by the distribution of four *F. cupressoides* remaining populations located in the Central Depression, Chile (Fraver et al., 1999) and by the extension of Landsat satellite images used to determine land-use change (41º20´S and 41º50´ S) (Fig. 2.1). The study area comprises 1,430 ha, with maximum elevation of 200 m a.s.l. The mean temperature is 11.5ºC and the annual rainfall is 1,912 mm. The area is characterised by oceanic cold temperate climate. The landscape in this study area is dominated by patches of broad-leaved evergreen native forest, also known as the Valdivian Rainforest.

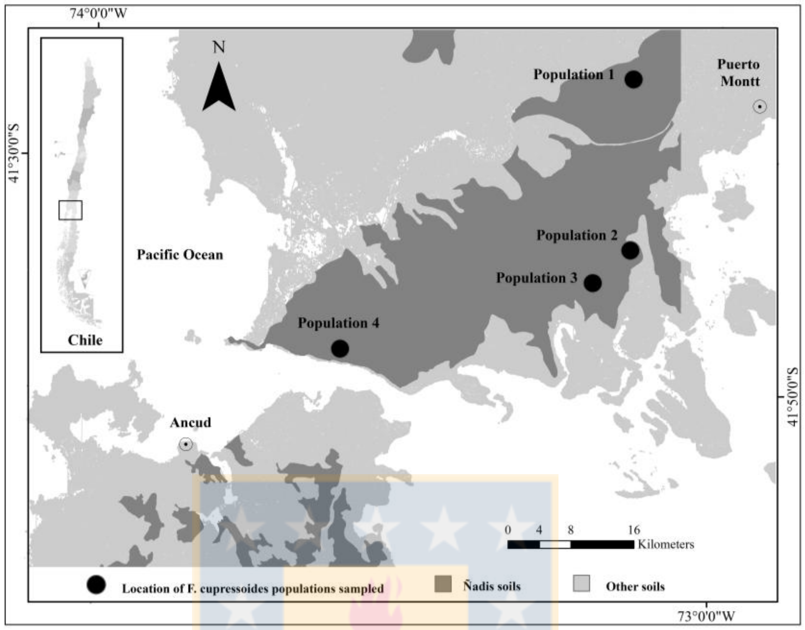

FIG. 2.1 Location of *F. cupressoides* populations in the Central Depression, Chile.

Methods

Land-use change analyses were performed based on previous study in which maps of land use (pixel 30 x 30 m) were derived from Landsat satellite images for the years 1999 and 2011 (Echeverria et al., 2012). In these images, the various categories of land use type were identified, as follows: native forest, shrubland, wetland, grassland, exotic species forest plantations and other uses (bare soil, urban area and water bodies).

Changes in spatial patterns of the forest landscape were evaluated by analysing the total area (ha) and number of native forest patches over time, using FRAGSTATS (version 3.3) (Mcgarigal et al., 2002) and ArcGIS 9.3.1 spatial analyst extension (ESRI, 2009).

This study evaluated four of the five currently known *F. cupressoides* remaining populations in the Central Depression. One of the populations was not evaluated due to limited access to information. Spatially explicit information on the location of populations was provided by the Corporación Nacional Forestal (CONAF). The four *F. cupressoides* remaining populations were found in patches of broad-leaved evergreen native forest. These are present in poorly drained acidic soils known as Ñadis (FAO-UNESCO, 1971). Ñadis means seasonal swamp in the indigenous language (Fig. 1). In this area, *F. cupressoides* is often associated with *Amomyrtus luma, Drimys winteri, Laureliopsis philippiana, Saxegothaea conspicua and Weinmannia trichosperma* (IUCN, 2013).

Previous study conducted in the same area showed that there is a relationship between the *F. cupressoides* habitat and the interior area of small native forest patches (Fraver et al., 1999). These patches have a core area of < 3ha and are located at least 137 m from the patch edge. Fraver et al. (1999) reported that the *F. cupressoides* habitat in the native forest patches of the Central Depression would not exist if edge effects (altered species composition, community structure, and microclimate) occur at a distance of less than 137m. Following the approach used by Fraver et al. (1999) for habitat parameters, our study defined the *F. cupressoides* habitat as that inner native forest area with a core area < 3 ha and a buffer zone distance of at least 120 m.

F. cupressoides potential habitat were evaluated by analysing the core area and buffer zone distance of the native forest patches, using "Land Change Modeller for Ecological Sustainability" (LCM) extension of IDRISI Andes software (Clark-Lab, 2007). Maps of native forest cover as well as values of core area and buffer zone distance were entered into LCM. Maps of potential habitat were obtained for each year of study. Changes in spatial patterns of *F. cupressoides* potential habitat were evaluated by analysing the total area (ha) and number of potential habitat patches over time, using FRAGSTATS (version 3.3). Subsequently, the potential habitat patches that include the four *F. cupressoides* remaining populations were spatially identified. This identification allowed us to analyse the changes in total area for each potential habitat patches over time.

In 2011, fieldwork was carried out in the four *F. cupressoides* remaining populations (Fig. 1). In each population, the density and regeneration of *F. cupressoides* trees were recorded. Due to the small size of native forest patches containing the populations and probable edge effects, two 20 x 25m sampling plots were randomly established within each patch. To facilitate data collection, each plot was divided into ten 5 x 10 subplots, following the methods of Peet et al. (1998). Tree was defined by a diameter at breast height (DBH) ≥ 5 cm and a height ≥ 2 m. *F. cupressoides* regeneration was defined as seedlings with DBH < 5 cm and/or height < 2 m. Also, the composition of the communities associated with native forest patches with the presence of *F. cupressoides* was estimated by measuring total plant species richness (Jiménez-Valverde & Hortal, 2003). This information was analysed together with the changes in habitat spatial patterns of each population in 2011, in order to evaluate relationships between them.

Results

Changes in spatial patterns of the forest landscape

The study landscape registered a loss of 18% of native forest between 1999 and 2011 at a rate of 1.6% per year (Fig. 2.2). In 1999, the study landscape comprised 9,478 native forest patches, equivalent to 46,129 ha (Fig. 2.2). By 2011, the number of native forest patches increased more than twice to 22,446 and the total area declined to 37,948 ha. By 2011, the native forest was restricted to small patches sparsely distributed across the landscape (Fig. 2.2), and the shrubland became the dominant land cover type in the study landscape.

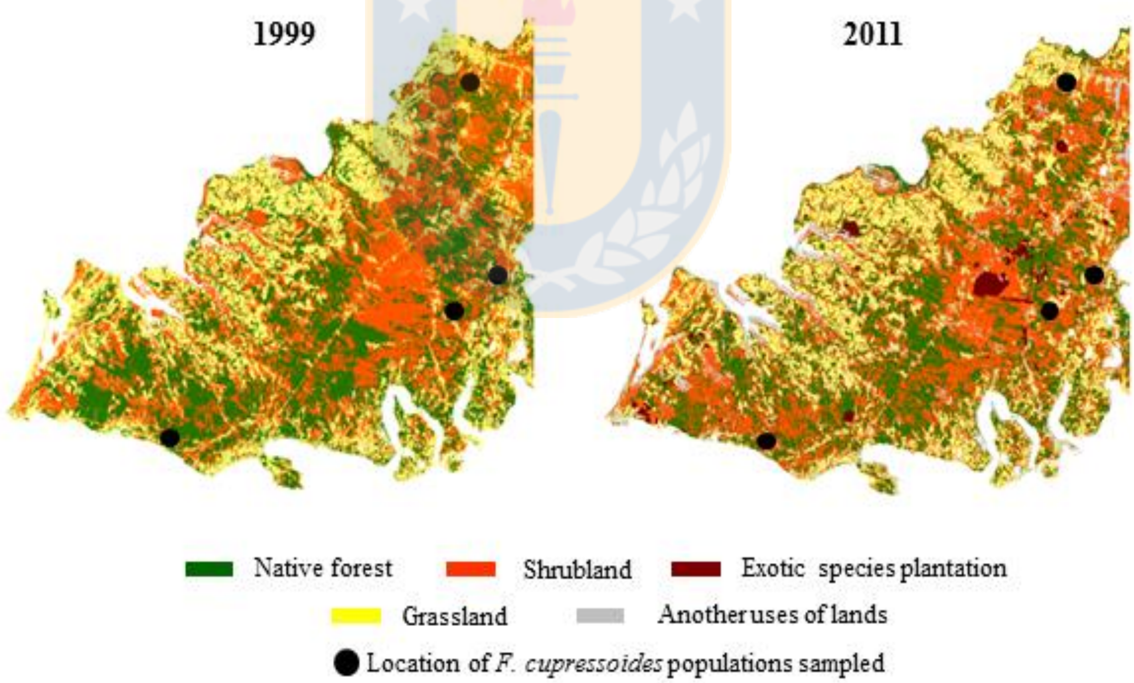

FIG. 2.2 Patterns of land use change and location of sampled *F. cupressoides* populations between 1999 and 2011.

Changes in the spatial patterns of *F. cupressoides* habitat

A loss of 46% of *F. cupressoides* potential habitat was recorded between 1999 and 2011 at a rate of 5.1% per year. In 1999, the *F. cupressoides* potential habitat was distributed in 112 native forest patches, equivalent to 26,122 ha. By 2011, the number of *F. cupressoides* potential habitat patches decreased to 36 and the total area declined to 14,076 ha.

In 1999, the four *F. cupressoides* populations were found in separate potential habitat patches ranging from 5 to 36 ha. By 2011, the four *F. cupressoides* populations were found in separate potential habitat patches ranging from 0.4 to 27 ha (Table 2.1). A loss of 38%, 100%, 54% and 79% of *F. cupressoides* potential habitat was recorded over the entire study period for populations 1, 2, 3 and 4, respectively (Table 2.1).

TABLE 2.1. Habitat spatial patterns, composition of *F. cupressoides* populations and plant species richness in native forest patches with presence of *F. cupressoides* habitat in 2011.

Population	Spatial patterns					Composition of *F. cupressoides* populations		Species richness of the associated communities	
	Area of the native forest patches with presence of *F. cupressoides* habitat (ha)	Habitat loss of *F. cupressoides*	Matrix Forest-Shrubland-Grassland			Tree/ha	Regeneration Seedling/ha	Richness of native species in sampling plots	Richness of exotic species in sampling plots
1	27	38%	50%	24%	26%	2,270	4,340	7	1
2	0.4	100%	2%	36%	62%	340	1,350	18	3
3	12	54%	35%	29%	36%	360	2,470	9	2
4	0.5	79%	3%	51%	46%	40	0	16	3

Habitat spatial patterns and population and community composition in 2011

A lower density of *F. cupressoides* seedlings and trees seem to be related to a greater loss of habitat (Table 2.1). In 2011, for the population that registered a loss of 38% of potential habitat, the density of *F. cupressoides* trees was 2,270 ind/ha and regeneration was 4,340 ind/ha. On the other hand, for the population that registered a loss of 100% of potential habitat, the density of trees was 40 ind/ha and regeneration was nill (Table 2.1).

The *F. cupressoides* populations that registered greater loss of habitat were those surrounded by an anthropic matrix and were associated with communities with the greatest number of native and exotic species (Table 2.1). In 2011, the population that registered a loss of 38% of potential habitat was located in the largest native forest patch (27 ha), embedded in a matrix dominated by native forest, and associated with a community with the lowest richness of exotic plant species (Table 2.1). On the other hand, the population that registered a loss of 100% of potential habitat was located in the smallest native forest patch (0.4 ha), embedded in a matrix dominated by grassland and associated with a community that registered the greatest richness of exotic plant species (*Plantago truncate, Poa annua* and *Prunella vulgaris*) (Table 2.1).

