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"Let's wake up! Let's wake up Humanity! There is no more time. Our consciences will be shaken by the fact that we are only contemplating self-destruction based on capitalist, racist and patriarchal depredation".

—Berta Caceres, Environmental activist

(1976-2016)

To Gisela and Carlos, to my family

and all my family abroad

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Abstract

Human-induced global change has driven drastic modifications of ecosystems that could lead to unexpected and unprecedented transformations in the present and future decades. Current systems modifications increased the need for more comprehensive and evenly distributed databases across ecosystems and spatio-temporal scales. Moreover, base knowledge that allows a better understanding of land-use trajectories and their impacts on the supply of multiple ecosystem services and thus supports the development of highly relevant guidelines for improving landscape management decisions. Hotspots of biodiversity are biogeographical areas identified as biodiversity reservoirs that have been recognized as under threat due to human impacts. These biodiversity reservoirs require further investigation to prevent the deterioration of their ecological functions. Hence, this dissertation aims to understand the impacts and effects of human activities in a biodiversity hotspot area, the Valdivian temperate rainforest by expanding the temporal resolution of land cover data and ecosystem services assessments. The Valdivian temperate rainforest is located in Southern Chile, Northern Chilean Patagonia (73°20’ W-39°25’ S - 71°59’ W-41° 14’ S). The area has been identified as a biodiversity hotspot due to a high number of endemic species (90% at the species level and 34% at the genus level for woody species), and its intense level of human appropriation. This dissertation addresses the following three objectives: 1) Uncovering landscape transformation by expanding the temporal resolution of analyses of landscape dynamics in a biodiversity hotspot area; 2) Understanding the spatio-temporal dynamics of the supply of multiple ecosystems services at the landscape scale; 3) Assessing the contribution of an integrated landscape management strategy to reconnect fragmented ecosystems, on ecosystem services supply and its beneficiaries.

The integration of different types of biophysical and socioeconomic data, as well as methodologies from diverse fields such as remote sensing, ecological modeling, and landscape ecology, were included to answer the main questions of the dissertation. A higher temporal resolution of land-cover dynamics was investigated by using all Landsat scenes available for the study area from 1985 to 2011 (7 periods) and a spatial resolution of 30 meters. An automatic classification with random forest and local ground information allowed to uncover the dynamic of land-cover composition and configuration in the area. Based on this analysis and additional biophysical and socioeconomic data, the trajectory of the ecosystem services supply in the area was revealed at the same temporal scale (7 periods) but with a spatial differentiation between the main four geomorphological units. All these datasets and the methodological procedure of this thesis resulted in the development of landscape planning recommendations that were assessed in the final chapter of this dissertation. The assessed landscape planning strategy builds on the protection of structural connectivity areas (SCA) —defined as the integration of linear (riparian corridors) and patchy (national conservation units) landscape elements— and its contribution to ecosystem services supply as well as its beneficiaries across the landscape.

Results from the land-cover dynamics analysis revealed a highly dynamic and transformed landscape influenced by processes such as clear-cuts of exotic forest plantations, regrowth of secondary forest, afforestation with exotic tree species, together with deforestation and fragmentation of native forest. These modifications impacted both the composition and configuration of the landscape. Areas with exotic forest plantation drastically increase especially from 1985 until 1999 with the highest net increase of 706% from 1985 until 2011. Old-growth forests showed a continuous decrease over time, with the highest deforestation rate of 1.2% - net loss - between 1985 and 1999, this deforestation rate tends to slow down in the last study period (2010-2011). Moreover, the fragmentation of old-growth forest rose especially between 1985-1999 with the decline of patch size and an increase of the total edge length. Secondary forest showed an increase over time but with small and fragmented patches across the landscape. In the case of the ecosystem services supply, the different geomorphological units revealed a diverse pattern with higher regulation services in the Andes and Coastal range in comparison with provisioning services mostly allocated in the Central Valley. The ecosystem services supply trajectory uncovered a decrease of carbon stocks in both mountain ranges as well as an increase in the Central Valley. Regulating services such as sediment and phosphorous retention showed irregular trends which reflected the diverse management strategies used in the area in addition to the low compliance of stream buffer protection, that highlighted the importance of protecting riparian areas. Cultural ecosystem services also declined, for example, the case of aesthetic value that decreased (degree of naturalness) over time and across the study area. In the case of recreational services, even though there is an increase in the service during the study period, these areas are isolated with low or limited access, and a low type of ecosystems represented. Concerning the recommended landscape planning strategy (SCA) assessed in the last chapter, the results reported a positive contribution —maintaining and enhancing— not only to ecosystem services supply but also to the conservation in the area. The assessment revealed the high potential of SCA as a conservation strategy by reconnecting this fragmented landscape and protecting vulnerable areas (riparian corridors); due to the high amount of services that they supply (more than 60%), with an also a higher density of beneficiaries, even when SCA only account for 40% of the total study area.