Discussion

Changes in the forest landscape and *F. cupressoides* habitat

Our results showed a substantial loss of native forests in the landscape studied between 1999 and 2011.The rate of forest loss reported in this study (1.6%) is lower than that recorded for other hotspot landscapes that have also been severely transformed, such as temperate forest Maulino in central Chile (Echeverría et al., 2006) and tropical montane forest of Chiapas, Mexico (Cayuela et al., 2006), whose rates are 4.5% and 3.05%, respectively. The loss of native forest area was associated with a drastic change in the number of patches during the study period. This was evident in the increased number of patches from 9,478 in 1999 to 22,400 in 2011, and this increase represents one of the main symptoms of fragmentation (Lindenmayer & Fischer, 2006). The trend of fragmentation shown in this study is similar to the one registered in other temperate landscapes in Chile, where the occurrence of endangered tree species was recorded (Bustamante & Castor, 1998), as well as in tropical montane landscapes in Mexico, which has been recognized as having global conservation importance (Cayuela et al., 2006). This trend of loss and fragmentation of native forest in the study landscape was associated with a significant decrease (46%) of the *F. cupressoides* potential habitat in the last decade. This significant decrease was associated with a drastic decrease in the number of potential habitat patches (112 to 136), that in turn, increased the severity of changes in the *F. cupressoides* potential habitat during the study period. If the current trajectory of loss and fragmentation of native forest continues, an increase in the loss of *F. cupressoides* potential habitat could be expected.

Changes at the population and community level in 2011

We observed that the habitat loss was different in each of the four *F. cupressoides* populations, which implied that each population registered differences in the habitat spatial patterns in 2011. These differences were associated with the densities of the populations. That is, the smallest habitat sizes were those associated with the lowest densities of *F. cupressoides*. A similar trend is reported for other endangered species in Chile, such as *Nothofagus alessandrii* (Bustamante & Castor, 1998) and *Legrandia concinna* (Altamirano et al., 2007), and for various species of birds (Vergara & Simonetti, 2004, Simonetti et al., 2006) and some populations of fauna (insectivorous birds, primates and mammals) in the Amazon rainforest (Laurance et al., 2000). Our results indicate that a smaller habitat size may decrease the density of the *F. cupressoides* population, and this decrease may increase the risk of extinction for this species.

Our results showed that differences in the habitat spatial patterns of the four *F. cupressoides* populations were related to differences in the plant species richness of the associated communities. As the size of native forest patches with presence of *F. cupressoides* habitat decreased, the number of native and exotic species plants increased. In the Chilean Temperate forest, a similar trend was reported for the plant communities composition (Bustamante & Grez, 1995), bird communities (Vergara & Simonetti, 2004) and small mammals (Kelt, 2000). In boreal (Chávez & Macdonald, 2010) and tropical areas (Trauernicht & Ticktin, 2005), this change was less dramatic, perhaps due to forest landscapes being slightly transformed. Furthermore, there are other factors that may influence changes in the plant species richness of the communities associated with native forest patches with presence of *F. cupressoides* habitat. One of these factors is the matrix, which may induce several types of drastic abiotic changes at the edge of native forest patches, resulting in the potential establishment and recruitment of generalist species plant (Bustamante et al., 2003). As has been shown, besides the difference in size of native forest patches with presence of *F. cupressoides* habitat, there is also the difference in composition of the matrix that surrounds each native forest patch. The combination of these two factors may have influenced the establishment of generalist native and exotic species in each community.

Implications for conservation

Central Depression is located in a landscape that has been subjected to constant anthropic pressure in the last centuries (Torrejón et al., 2011). From the XVI to the XIX century, important changes in the spatial patterns of native forest were mainly generated by massive and continuous wood extraction (Torrejón et al., 2011), whereas, in the last four decades the greatest changes in the spatial patterns of native forests has been generated by pasture expansion for cattle grazing and commercial plantations (Echeverría et al., 2007, 2012). Presently, Central Depression is part of a landscape under increasing urban pressure from the cities of Puerto Montt and Puerto Varas (Fraver et al., 1999). As a result, this anthropic pressure may increase the changes in the spatial patterns of the native forest and *F. cupressoides* habitat due to anthropogenic land use (Armesto et al., 1992; Wilson et al., 2005).

Owing to the historical overexploitation that has affected the *F. cupressoides* forest (Torrejón et al., 2011), the species was listed in Appendix I of the Convention on International Trade in Endangered Species (CITES) in 1975 and declared a “Natural Monument” by the Chilean government in 1976. Our multiscale study showed a progressive and severe loss of native forest with presence of *F. cupressoides* between 1999 and 2011. This loss involved drastic decreases of potential habitat for the species in the last decade, which in turn, reduced the habitat sizes of the four *F. cupressoides* populations studied. In 2011, the differences in habitat sizes of the four populations were associated with differences in their density. If habitat loss continues, it is possible that the population density could decrease, which may increase the risk of extinction for this species. Therefore, urgent conservation efforts are needed. As a first step, we suggest that a landscape approach be taken, using the corridor-patch-matrix model (Lindenmayer & Franklin, 2002) as an appropriate strategy for planning the conservation. Given the current configuration of the landscape, in which the native forest is restricted to small patches sparsely distributed across the landscape, the use of this model is well suited. The main objective of the corridor-patch-matrix model is to maintain the quality and quantity of native forest patches through the management of the matrix (Lindenmayer & Franklin, 2002). The conditions of the matrix may be more important in determining the survival of the species than the isolation of patches (Lindenmayer & Franklin, 2002). In the landscape studied, the management of the matrix should focus on buffer sensitive areas that contribute to improve the connectivity among native forest patches and to increase the ability of the matrix to support the *F. cupressoides* populations. We recommend that this strategy be complemented with land use planning, which must consider the conservation of native forest patches through sustainable production practices, such as agroforestry. The design and implementation of the proposed strategy requires studies that identify and evaluate the buffer sensitive areas and their connectivity. Moreover, this strategy must be supported in a framework of environmental policies that must be issued by the Chilean State. As a second step to implement conservation efforts, we suggest that the Chilean State mandates to expand the protected areas in order to protect other *F. cupressoides* populations to promote conservation of the species. At the present moment, there is only one protected *F. cupressoides* population in the Central Depression, which is the Monumento Natural Lahuen Ñadi. As a third step, we suggest to strengthen the existing restoration programs of *F. cupressoides* and to develop new programs, both of which aim to improve the quality of habitats and the natural dynamic of *F. cupressoides*. These programs should include i) the native forest patches that registered potential habitat for the species, ii) monitoring to assess the viability and success of these programs, and iii) comprehensive investment of capital. It has been documented that researchers from the Universidad Austral de Chile, land owners and CONAF are participants in restoration programs (Premoli et al., 2013). The *F. cupressoides* conservation is a challenge that requires the commitment and the active participation of land owners, the Chilean State and the general community.

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Capítulo III

Impacts of Land-Use Change on the Biodiversity and Ecosystem Services in the Hotspot of Valdivian Temperate Forest in Southern Chile

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Abstract

Land-use change (LUC) impacts biodiversity, which is intrinsically related to ecosystem services (ES). There is limited information on how LUC affects this relationship at the landscape level, where major impacts have been recorded. Such information would provide crucial knowledge for planning conservation strategies. The forest landscape of southern Chile, which includes the Hotspot of Valdivian Temperate Forest, has undergone a progressive LUC and important changes in biodiversity in recent decades. Because of this there is an urgent need for planning conservation strategies in this hotspot. Therefore, this landscape represents a good opportunity to study the relationship among LUC, biodiversity and ES. This study assessed, using satellite images and landscape metrics, the impact of LUC on the spatial patterns of the habitat diversity of the native forest (HDNF) in the Río Cruces watershed, Chile, between 1986 and 2011. HDNF was used as a proxy of biodiversity. The ES water supply, control of erosion, and soil accumulation were mapped and quantified. Using generalized linear models (GLMs), the relationship between changes in the spatial patterns of the HDNF and ES provision were analyzed. Between 1986 and 2011, 12% of HDNF area, more than 150% increase in the number of patches with presence of HDNF, and loss of 0.20 in the Shannon diversity index were recorded. The greatest decrease in the provision of ES was recorded for erosion control (346%), and the lowest for water supply (11%). The loss of provision of the ES was explained by the interaction of changes in the spatial patterns HDNF ($p < 0.001$). This study constitutes the largest analysis of the relationship between impacts of LUC on HDNF and ES that has been done in Chile. It provides fundamental information for optimizing the conservation strategies and provision of multiple ES.

Key words: anthropogenic processes, changing landscape, habitat diversity, natural capital, spatially explicit models.

INTRODUCTION

In the last century, the biggest changes for ecosystems in the world, due to global change drivers, have been reported (MA 2005), which has led to a rapid reduction of biodiversity, broadly defined as the richness and abundance of genes, species, and ecosystems (Balmford & Bond 2005). Because of these significant changes, the United Nations Organization (UNO) promulgated, in 1992, the Convention on Biological Diversity, which proposes among its objectives to achieve a significant reduction in the rate of biodiversity loss. Later, in 2005, the Millennium Ecosystem Assessment issued an important declaration about: i) the intrinsic relationship between biodiversity and ecosystem services (ES), broadly defined as the benefits provided by ecosystems that contribute to making human life both possible and worth living; and ii) the perspective of the protection of the ES as a justification for biodiversity conservation (MA 2005). Although this has been a positive advance to generate action that reduces biodiversity loss and changes in the provision of ES (Díaz *et al.* 2006), currently there are few studies that allow an understanding of how drivers of global change impact biodiversity and the influence of these changes on the provision of ES (MA 2005). Such studies would provide crucial knowledge for planning conservation strategies (MA 2005; Onaindia *et al.* 2013).

Several studies have identified the land-use change (LUC), an anthropogenic process, as the main driver of ecosystem modification in the world (Baillie *et al.* 2004), because this may lead to habitat fragmentation and loss in forest landscapes (Lindenmayer & Franklin 2002). This is evidenced by alterations of landscape spatial patterns (composition and spatial configuration), such as total habitat area (ha), habitat connectivity, and habitat patch density (Lindenmayer & Fischer 2006). These changes, in turn, can impact the diversity of habitats, resulting in a loss of biodiversity (MA 2005; Díaz *et al.* 2006), because this has a key role in the maintaining of the different levels and attributes of the biodiversity (Lindenmayer *et al.* 2006).

Biodiversity loss usually involves unexpected and irreversible alterations in the provision of the ES, which affects the wellbeing of humanity (MA 2005). Currently, this complex relationship has not been widely studied (Costanza *et al.* 2007; Schneiders *et al.* 2012). Only a few studies have evidenced a direct relationship between species richness, used as a proxy of biodiversity, and the provision of ES. Pfisterer & Schmid (2002) and Díaz *et al.*

(2006) have reported that the increase in plant species richness positively affects primary production. Additionally, Costanza *et al.* (2007) report that in the ecoregion of North America, a positive correlation exists between plant species richness and primary production at a temperature of 13^{o}C. Additional effort should be conducted to allow an understanding of how the different components of biodiversity are related to the provision of the ES (Costanza *et al.* 2007; Schneiders *et al.* 2012). Moreover, the greatest impacts on biodiversity and provision of ES have occurred at the level of habitat, landscape and ecosystem (Swift *et al.* 2004; MA 2005). This has highlighted the urgent need to understand the relationship between this component of biodiversity and ES at large-scale (Swift *et al*. 2004; Díaz *et al.* 2006), in order to develop conservation actions to ensure the maintenance of both resources in changing landscapes (Iverson *et al.* 2014).

The forest landscape of southern Chile, which includes Valdivian Temperate Forest, has been identified as a hotspot for biodiversity conservation in the world (Myers *et al.* 2000). This landscape support the provision of important ES that are the basis for the human well-being, such as: water supply for the consumption by humans, agricultural and aquaculture activities, and fishing (Oyarzún *et al.* 2005); erosion control and soil accumulation services, which are important for the soil productivity and conservation of the ecosystem integrity (de Groot *et al.* 2010), so that the provision of food, wood, fiber and medicine may be maintained over time (Díaz *et al*. 2006). Also, erosion control and soil accumulation services are related with the water flow regulation services, which regulate the water distribution along the surface of the landscape by avoiding runoff and flooding problems (de Groot *et al*. 2002). The forest landscape of southern Chile has undergone a progressive anthropization in recent decades due to intense and permanent land use changes (Echeverría *et al.* 2006; Echeverría *et al.* 2012), which have led to important changes in biodiversity at the species level (Bustamante & Grez 1995; Vergara & Simonetti 2004) and the provision of the ES water supply and erosion control (Little *et al.* 2008; Oyarzún *et al.* 2011).

The Río Cruces watershed in southern Chile is a landscape that represents a good opportunity to study the relationship among LUC, biodiversity and ES. This watershed is characterized by a high biodiversity and progressive anthropization registered in the last decades (DGA 2004; Conaf 2006) and, particularly, by a high conservation priority (Myers *et al.* 2000). Additionally, in the watershed exists the Sanctuary of Nature "Carlos Anwandter", which has been recognized as the first Ramsar site of importance in Chile and a first Neotropical Wetland of International Importance (Di Marzio & McInnes 2005; Conaf 2006). This Ramsar site is of great importance due to its biological diversity and that provide shelter to a significant number of seasonal migratory waterfowl (Di Marzio & McInnes 2005). Since the promulgation of Legislative Decree No. 741 of 1974, which encouraged the planting of commercial species in the country, the Rio Cruces watershed has exhibited a growing conversion of native forest habitat to commercial plantations (Conaf 2006). In recent years, it has been reported that various impacts derived from the forest industry have led to a significant loss of wildlife (Di Marzio & McInnes 2005) and a significant increase in the export of sediment in different affluents of the watershed (Oyarzún *et al.* 2011). The foregoing has been the focus of discussion among governmental environmental entities, forestry companies, and the general community, which have highlighted the need to implement effective conservation strategies (Di Marzio & McInnes 2005; Conaf 2006). Therefore, studying the impacts of anthropogenic LUC on biodiversity, and, in turn, the influence of these impacts on the ES is of great importance for the understanding of this relationship and to carry out actions which maximize the conservation HDNF and provision of multiple ES.

In this study, we assessed the impact of LUC on the spatial patterns of the HDNF and, in turn, the influence of these changes on the provision of the ES water supply, control of erosion, and soil accumulation in the Rio Cruces watershed, Chile. Using spatially explicit models, we analyzed the changes in the provision of ES. Through generalized linear models (GLMs) we analyzed the relationship between changes in the spatial patterns of HDNF and provision of the ES between 1986 and 2011. We hypothesise that human-induced LUC generated HDNF loss, which, in turn, caused a decrease in the provision of the ES.

METHODS

Study Area

The Río Cruces watershed is located in the Los Ríos region, in southern Chile (Fig. 3.1). It is located between the cordilleras of the Andes and the coast (39º17´S y 39º50´ S), north of the city of Valdivia. It has an area of 3,640 km2, a range between 56 and 80 km, and reaches up to 826 m a.s.l. in the Coastal Range. The mean temperature is 12º C and the annual rainfall is

2,293 mm (Di Castri & Hajek 1976). The watershed is characterized by a warm temperate climate in the north and temperate rain in the southern (DGA 2004). The landscape in the Cordillera is dominated by native evergreen forest, also known as the Valdivian Temperate Forest, and commercial plantations of pine and eucalyptus. In contrast, the plain area is dominated by agricultural and livestock pasture. The watershed has a large human population of about 206 000, which is equivalent to population density of 46 people km^{-2} (DGA 2004). The main economic activities of the watershed correspond to forestry, agriculture and livestock farming (DGA 2004).

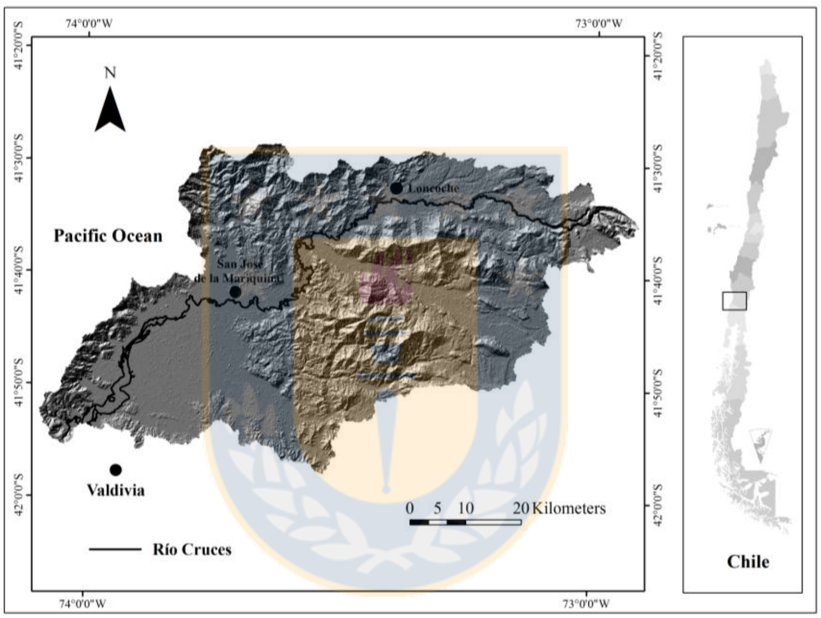


Fig. 3.1. Location of the Río Cruces watershed, Chile.

A total of 262 subwatersheds, ranging in size from 500 to 4,000 ha, were analyzed. These were defined as the spatial unit of analysis. The delimitation of the subwatersheds was carried out through the ArcGIS 9.3 Arc Hydro Tools extension (ESRI 2009).

Analysis of Biodiversity

We analyzed the diversity of native forest habitats as a proxy for biodiversity at the landscape level. This was determined by the presence of native forest habitats in different vegetation levels (Luebert & Pliscoff 2004), climatic zones (Schlatter *et al.* 1995) and soil orders (Ciren 2003) present in the study area. Biodiversity mapping was carried out through the following maps: 1) Map of native forest habitat, which was extracted from land-use map for the year 2011. In this map the following categories of use were identified: native forest habitat, exotic species forest plantations, shrubland, grassland, wetland, and other uses (bare soil, urban area and water bodies). 2) Map of vegetation levels, which was provided by Estudio de Clasificación de Pisos de Vegetación (Luebert & Pliscoff 2004). 3) Map of climatic zones, which was provided by Sistema de Ordenamiento de la Tierra (Schlatter *et al*. 1995). 4) Map of soil orders, which was provided by Estudio Agrológico de Suelos de Chile (Ciren 2003). Through the overlapping of these maps was obtained different types of habitat. This calculation was carried out through the ArcGIS 9.3 spatial analyst extension (ESRI 2009). Biodiversity was assessed through Shannon diversity index, which is a landscape metric that relate the variety and abundance of different habitat types in the landscape. This analysis was carried out through FRAGSTATS (version 3.3) (Mcgarigal *et al.* 2002). Map of biodiversity was entered into FRAGSTATS software to obtain value of the Shannon diversity index for each spatial unit of analysis. The assessment of the impacts of LUC on HDNF was carried on the basis of land-use maps for the years 1986, 2001, and 2011. The changes in HDNF were analysed by comparison of the following landscape metrics over time: total area (ha), total number of patches with HDNF, index richness of habitat patches (the number of different types of habitat patches) and Shannon diversity index. Landscape metrics were calculated through FRAGSTATS (versión 3.3) (Mcgarigal *et al.* 2002). Maps of biodiversity were entered into FRAGSTATS software to obtain value of the landscape metrics. This tool allowed us assessment the spatial patterns of biodiversity in the landscape studied.

N-Spect Model and Ecosystem Services

The software N-Spect (Non Point Source Pollution and Erosion Comparison Tools) was used to map and analyze the provision of the ES water supply and erosion control. The N-Spect software was developed by The National Oceanic and Atmospheric Administration (NOAA) of the United States, and is used as an extension of Arc GIS 9.3 (ESRI 2009). This software was developed to analyze and predict sediment discharges and the potential impacts on water quality from nonpoint sources of pollution (NOAA 2009). It is software that serves as a tool for management and decision-making concerning water resources, land use planning, agricultural policies and practices. N-Spect is a spatially explicit model that examines the relationship between land cover, nonpoint source pollution, and erosion (NOAA 2009), through a combination of information from the physical environment (elevation, slope, soils, and precipitation) (World Resources Institute 2006). The software generates maps that register the estimates of cumulative runoff and sediment loads (NOAA 2009).

The parameterization of N-SPECT was carried out for 1986 using the following inputs: 1) digital elevation model (DEMs) 30 x 30 m, 2) map of land use (píxel 30 x 30 m), 3) maps of precipitation and rainfall erosivity (R factor). By evaluation of the average of daily rainfall data registered in 12 meteorological stations present in the study area and the method proposed by Angulo-Martínez & Beguería (2009) the average annual rainfall and the coefficient of rainfall erosivity (R factor) were calculated, which were spatialized through geostatistical method of topoclimatological interpolation interpolation (Díaz *et al*. 2010). 4) Values of vegetation cover (C factor) were estimated on the basis of values proposed by Wischmeier & Smith (1978). 5) Coefficient values soil erodibility (K factor), were calculated from the information of the Estudio Agrológico de Suelos de Chile (Ciren 2003) and using the equation of the nomogram proposed by Wischmeier and Smith (1978). 6) Values for hydrological groups for each soil series, which are related to the number curve method (NOAA 2009), were estimated from soil texture data (Ciren 2003). Information about the different soil series present in the study area was entered into a map, which was provided in the Estudio Agrológico de Suelos de Chile (Ciren 2003). Model validation was carried out for entire study period with the data registered in three pluviometrica and sediment stations of the Dirección General de Aguas (DGA) present in the study area. The goodness of model fit was assessed by the method of quantitative assessment of "relative efficiency (E_{rel})" proposed by Krause *et al*. (2005) and Thanapakpawin *et al*. (2006).

Water Supply

This ES is the volume of water produced per unit area (m3/ha) (de Groot *et al.* 2010) that is potentially viable for human consumption (Chan *et al.* 2006). The amount and distribution of rainfall is the main determinant of the amount of water produced in a watershed (Egoh *et al.* 2008). Rainfall patterns, in turn, depend mainly on abiotic factors, such as regional climate and topography systems and not on ecosystems per se (van Jaarsveld *et al.* 2005). This service was modeled on the basis of rainfall, vegetation cover, soil and topoclimatological variables such as latitude, latitude and proximity to the sea, which determined the annual cumulative runoff.

Erosion Control

This is the ability of natural vegetation to curb erosion by holding onto soil (Egoh *et al.* 2009), which is measured as the amount (ton/ha) of sediment exported (de Groot *et al.* 2010). Soil erosion removes nutrients and reduces fertility (de Groot *et al.* 2002), and may generate sedimentation and eutrophication of nearby rivers (Egoh *et al*. 2008). Therefore, areas in which vegetation cover holds the soil need to be managed to allow continuous delivery of multiple services (de Groot *et al.* 2002). In this study the erosion control services was modeled on the basis of the amount of sediment exported. The modeling of this service was based on the Revised Universal Soil Loss Equation (RUSLE), which is used by N-Spect.

Soil Accumulation

This ES is directly linked to the accumulation of organic matter in the soil (Yuan *et al.* 2006; Egoh *et al.* 2009). Experts in the area have registered a positive correlation among soil depth and vegetation coverage area with the organic matter present in the soil (Yuan *et al.* 2006). Accordingly, these two variables have been used for modeling soil formation (Yuan *et al.* 2006). In this study the soil depth and coverage areas of different habitat types of native forest were used as proxies for soil accumulation. This ES was modeled on the basis of the index of soil accumulation, which was calculated based on the relationship of: i) values of the depth of different soil series present in the study area, which were obtained from Estudio Agrológico de Suelos de Chile (Ciren 2003); and ii) and the coverage area of different habitat types of native forest, which were obtained from the land use map.

These ES were selected due to their importance in the study landscape (Oyarzún *et al.* 2005; 2007; 2011), relevance for the conservation planning (Conaf 2006) and availability of data.

Analysis between HDNF and ES

The relationship between changes in the spatial patterns of HDNF and provision of the ES was analyzed through generalized linear models (GLMs). This analysis was carried out through R statistical software (version 3.0.1) (Venables *et al.* 2013). The function “drop1” was used to assess statistical significance of each of the variables in the models created for a p-value <0.05 using a test distribution $\chi 2$. The variables of less significance were removed in order to find the most parsimonious model.

RESULTS

Model accuracy

The modeling of the ES carried out in the N-Spect software registered a high accuracy. The modeling of the ES water supply registered an efficiency of 0.93, whereas the ES erosion control registered an efficiency of 0.95.

Changes in spatial patterns of the HDNF

The study landscape registered a loss of 12% of area of the HDNF between 1986 and 2011 at a rate of 0.5% per year (Fig. 3.2 and 3.4). The greatest loss (10.3%) occurred between 1986 and 2001 with a rate of 0.73% per year (Fig. 3.2 and 3.4). In 1986, the study landscape was composed of 17,031 patches with a presence of HDNF, equivalent to 37,490 ha (Fig. 3.2 and 3.4). By 2011, the number of patches increased to 26,352, decreasing the total area to 33,084 ha. Twenty-five years later, the HDNF was restricted to small patches, sparsely distributed across the landscape (Fig. 3.2).

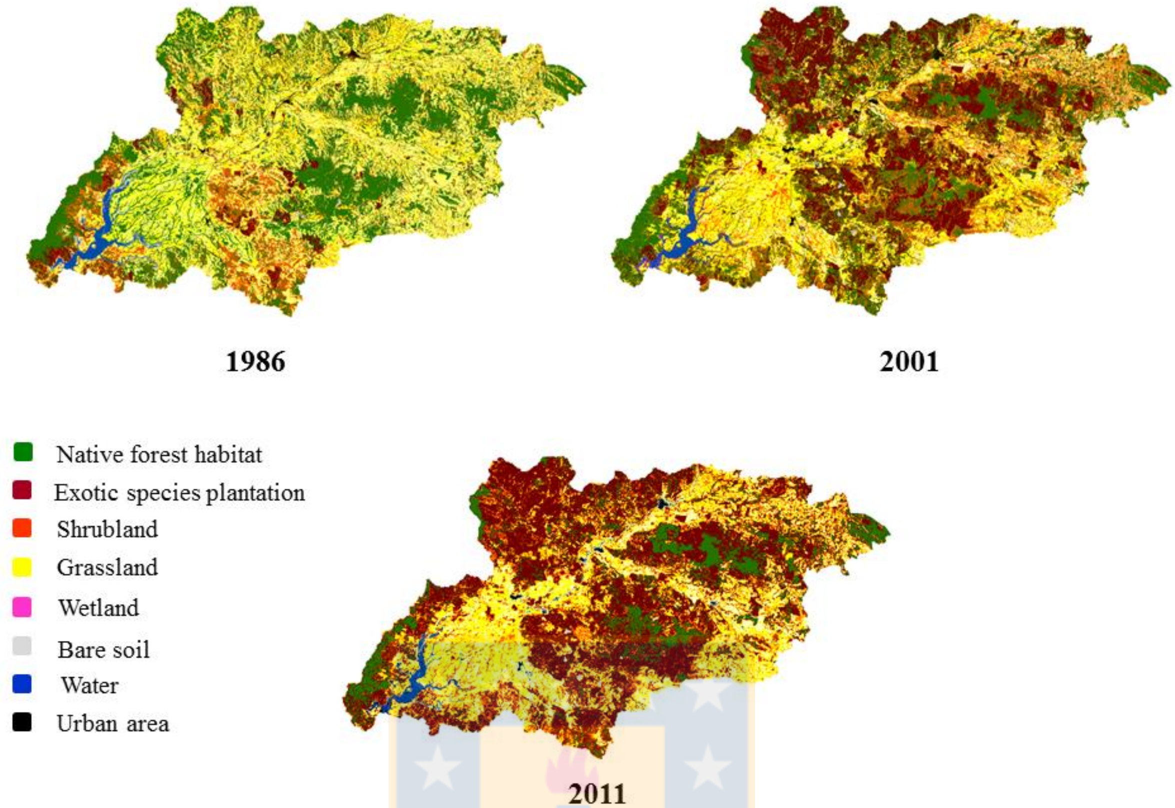


Fig. 3.2. Patterns of land-use change in the Río Cruces watershed between 1986 and 2011.

The landscape studied recorded ten types of native forest habitats (Table 3.1), which registered an average loss of 440 ha of habitat between 1986 and 2011 (Table 3.2). The greatest average loss of habitat (387 ha) occurred between 1986 and 2001 (Table 3.2). An important decrease was registered in the Shannon diversity index (0.20) between 1986 and 2011 (Fig. 3.4a). The greatest decrease in the Shannon diversity index (0.17) occurred between 1986 and 2001. A slight decrease in the index richness of habitat patches (0.02) was registered between 1986 and 2001 (Table 3.2). No change was recorded in the index richness during the second study period.

Table. 3.1. Types of native forest habitat in the Río Cruces watershed. These were determined according to the presence of this habitat in the following variables: vegetation levels, climate and soil.

Habitat type	Variables			
	Native forest	Vegetation levels	Climatic zone	Soil orders
I	VTF ‡	Andean temperate deciduous forest of *Nothofagus alpina* and *Dasyphyllum diacanthoides*	Zone 2, district 0	Andisol
II	VTF	Andean temperate deciduous forest of *Nothofagus alpina* and *Nothofagus dombeyi*	Zone 2, district 0	Andisol
III	VTF	Temperate deciduous forest of *Nothofagus obliqua* and *Laurelia sempervirens*	Zone 1, district 0	Andisol
IV	VTF	Temperate deciduous forest of *Nothofagus obliqua* and *Laurelia sempervirens*	Zone 1, district 0	Ultisol
V	VTF	Temperate deciduous forest of *Nothofagus obliqua* and *Laurelia sempervirens*	Zone 2, district 0	Andisol
VI	VTF	Temperate deciduous forest of *Nothofagus obliqua* and *Laurelia sempervirens*	Zone 2, district 0	Ultisol
VII	VTF	Temperate laurifolio forest of *Nothofagus dombeyi* and *Eucryphia cordifolia*	Zone 1, district 0	Andisol
VIII	VTF	Temperate laurifolio forest of *Nothofagus dombeyi* and *Eucryphia cordifolia*	Zone 1, district 0	Ultisol
IX	VTF	Temperate laurifolio forest of *Nothofagus dombeyi* and *Eucryphia cordifolia*	Zone 2, district 0	Andisol
X	VTF	Temperate laurifolio forest of *Nothofagus dombeyi* and *Eucryphia cordifolia*	Zone 2, district 0	Ultisol

Characteristics of climatic zones and soil orders:

Climatic zone 1, district 0: Total annual precipitation (mm): 1900 min - 2000 max. Annual moisture index 2.0 min. - 2.5 max. Dry period 1 - 2 months/year. Frost-free period 200 - 250 days/year. Total number of frost 10 - 20 days/year. Estival moisture index 0.5 min - 0.6 max. Average relative humidity in January 70% - 80%. Annual absolute temperature min -6 °C, frequency of occurrence 1 month/year.

Climatic zone 2, district 0: Total annual precipitation (mm): 1900 min - 3000 max. Annual moisture index 2.0 min. - 2.5 max. Dry period 1 - 2 months/year. Frost-free period 120 - 200 days/year. Total number of frost 20 - 30 days/year. Estival moisture index 0.5 min - 0.6 max. Average relative humidity in January 65% - 70%. Annual absolute temperature -6 °C, frequency of occurrence 2 month/year.

Andisol order: Soil derived from volcanic ash. These soils in Chile correspond to Trumaos and Ñadis soils. Andisol soils have excellent physical and morphological conditions, whereby can be grown easily. These soils have large amounts of phosphorus but it is retained in the soil in a form that is not available to plants. Therefore, these soils require large amounts of phosphatic fertilizations to obtain high yields.

Ultisol order: Soils with B Horizon well expressed due to an increase of clay in the A horizon. These soils are highly leached. Consequently, it has low levels of nutrients. These soils require large amounts of fertilization to obtain reasonable yields.

‡ Valdivian Temperate Forest

Table 3.2. Area of the different habitat types of native forest and richness of habitat patches in the Río Cruces watershed between 1986 and 2011.

	Area (ha) of the different habitat types										Index richness of habitat patches
Year	I	II	III	IV	V	VI	VII	VIII	IX	X	
1986	2,816	860	794	5,093	8,391	10,453	122	1,783	2,034	5,143	2.55
2001	2,098	842	791	4,844	8,058	9,722	88	1,278	1,384	4,517	2.53
2011	2,013	836	771	4,714	7,990	9,624	83	1,227	1,319	4,507	2.53

Relationship between changes in spatial patterns HDNF and provision of the ES

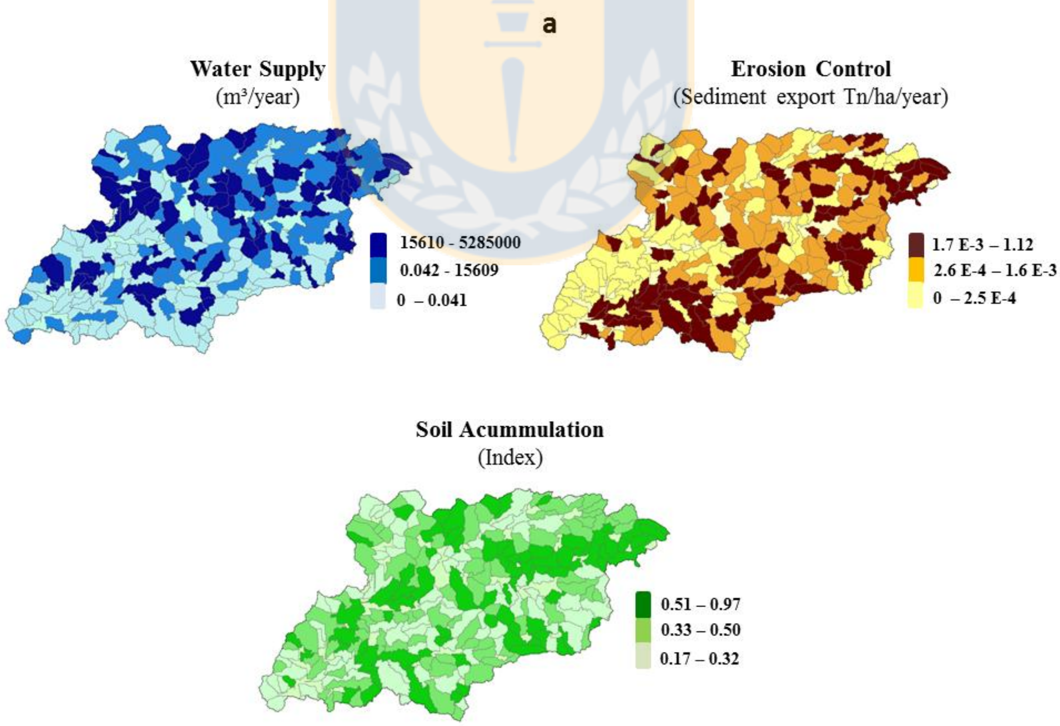
The greatest decrease in the provision of the ES (almost 35%), was registered during the period in which was reported the greatest loss of HDNF area (10.3%), increasing number of patches with presence of HDNF (13,269), and diversity loss (0.7) (Fig. 3.4a and 3.4b). Over the 25 years of this study, the greatest decrease in provision was recorded for erosion control services, which consisted of an increase of 346% of sediment export; and the lowest was registered for water supply, which registered a decrease of 11% (Fig. 3.4a and 3.4b). The soil accumulation services recorded a decrease in provision of 41% (Fig. 3.4a and 3.4b).

A important decrease in the provision of the ES water supply and erosion control was mainly explained by the interaction of HDNF area loss, increasing number of patches, and diversity loss ($p < 0.001$) between 1986 and 2011 (Table 3.3; Fig. 3.3 and 3.4a). The important decrease in the provision of soil accumulation services was explained by the interaction of HDNF area loss and the increasing number of patches ($p < 0.01$) during the twenty-five years of study (Table 3.3; Fig.3.3 and 3.4b). The moderate change registered in the index richness of habitat patches did not explain the decrease in the provision of the ES.

Table 3.3. Generalized linear models (GLMs) built based on the interaction of the changes in the spatial patterns of the HDNF, which explain the loss of provision of the ES in the Río Cruces watershed between 1986 and 2011.

Variables	Estimate	Std. Error	t-value	p-value
Water Supply (Intercept)	8235.102	7335.192	0.874	0.345107
Area loss : Diversity loss	-531.5451	104.578	-4.731	3.75e-05 ***
Increase in patches number : Diversity loss	271.3529	117.278	2.35	0.010645 *
Area loss : Increase in patches number: Diversity loss	-2.1272	0.498	-3.508	0.000456 ***
Erosion Control (Intercept)	-3.12E-03	1.15E-03	-2.890	0.00123***
Area loss : Diversity loss	3.76E-05	1.22E-05	1.282	0.02587 *
Area loss: Increase patches number: Diversity loss	1.79E-07	6.21E-08	1.87	0.02689 *
Soil Accumulation (Intercept)	2.27E-01	4.03E-02	4.745	8.31e-06 ***
Area loss : Increase patches number	3.285E-06	1.01E-06	1.989	0.00478 **

N=262; *p<0.05; **p<0.01; ***p<0.001



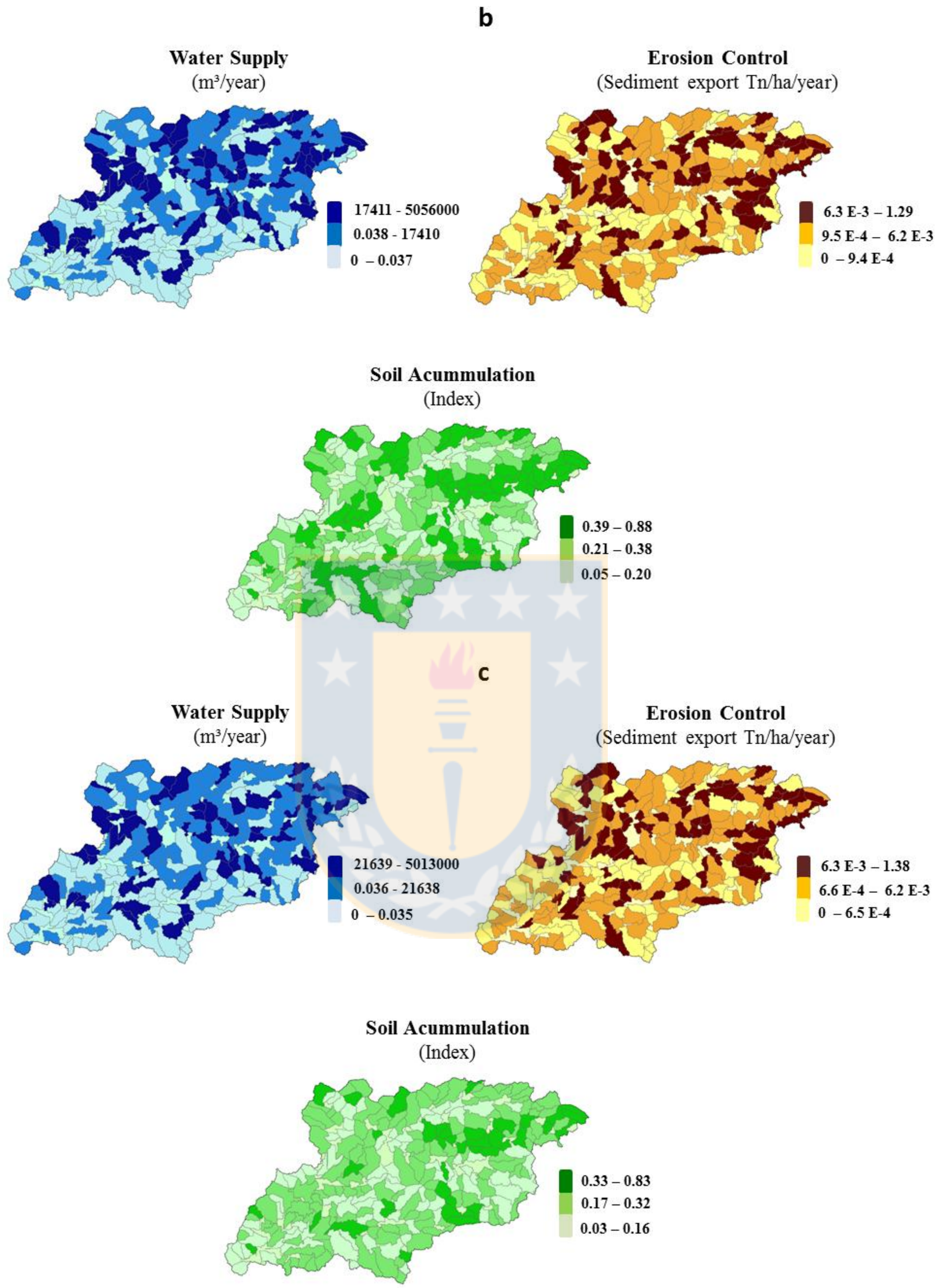


Fig. 3.3. Mapping of the ES water supply, erosion control and soil accumulation in the Río Cruces watershed: a) 1986, b) 2001 and c) 2011.

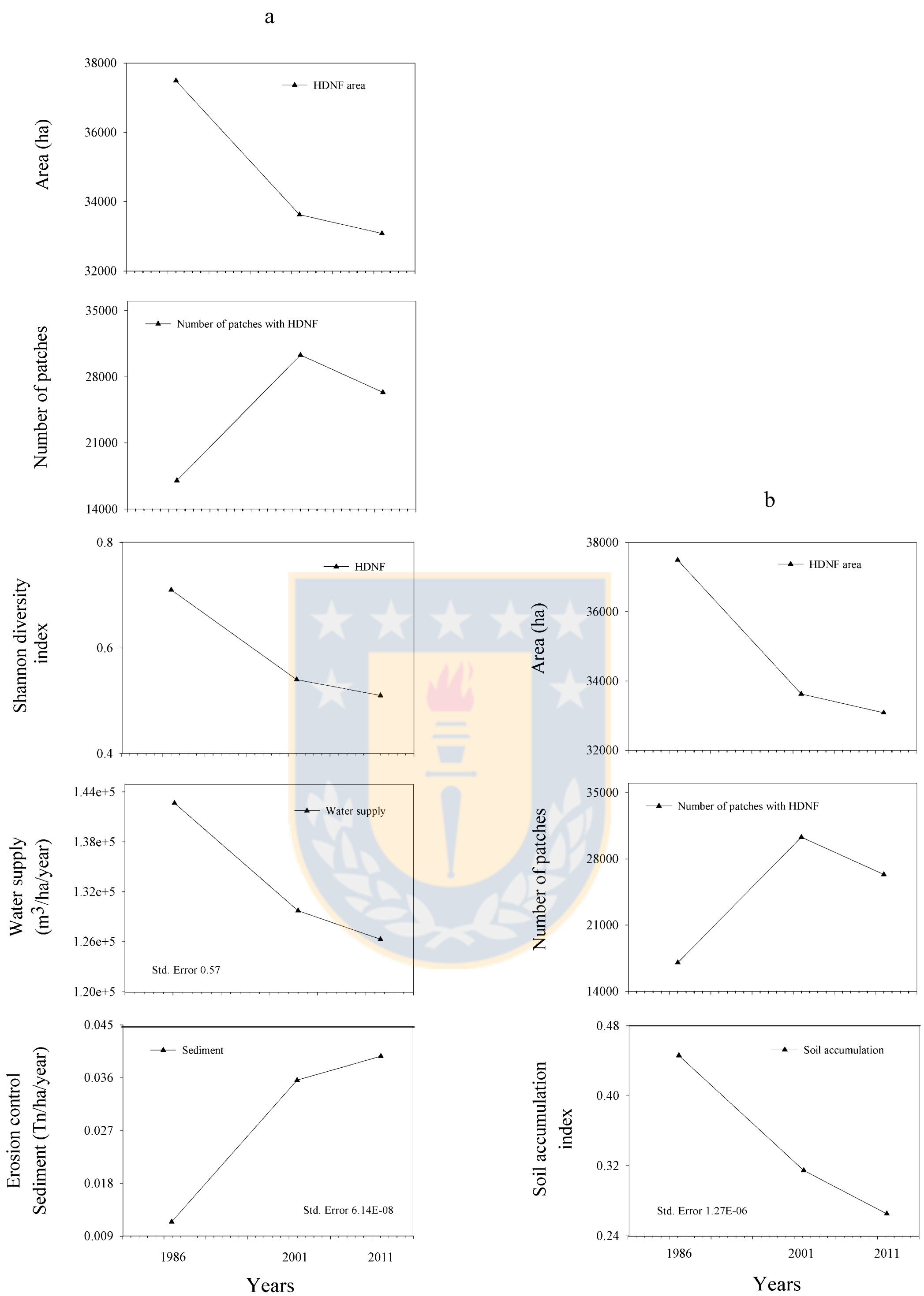


Fig. 3.4. Relationship among loss of HDNF area, increasing number of habitat patches and diversity loss with provision of the ES: a) erosion control and water supply; b) soil accumulation in the Río Cruces watershed between 1986 and 2011.

DISCUSSION

Our study is the most extensive analysis of the relationship between the impacts of LUC on HDNF and their influence on the provision of the ES that has been carried out in Chile. Results demonstrate how the decrease in provision of ES is related to the interaction among loss of HDNF area, increasing numbers of habitat patches, and loss of diversity. The foregoing was determined to be caused by the intensification of land use. N-Spect model was used to map and assess the provision of ES under different land-use maps that recorded the increase of anthropogenic land-use over time. The analysis through N-Spect helps integrate a variety of information from the physical environment and to allow the adaptation of modeling tools to make an innovative landscape-level analysis in the Hotspot of Valdivian Temperate Forest in southern Chile. Our study contributes to the emerging literature that attempts to map and quantify the provision of multiple ES and to know its relationship with biodiversity in a spatial and temporal scale.

Assessment of model accuracy

The model accuracy may be evidenced by comparing the increase in sediment export and decrease in water supply reported in this study, which were 346% and 11% respectively, with the recorded by the DGA during the same study period, which were 364% and 12% respectively. Therefore, our results evidence the advantages of the use of spatially explicit models, supported by remote sensing data, in spatial and temporal assessment of the provision of the ES.

Changes in the HDNF

Our results evidence a substantial loss of HDNF area between 1986 and 2011 in the study landscape. The rate of forest loss reported in this study (0.5%) is lower than that recorded in other hotspots of habitat diversity that have also been severely transformed, such as the Lancang River Valley in the south of China (Liu *et al.* 2014) and Dorset County, on the south coast of England (Hooftman & Bullock 2012), whose rates are 5.5% and 2.05%, respectively. The loss of HDNF area was associated with a severe fragmentation during the study period. This was evident in the increased number of patches with the presence of HDNF, which was impacted by the division of 17,031 fragments into more than 26,300 smaller patches. This division of large and continuous fragments into other smaller and in greater numbers of patches represents one of the main symptoms of fragmentation (Jackson & Fahrig 2013). This trend of an increased number of patches has also been observed in other areas of the world, where the provision of ES has been studied (Baral *et al*. 2014; Xu *et al*. 2014). The severe deforestation and fragmentation evidenced in the HDNF is associated with increased anthropogenic land-use over time, which may also impact the diversity and richness of patches with different habitat types.

Our results register a considerable loss (0.20) in the HDNF between 1986 and 2011. This loss was similar to that recorded in other landscapes that have also undergone considerable transformations, such as the Lamone River watershed in northern Italy (Benini *et al.* 2010), whose loss in HDNF was 0.23 in over a period of 27 years. In contrast, for the Dalinor Nature Reserve, on the plateau of Mongolia - China, a loss of diversity of 0.04 between 1995 and 2008 was recorded (YuhaiBao *et al.* 2011). The diversity loss was associated with a slight loss of richness habitat patch (0.02) during the study period. The foregoing demonstrates that anthropic LUC mainly impacts the abundance of different habitat types and not their variety. Our study showed that LUC generated a severe deforestation, fragmentation, and loss in the HDNF in the last three decades, which could result in alterations in the provision of the ES.

Influence of changes in the HDNF on provision of the ES

Our study evidences that the changes in the spatial patterns of HDNF were strongly associated with a decrease in the provision of the ES in the last three decades. As deforestation, fragmentation, and loss of HDNF increase, the provision of ES declined. A similar relationship is reported in other anthropized landscapes (Zhao *et al.* 2006; Qi *et al.* 2014). The way in which alterations in provision of the ES and changes in spatial patterns of the diversity are related may differ among different regions of the world, due to the specific characteristics of each ecosystem and the different responses of the ecosystem to different anthropogenic interventions (Onaindia *et al.* 2013). In the landscape studied, the loss of provision of each ES was related to specific changes in spatial patterns of HDNF. The loss of provision of soil accumulation services was significantly correlated with the deforestation and fragmentation of the HDNF. The foregoing, probably due to the importance of the abundance of this habitat, independent of their variety, is in the accumulation of organic matter (de Groot *et al.* 2002; Egoh *et al.* 2009). In contrast, the loss of erosion control and water supply services recorded a highly significant relationship with deforestation, fragmentation, and loss of HDNF. This is possibly due to the necessary interaction between the abundance and variety (diversity) of this habitat with different abiotic factors, such as regional climate and topography systems for the provision of services (van Jaarsveld *et al*. 2005; Egoh *et al*. 2009). It is important to consider that a decrease of 8.5% in precipitation was recorded by the DGA during the study period. Therefore, decreasing of water supply services is not only influenced by different impacts of anthropogenic LUC, but also by the variability of precipitation. The results of this study evidence that HDNF has a key role in the ES provision. That is, the variety and abundance of this habitat type play a fundamental role in the amount of provision of the SE studied. In the studied landscape, it is evident that the loss in provision of each ES occurs differently depending on the interaction of different impacts of anthropogenic LUC. Consequently, the different losses in provision of the ES may impact in various ways the wellbeing of people.

Of great relevance, for the future welfare of the people who inhabit the landscape studied, is the reduction of the areas that provide erosion control and soil accumulation services. The importance of these decreases are related to the fundamental role of these services in conservation and soil productivity (de Groot *et al.* 2002), and especially with the regulation of water flow services (de Groot *et al.* 2010), which regulates the water distribution along the surface of the watershed by avoiding runoff and flooding problems (de Groot *et al.* 2002). Therefore, the loss of these services, which are essential to sustaining the agricultural economy in the landscape (DGA 2004), may have serious implications for the well-being of people. Also relevant is the loss of water supply services in the landscape studied. Although these services recorded a lower loss, variations in the water supply can affect economic activities related to the consumptive use and production of market goods, such as drinking water (Oyarzún *et al.* 2005). Therefore, if the trend of loss in provision of ES in the landscape studied continues, over time, the welfare of people may be affected.

Our study maps quantify and analyze the relationships among LUC, HDNF, and provision of ES in a threatened landscape and severely transformed by the increased use of anthropic soil in the last three decades. The results recorded that the substantial loss in provision of ES was due to deforestation, fragmentation, and loss of DHBN, which were the result of anthropogenic LUC.

This study provides fundamental information to optimize planning conservation strategies conservation strategies of the biodiversity and provision of multiple ES in the landscape studied. However, the major challenge is the inclusion of the ES in conservation planning because these are not considered in decision-making due to the fact that this data had not been mapped and quantified. Our results also highlight the urgent need for land-use planning, which should include regulations and incentives for the management of HDNF. Furthermore, it is necessary: i) that the conservation actions and land-use planning consider the current landscape configuration; ii) that they are based on environmental policies; and iii) they require the commitment and active participation of farm owners, the Chilean State, and the general community.

Due to that the parameterization of N-SPECT was carried out with data recorded in the literature, the modeling may be subject to small errors. Therefore, is necessary that future studies working with data taken in the field to obtain greater accuracy in the modeling.

The assessment of multiple ES that delivers HDNF provides key knowledge for the incorporation of the economic value as an information tool in environmental policy decisions. That is to say, the results of our study are the basis for the development of the economic valuation of ES, a pioneering research area that is being developed in Chile. Finally, studies that assess the spatial congruence between HDNF and ES are needed because this information would analyze the consequences of the development of a conservation plan that includes HDNF and ES.

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Capítulo IV

Spatial Congruence between Biodiversity and Ecosystem Services in an Anthropic Landscape in Southern Chile: Basis for Planning Decision-Making

James Rodríguez-Echeverry, Cristian Echeverría, Carlos Oyarzún, Luis Morales

Abstract

There is a need for a broader understanding about how biodiversity and ecosystem services (ES) are spatially related, because a spatial congruence would allow the planning of shared conservation actions, which would contribute in ensure human well-being. In the present study in the Río Cruces watershed, Chile, we assessed the spatial congruence between biodiversity and ecosystem services using spatially explicit models, spatial correlations and overlap analysis. Biodiversity was positively correlated with water supply and soil accumulation. The highest positive spatial correlations between ES were reported for erosion control and water supply, and erosion control and soil accumulation. 68% of biodiversity registered high spatial congruence with: 77% of water supply, 69% and 67% of erosion control and soil accumulation, respectively. The conservation of biodiversity may ensure an important maintenance of the ES. Our study contributes to the development of alternatives for conservation planning and decision-making, which can ensure human well-being.

Keywords Anthropic landscape, Conservation planning, Human well-being, Natural capital, Spatially explicit models.

INTRODUCTION

Biodiversity, broadly defined as the richness and abundance of genes, species and ecosystems (Balmford and Bond 2005), is intrinsically related to ecosystem services (ES), which are the benefits provided by ecosystems that contribute to making human life both possible and worth living (MA 2005). In the last decade, it has become evident the urgent need to conserve these two resources to ensure human well-being (Díaz et al. 2006). In this context, the protection of ES has been used to justify the actions of biodiversity conservation (IUCN 2009). Adopting this perspective may contribute to the optimization of conservation strategies (MA 2005). However, several authors have highlighted the need for a broader understanding of how ES are related to biodiversity (Turner et al. 2007; Onaindia et al. 2013) and to what extent biodiversity conservation ensures the provision of multiple services (MA 2005; Díaz et al. 2006).

One of the great challenges of conservation experts and decision makers is to get a broader understanding about the spatial relationships between biodiversity and ES (Vihervaara et al. 2010; Bai et al. 2011), because the spatial congruence between these may allow simultaneous actions conservation (Turner et al. 2007; de Groot et al. 2010). Currently, these relationships have not been extensively studied (Costanza et al. 2007; Schneiders et al. 2012). Some studies have shown a low correlation and moderate spatial congruence between biodiversity and ES (Chan et al. 2006; Schneiders et al. 2012), others have reported a high spatial congruence between biodiversity conservation and provision of ES (Turner et al. 2007; Egoh et al. 2009). The ambiguity of these findings suggests that there is a need to extend the investigation into new regions and space scale that have not been extensively researched (Egoh et al. 2009), which would provide a more comprehensive understanding of this relationship (Onaindia et al. 2013) and an opportunity for efficient planning decision-making (Turner et al. 2007).

The greatest impacts on biodiversity and provision of ES have occurred at the levels of habitat, ecosystem and landscape (Swift et al. 2004; MA 2005; Díaz et al. 2006), this due to the increase of the human population and its different anthropogenic impacts (Ramankutty et al. 2002; Vihervaara et al. 2010). Accordingly, conservation actions are needed that bring together biodiversity and ES in anthropogenic landscapes (Eigenbrod et al. 2009), which would assure the maintenance of multiple benefits for human populations that inhabit them (MA 2005; Eigenbrod et al. 2009). Therefore, investigating the spatial relationship between habitat diversity and the provision of ES in anthropogenic landscapes will contribute valuable knowledge for optimal and efficient conservation strategies (Egoh et al. 2009).

The forest landscape of southern Chile, which includes Valdivian Temperate Forest, has been identified as a high priority area for biodiversity conservation in the world (Myers et al. 2000). This landscape support the provision of important ES that are the basis for the human well-being, such as: water supply for the consumption by humans, agricultural and aquaculture activities, and fishing (Oyarzún et al. 2005); erosion control and soil accumulation services, which are important for the soil productivity and conservation of the ecosystem integrity (de Groot et al. 2010), so that the provision of food, wood, fiber and medicine may be maintained over time (Díaz et al. 2006). Also, erosion control and soil accumulation services are related with the water flow regulation services, which regulate the water distribution along the surface of the landscape by avoiding runoff and flooding problems (de Groot et al. 2002). The forest landscape of southern Chile has undergone a progressive anthropization in recent decades due to intense and progressive land use change (Echeverría et al. 2006), which has led to important changes in biodiversity at the species level (Bustamante and Grez 1995; Vergara and Simonetti 2004) and the provision of the ES water supply and erosion control in different watersheds (Little et al. 2008; Oyarzún et al. 2011).

The Río Cruces watershed in southern Chile is a landscape that represents a good opportunity to study the spatial relationship between biodiversity and ES, due to the high need of conservation actions that optimize the maintenance of these two resources (Di Marzio and McInnes 2005; Conaf 2006). This landscape has registered a high biodiversity, progressive anthropization (DGA 2004; Conaf 2006) and a high conservation priority in the last decades (Myers et al. 2000). Since the promulgation of Legislative Decree No. 741 of 1974, which encouraged the planting of commercial species in Chile, the Rio Cruces watershed has been an increasing conversion of native forest habitat to commercial plantations (Conaf 2006). In recent years, it has been reported that various impacts derived from the forest industry have led to a significant loss of wildlife (Di Marzio and McInnes 2005; Jaramillo et al. 2007) and a significant increase in the export of sediment in different affluents of the watershed (Oyarzún et al. 2011). The foregoing has been the focus of discussion among governmental environmental entities, forestry companies and the general community, which have highlighted the need to implement effective conservation strategies (Di Marzio and McInnes 2005). Therefore, studying the spatial relationship between biodiversity and SE would provide information very relevant for the understanding of the relationship and for optimal conservation planning, which would contribute in ensure human well-being of the people who inhabit the landscape.

In this study, we assessed the spatial congruence between biodiversity and the following ES: water supply, erosion control and soil accumulation in the Río Cruces watershed in southern Chile. This study presents a systematic methodology that allows the identification of areas where the protection of biodiversity and ES would be the most efficient. The study aims to answer the follow questions: (i) How much of each service is generated in the landscape?, (ii) To what extent does biodiversity correlate with each ES?, (iii) To what extent do synergies exist between different ES? and (iv) To what extent does biodiversity overlap with ES?.

METHODS

Study Area

The Río Cruces watershed is located in the Los Ríos region, in southern Chile (Fig. 4.1). It is located between the cordilleras of the Andes and the coast (39º17´S y 39º50´ S), north of the city of Valdivia. It has an area of 3,640 km^2, a range between 56 and 80 km, and reaches up to 826 m a.s.l. in the coastal cordillera. The mean temperature is 12º C and the annual rainfall is 2,293 mm (Di Castri and Hajek 1976). The watershed is characterized by a warm temperate climate in the north and temperate rain in the southern (DGA 2004). The landscape in the Cordillera is dominated by native evergreen forest, also known as the Valdivian Temperate Forest, and commercial plantations of pine and eucalyptus. In contrast, the plain area is dominated by agricultural and livestock pasture. The watershed has a large human population of about 206 000, which is equivalent to population density of 46 people km^{-2} (DGA 2004). The main economic activities of the watershed correspond to forestry, agriculture and livestock farming (DGA 2004).

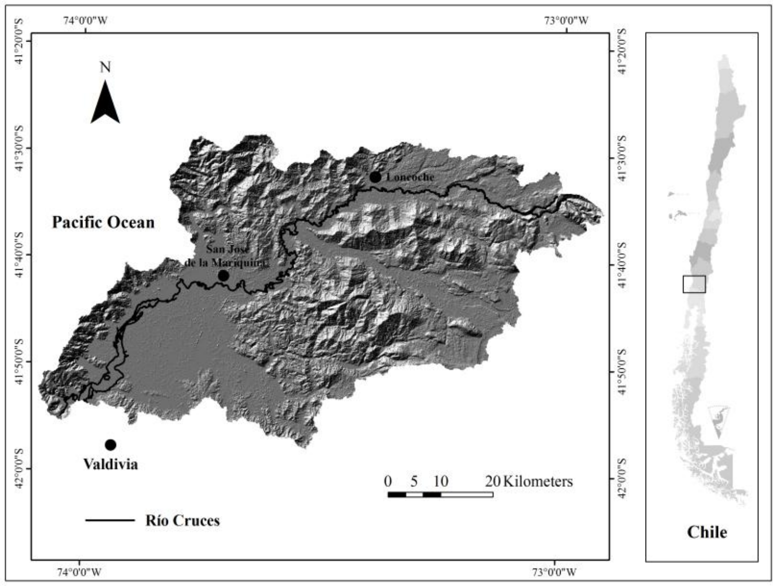

Fig. 4.1 Location of the Río Cruces watershed, Chile

A total of 262 subwatersheds, ranging in size from 500 to 4,000 ha, were analyzed. These were defined as the spatial unit of analysis. The delimitation of the subwatersheds was carried out through the ArcGIS 9.3 Arc Hydro Tools extension (ESRI 2009).

Analysis of Biodiversity

We analyzed the diversity of native forest habitats as a proxy for biodiversity at the landscape level. This was determined by the presence of native forest habitats in different vegetation levels (Luebert and Pliscoff 2004), climatic zones (Schlatter et al. 1995) and soil orders (Ciren 2003) present in the study area. Biodiversity mapping was carried out through the following maps: 1) Map of native forest habitat, which was extracted from land-use map for the year 2011. In this map the following categories of use were identified: native forest habitat, exotic species forest plantations, shrubland, grassland, wetland, and other uses (bare soil, urban area and water bodies). 2) Map of vegetation levels, which was provided by Estudio de Clasificación de Pisos de Vegetación (Luebert and Pliscoff 2004). 3) Map of climatic zones, which was provided by Sistema de Ordenamiento de la Tierra (Schlatter et al. 1995). 4) Map of soil orders, which was provided by Estudio Agrológico de Suelos de Chile (Ciren 2003). Through the overlapping of these maps was obtained different types of habitat. This calculation was carried out through the ArcGIS 9.3 spatial analyst extension (ESRI 2009).

Biodiversity was assessed through Shannon diversity index, which is a landscape metric that relate the variety and abundance of different habitat types in the landscape. This analysis was carried out through FRAGSTATS (version 3.3) (Mcgarigal et al. 2002). Map of biodiversity was entered into FRAGSTATS software to obtain value of the Shannon diversity index for each spatial unit of analysis.

N-Spect Model and Ecosystem Services

The software N-Spect (Non Point Source Pollution and Erosion Comparison Tools) was used to map and analyze the provision of the ES water supply and erosion control. The N-Spect software was developed by The National Oceanic and Atmospheric Administration (NOAA) of the United States, and is used as an extension of Arc GIS 9.3. This software was developed to analyze and predict sediment discharges and the potential impacts on water quality from nonpoint sources of pollution (NOAA 2009). It is software that serves as a tool for management and decision-making concerning water resources, land use planning, agricultural policies and practices. N-Spect is a spatially explicit model that examines the relationship between land cover, nonpoint source pollution, and erosion (NOAA 2009), through a combination of information from the physical environment (elevation, slope, soils, and precipitation) (World Resources Institute 2006). The software generates maps that register the estimates of cumulative runoff and sediment loads (NOAA 2009).

The parameterization of N-SPECT was carried out for 1986 with the following inputs: 1) Digital elevation model (DEMs) 30 x 30 m. 2) Map of land use (píxel 30 x 30 m), in which the following use categories were identified: native forest habitat, exotic species forest plantations, shrubland, grassland, wetland, and other uses (bare soil, urban area and water bodies). 3) Maps of precipitation and rainfall erosivity (R factor). By evaluation of the means of daily rainfall data registered in 12 meteorological stations present in the study area and the method proposed by Angulo-Martínez and Beguería (2009) the average annual rainfall and the coefficient of rainfall erosivity (R factor) were calculated, which were spatialized through geostatistical method of topoclimatological interpolation interpolation (Díaz et al. 2010). 4) Values of vegetation cover (C factor) were estimated on the basis of values proposed by Wischmeier and Smith (1978). 5) Coefficient values soil erodibility (K factor), were calculated from the information of the Estudio Agrológico de Suelos de Chile (Ciren 2003) and using the equation of the nomogram proposed by Wischmeier and Smith (1978). 6) Values for hydrological groups for each soil series, which are related to the number curve method (NOAA 2009), were estimated from soil texture data (Ciren 2003). Information about the different soil series present in the study area was entered into a map, which was provided in the Estudio Agrológico de Suelos de Chile (Ciren 2003). Model validation was carried out for entire study period with the data registered in three pluviometrica and sediment stations of the Dirección General de Aguas (DGA) present in the study area. The goodness of model fit was assessed by the method of quantitative assessment of "relative efficiency (Erel)" proposed by Krause et al. (2005) and Thanapakpawin et al. (2006).

Water Supply

This ecosystem service is the volume of water produced per unit area (m^3/ha) (de Groot et al. 2010) that is potentially viable for human consumption (Chan et al. 2006). The amount and distribution of rainfall is the main determinant of the amount of water produced in a watershed (Egoh et al. 2008). Rainfall patterns, in turn, depend mainly on abiotic factors, such as regional climate and topography systems and not on ecosystems per se (van Jaarsveld et al. 2005). This service was modeled on the basis of rainfall, vegetation cover, soil and topoclimatological variables such as latitude, latitude and proximity to the sea, which determined the annual cumulative runoff.

Erosion Control

This is the ability of natural vegetation to curb erosion by holding onto soil (Egoh et al. 2009), which is measured as the amount (ton/ha) of sediment exported (de Groot et al. 2010). Soil erosion removes nutrients and reduces fertility (de Groot et al. 2010), and may generate sedimentation and eutrophication of nearby rivers (Egoh et al. 2008). Therefore, areas in which vegetation cover holds the soil need to be managed to allow continuous delivery of multiple services (de Groot et al. 2010). In this study the erosion control services was modeled on the basis of the amount of sediment exported. The modeling of this service was based on the Revised Universal Soil Loss Equation (RUSLE), which is used by N-Spect.

Soil Accumulation

This ES is directly linked to the accumulation of organic matter in the soil (Yuan et al. 2006; Egoh et al. 2009). Experts in the area have registered a positive correlation among soil depth and vegetation coverage area with the organic matter present in the soil (Yuan et al. 2006). Accordingly, these two variables have been used for modeling soil formation (Yuan et al. 2006). In this study the soil depth and coverage areas of different habitat types of native forest were used as proxies for soil accumulation. This ES was modeled on the basis of the index of soil accumulation, which was calculated based on the relationship of: i) values of the depth of different soil series present in the study area, which were obtained from Estudio Agrológico de Suelos de Chile (Ciren 2003); and ii) the coverage area of different habitat types of native forest, which were obtained from the land use map.

These ES were selected due to their importance in the study landscape (Oyarzún et al. 2005, 2007, 2011), relevance for the conservation planning (Conaf 2006) and availability of data.

Ecosystem Services Hotspots

The term ES hotspot is used to refer to areas that provide large proportions of a particular service, and do not include measures of threat or endemism (Egoh et al. 2008; Bai et al. 2011; Onaindia et al. 2013). The hotspot mapping for each service was carried out by using the maps obtained in the modeling. In these maps of continuous variables the hotspots were determined using the Jenks Natural Breaks classification in ArcGIS (Reyers et al. 2009; O'Farrell et al. 2010; Onaindia et al. 2013). Natural Breaks classes are based on natural groupings inherent in the data. Class breaks identify the best group of similar values, and they maximize the differences between classes. The data are divided into classes whose boundaries are set, where there are relatively large differences in the data values (O'Farrell et al. 2010; Onaindia et al. 2013). In this way, each map was divided into five equal thresholds, where the highest value was considered an ES hotspot.

Evaluating spatial congruence

Two types of tests (correlation and overlap) were used in our research to evaluate the spatial congruence of biodiversity and ES (Egoh et al. 2008; Reyers et al. 2009; Onaindia et al. 2013).

We calculated correlation (Pearson´s *r*) between the spatial distribution of biodiversity and provision of ES across all 262 subwatersheds present in the landscape. Spatial overlap between biodiversity and ES hotspots was calculated using proportional overlap (Prendergast et al. 1993; Egoh et al. 2008), which expresses the area shared between two services as a percentage of the area of the service with a smaller total area (Egoh et al.,2009). This analysis was performed using ArcGis 9.3 (ESRI 2009).

RESULTS

Model accuracy

The modeling of the ES carried out in the N-Spect software registered a high accuracy. The modeling of the ES water supply registered an efficiency of 0.93, whereas the ES erosion control registered an efficiency of 0.95.

Spatial distribution

Biodiversity and hotspot services recorded important differences in the spatial distribution. Biodiversity registered 85% of their distribution in mountainous areas (Fig. 4.2). The hotspot of water supply and soil accumulation services registered their distribution in the northwest, southeast and central areas of the Cordillera (Fig. 4.2). In contrast, the hotspot of erosion control service is reported in the flat and pre-mountainous areas.

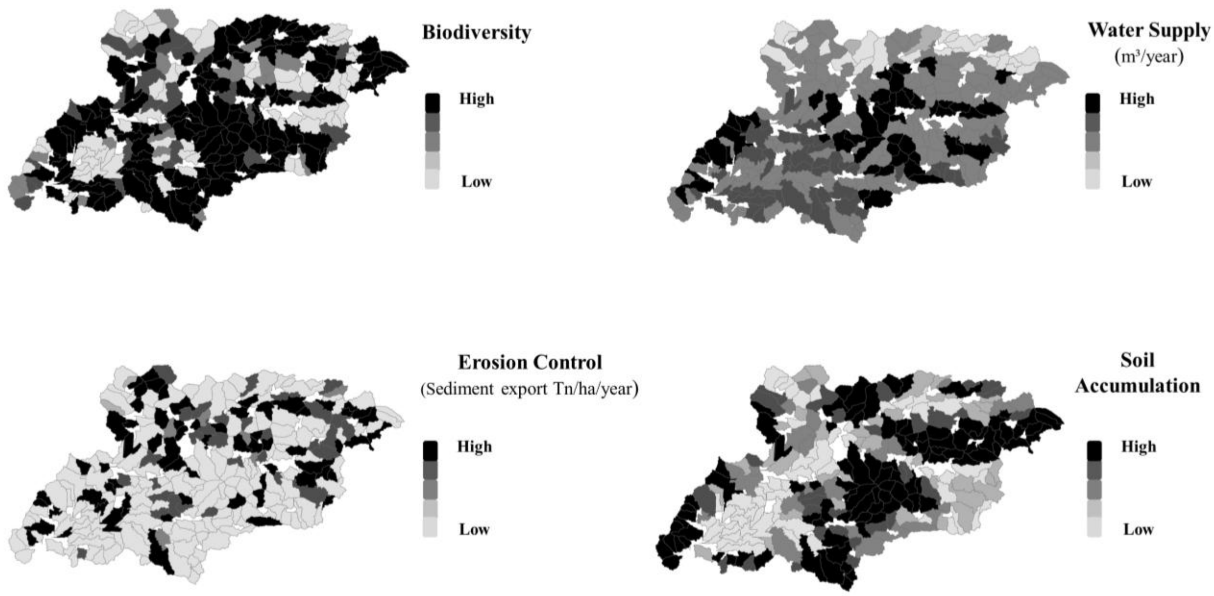

Fig. 4.2 Spatial distribution of biodiversity and hotspot ecosystem services in the Río Cruces watershed

Spatial relationships

The study landscape registered significant correlations between biodiversity and the supply of the ES (Table 4.1). Biodiversity recorded a moderate positive correlation with water supply services (0.43**) and soil accumulation (0.33*) (Table 4.1). Among ES were recorded significant correlations. High positive correlations were reported between erosion control and water supply services (0.41***), and erosion control and soil accumulation (0.15***) (Table 4.1).

Table 4.1 Correlations between biodiversity and ecosystem services in the Río Cruces watershed

	Biodiversity	Water Supply	Erosion Control	Soil Accumulation
Biodiversity	1			
Water Supply	0.43**	1		
Erosion Control	0.09	0.41***	1	
Soil Accumulation	0.33*	0.02	0.15***	1

*p<0.05; **p<0.01; ***p<0.001

The study landscape registered important differences in the size of the ES hotspot (Table 4.2). The hotspot of the erosion control and soil accumulation services reported the largest areas in the landscape, 58% and 36% respectively (Table 4.2). Important spatial overlaps were registered in the landscape studied (Table 2). Biodiversity registered high overlap with hotspots of water supply (77%), erosion control (69%) and soil accumulation (67%) (Table 4.2). Among services, the highest overlap occurred between the hotspot of erosion control and soil accumulation (68%) and erosion control and water supply (59%) (Table 4.2). The study landscape registered areas with a significant number of spatial overlaps between biodiversity an ES (Fig. 4.3).

Table 4.2 Extent and proportional overlap between biodiversity and ecosystem services hotspots in the Río Cruces watershed

	Proportional overlap				Area
	Biodiversity	Water Supply	Erosion Control	Soil Accumulation	(% of study area)
Biodiversity	100				55
Water Supply	77	100			20.3
Erosion Control	69	59	100		58
Soil Accumulation	67	53	68	100	36

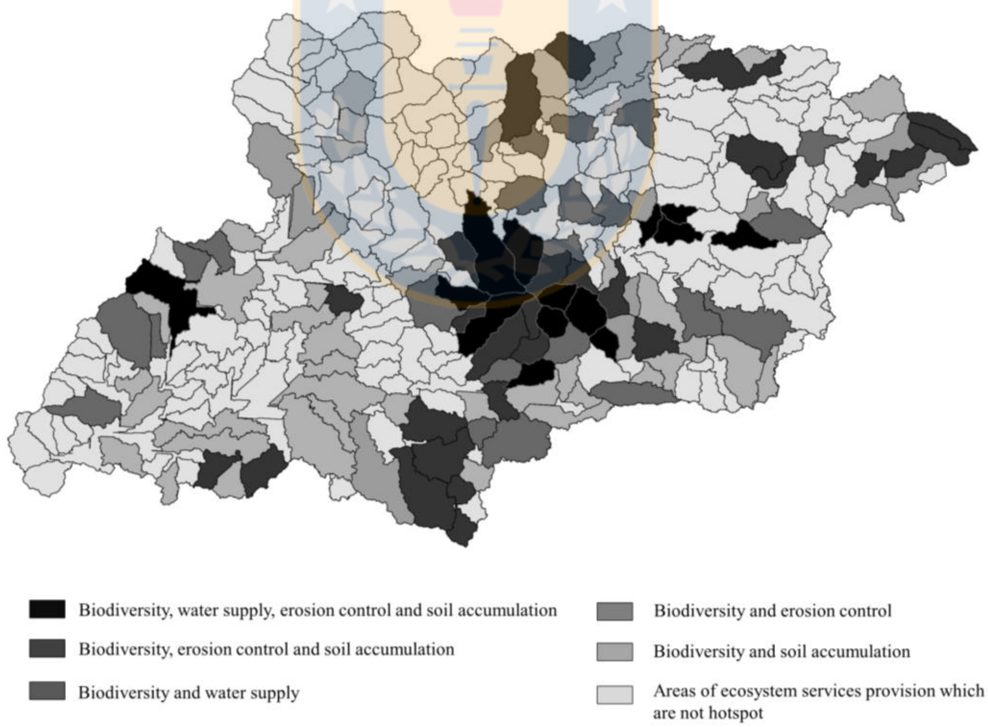

Fig. 4.3 Spatial congruence between biodiversity and ecosystem service hotspot in the Río Cruces watershed

DISCUSSION

The present study constitutes the most extensive analysis of spatial congruence between biodiversity, using as a proxy the diversity of native forest habitat, and the provision of ES ever conducted in Chile. The results demonstrate, at landscape scale, the different spatial relationships and identify the areas in which conservation of the biodiversity and ES would be the most efficient. This research contributes to a broader understanding of the spatial relationship between the ES and biodiversity and to what extent the conservation of this ensures the provision of multiple services.

Our results showed that the spatial distribution of biodiversity is significantly important in the provision of soil accumulation and water supply services. A significant correlation was determined for these two services in this study ($p<0.05$) that is similar to that reported in other threatened landscape (Bai et al. 2011). This is due to the importance of the presence and distribution of biodiversity in the provision of these services (van Jaarsveld et al. 2005; Egoh et al. 2009). The relationship between biodiversity and ES may offer opportunities to permit biodiversity conservation to protect the provision of ES (Turner et al. 2007). In the landscape studied, the provision of soil accumulation and water supply services is significantly related with the biodiversity. Therefore, our study indicates that the conservation of biodiversity would help in the maintenance of these two services.

This study and Bai et al. (2011) recorded high significant positive correlations between ES, which demonstrates that a service may be used for planning the conservation of others (Egoh et al. 2008). This agrees with findings in conservation biology that the protection of biodiversity surrogates contributes to the such conservation (Lombard et al. 2003; Sarkar et al. 2005). Although services do not appear to act as surrogates for other services (Egoh et al. 2008), our study provides the opportunity to use erosion control conservation services to help the maintenance water supply and soil accumulation services.

Our results showed high synergies (almost 70%) between ES hotspots. These are similar to registered in other threatened landscapes, such as in the Little Karoo region, a semiarid biodiversity hotspot in South Africa (Reyers et al. 2009) and the Baiyangdian watershed, China (Bai et al. 2011), whose synergies are < 76% and < 57%, respectively. Synergies between ES differ among landscapes, due to space availability of the services (Turner et al. 2007). In the landscape studied was high availability of the ES hotspot (<58% of the landscape), which influenced a high synergies mainly in the mountainous areas. This study indicates that the protection of these areas would be the most efficient option to conserve the provision of multiple ES.

This study shows high spatial congruence between biodiversity and ES hotspots. Spatial congruences recorded in this study (almost 80%) are higher than those registered in other biodiversity hotspots, such as South Africa (Egoh et al. 2009) and in the Central Coast ecoregion of California, United States (Chan et al. 2006), whose congruences are < 70% and < 57%, respectively. The congruence between biodiversity and ES differs among landscapes according to the spatial characteristics of each ecosystem (Turner et al. 2007). The landscape studied registered an important presence of biodiversity and provision of ES, which, in turn, have very high spatial congruence in different areas of the landscape. Accordingly, conservation of 68% of the area with biodiversity ensures 77% of hotspot erosion control, and 69% and 67% of the water supply and soil accumulation hotspot, respectively. At the same time, the conservation of erosion control ensures 68% and 59% of the provision of soil accumulation and water supply hotspots, respectively. Therefore, the areas that recorded the spatial congruence between biodiversity and ES provide the greatest opportunity to simultaneous actions conservation.

CONCLUSIONS

The study of the spatial congruence between biodiversity and ES contributes in the efficient planning decision-making and conservation strategies, which can ensure human well-being.

This study register important areas that may be ensure the protection of biodiversity and greater provision of multiple ES in the landscape studied. Although the distribution of these areas was relatively large in the landscape, a moderate proportion of them (42%) that did not report congruences are also important for the provision of at least one ES. Successful management of ecosystem services and biodiversity, however, demands a multidisciplinary approach that takes many factors into consideration, and involves all stakeholders (Montagnini and Finney 2011). At present, planning and management of the biodiversity and ES is carried out by different organizations of the Chilean government. Therefore, an integrated approach of the different forms of management is required, which would help to ensure the optimization and efficiency of conservation actions.

The study of biodiversity and ES in landscape scale, using the diversity of native forest habitat as a proxy for biodiversity, contributes to a broader understanding of the spatial relationship between them and the development of new alternatives for planning decision-making.

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Capítulo V
Conclusiones Generales

Este estudio constituye un profundo análisis de la relación poco explorada de: i) los impactos del CUS en la biodiversidad y provisión de SE, y ii) los impactos del CUS en el hábitat una especie amenazada y la composición de sus poblaciones que se ha realizado en Chile. Este estudio entrega evidencia de cómo, dónde y cuándo la pérdida de biodiversidad impacta la provisión de SE; en que la conservación de la biodiversidad asegura la provisión de múltiples SE; identifica las áreas con alto valor para la conservación de la biodiversidad y del bienestar humano; y provee conocimiento clave para las urgentes acciones de conservación que se necesitan para *una especie amenazada.* A continuación, las conclusiones que se presentan siguen el orden de los capítulos presentados anteriormente.

Los resultados registran una progresiva y severa pérdida de bosque nativo con presencia de *F. cupressoides* debido al CUS antrópico. Lo anterior involucró una importante pérdida de hábitat para las cuatro poblaciones de *F. cupressoides* estudiadas. En 2011, el tamaño del hábitat de las cuatro poblaciones estuvo asociado con sus densidades, de tal manera que a menor tamaño hubo una menor densidad. En consecuencia, si la pérdida de hábitat continúa es posible que la densidad de las poblaciones llegue a ser menor, lo que puede aumentar el riesgo de extinción de la especie. De tal modo que urgentes esfuerzos de conservación son requeridos. Estos deben implicar un enfoque de paisaje como estrategia adecuada para detener la fragmentación y pérdida del bosque nativo y mejorar la conectividad del paisaje. Esta estrategia debe apoyarse en un marco de políticas ambientales, las cuales deben ser emitidas por el Estado chileno. Además, a nivel de población es necesario el fortalecimiento de los programas de restauración que han llevado a cabo investigadores de la Universidad Austral de Chile, propietarios de los fundos y la CONAF en los últimos años. La conservación de *F. cupressoides* es un reto que requiere el compromiso y la participación activa del sector privado, el Estado de Chile y la comunidad en general.

Este estudio evidencia que la disminución en la provisión de los SE está relacionada con la pérdida de biodiversidad, la cual fue generada por la intensificación del uso del suelo antrópico. La disminución en provisión de cada SE estuvo asociada a una interacción específica entre deforestación, fragmentación y/o pérdida de diversidad. Lo anterior evidencia la urgente necesidad de la planificación del uso del suelo, la cual contribuya en la conservación de la biodiversidad y de múltiples SE. Para alcanzar lo anterior, ésta debe incluir regulaciones e incentivos, considerar la configuración actual del paisaje, estar sustentada en políticas ambientales y debe implicar el compromiso y participación activa del Estado chileno y la comunidad en general.

Los resultados de este estudio evidencian las ventajas del uso de los modelos espacialmente explícitos, apoyados en datos de teledetección, en la evaluación espacial y temporal de la provisión de los SE. La modelación realizada a través del programa N-Spect permitió integrar una variedad de información del ambiente físico y hacer un innovador análisis a nivel de paisaje en el sur de Chile. De tal modo que este estudio contribuye a la literatura emergente que intenta modelar y evaluar la provisión de múltiples ES.

En el presente estudio, los resultados destacan, a escala de paisaje, las diferentes relaciones espaciales entre la biodiversidad y SE e identifica a las áreas cordilleranas como aquellas con el más alto valor para la conservación de estos. Este estudio contribuye a una mayor comprensión de la relación espacial entre los SE y la biodiversidad, y la medida en que la conservación de uno asegura la provisión del otro, lo cual es información relevante para la toma de decisiones que aseguren el bienestar humano.

Los resultados de este estudio sugieren que las áreas con el más alto valor para la conservación de la biodiversidad y provisión de SE exigen un enfoque interdisciplinario que tome en cuenta los diferentes sectores y actores sociales. En la actualidad, el manejo de la biodiversidad y SE es realizado únicamente por diferentes organizaciones del gobierno, lo que evidencia la necesidad de implementar un nuevo enfoque de manejo, el cual integre las diferentes organizaciones de los distintos sectores sociales con sus respectivos actores. Este nuevo enfoque contribuiría en la optimización y eficiencia de las acciones de conservación.

Finalmente, este estudio contribuye desde la evaluación y comprensión de la relación entre CUS, biodiversidad y SE, con conocimiento importante para el desarrollo de la sustentabilidad del paisaje estudiado. Es decir, provee conocimiento clave sobre la capacidad del paisaje para proporcionar SE esenciales para mantener y mejorar el bienestar humano a pesar de los cambios ambientales y socioculturales.