The results of this dissertation confirm the relevance of integrated research by combining various techniques, disciplines and the consideration of different spatio-temporal scales to achieve better awareness of the functioning of socio-ecological systems by using the ecosystem services concept as a framework. Furthermore, the results highlight the necessity of an expanded temporal resolution in land-cover and ecosystem service assessments to provide more targeted and grounded recommendations for landscape planning.

Zusammenfassung

Der vom Menschen verursachte globale Wandel hat zu drastischen Veränderungen der Ökosysteme geführt, die in den gegenwärtigen und zukünftigen Jahrzehnten zu unerwarteten und beispiellosen Veränderungen führen könnten. Aktuelle Systemänderungen erhöhten den Bedarf an umfassenderen und gleichmäßig verteilten Informationenen (Datenbanken) über verschiedene Ökosysteme und raumzeitliche Skalen hinweg. Darüber hinaus trägt das Basiswissen, das ein besseres Verständnis der Landnutzungstrajektorie und ihrer Auswirkungen auf die Bereitstellung mehrerer Ökosystemdienstleistungen ermöglicht, zur Ableitung hochrelevanter Leitlinien für die Verbesserung von Entscheidungen im Landschaftsplanung. Hotspots der Biodiversität sind biogeografische Gebiete, die als Biodiversitätsreservoirs identifiziert wurden und als durch menschliche Einflüsse gefährdet gelten. Diese Biodiversitätsreservoire bedürfen weiterer Untersuchungen, um eine Verschlechterung ihrer ökologischen Funktionen zu verhindern. Daher zielt diese Dissertation darauf ab, die Auswirkungen menschlicher Aktivitäten in einem Biodiversitäts-Hotspot-Gebiet, dem Valdivianischen gemäßigten Regenwald, zu verstehen, indem die zeitliche Auflösung von Landbedeckungdaten und Ökosystemdienstleistungsbewertungn erhöht wird. Der gemäßigte Regenwald von Valdivian liegt im Süden Chiles, im nördlichen chilenischen Patagonien (73°20' W-39°25' S - 71°59' W-41° 14' S). Das Gebiet wurde aufgrund einer hohen Anzahl endemischer Arten (90% auf Artenebene und 34% auf Gattungsebene für holzige Arten) und seiner intensiven Nutzung durch den Menschen als Hotspot für die Biodiversität identifiziert. Diese Dissertation befasst sich mit den folgenden drei Zielen: 1) Analyse der Landschaftsveränderungen durch Erhöhung der zeitlichen Auflösung von Analysen der Landschaftsdynamik in einem Biodiversitäts-Hotspot-Gebiet; 2) Verständnis der räumlich-zeitlichen Dynamik der Bereitstellung mehrerer Ökosystemdienstleistungen im Landschaftsmaßstab; 3) Bewertung des Beitrags einer integrierten Landschaftsmanagementstrategie, die auf die Wiederanbindung fragmentierter Ökosysteme ausgerichtet ist, zur Versorgung mit Ökosystemdienstleistungen und ihren Nutznießern.

Die Integration verschiedener Arten von biophysikalischen und sozioökonomischen Daten sowie von Methoden aus verschiedenen Bereichen wie Fernerkundung, ökologischer Modellierung und Landschaftsökologie wurden zur Beantwortung der wichtigsten Fragen der Dissertation einbezogen. Eine höhere zeitliche Auflösung der Landbedeckungsdynamik wurde unter Verwendung aller für das Untersuchungsgebiet von 1985 bis 2011 verfügbaren Landsat-Szenen (7 Perioden) und einer räumlichen Auflösung von 30 Metern untersucht. Die automatische Klassifizierung mit zufälligen Wald- und lokalen Bodeninformationen ermöglichte es, die Dynamik der Zusammensetzung und Konfiguration der Landbedeckung in dem Gebiet aufzudecken. Basierend auf dieser Analyse und zusätzlichen biophysikalischen und sozioökonomischen Daten wurde die Trajektorie der Versorgung mit Ökosystemdienstleistungen in dem Gebiet auf der

gleichen zeitlichen Skala (7 Perioden), aber räumlichen differenziert in den vier wichtigsten geomorphologischen Einheiten, aufgezeigt. Alle diese Datensätze und das methodische Vorgehen dieser Arbeit ermöglichten die Entwicklung von landschaftsplanerischen Empfehlungen, die im letzten Kapitel dieser Arbeit bewertet wurden. Die Strategie der bewerteten Landschaftsplanung baut auf dem Schutz von strukturellen Konnektivitätsgebiete (SCA) auf - definiert als die Integration von linearen (Uferkorridore) und patchy (nationale Naturschutzeinheiten) Landschaftselementen - und ihrem Beitrag zur Versorgung mit Ökosystemdienstleistungen sowie ihren Nutznießern in der gesamten Landschaft.

Die Ergebnisse der Analyse der Landbedeckungsdynamik zeigten eine hochdynamische und transformierte Landschaft, die von Prozessen wie dem Kahlschlag von exotischen Waldpflanzungen, dem Nachwachsen von Sekundärwald, der Aufforstung mit exotischen Baumarten sowie der Entwaldung und Fragmentierung von Urwald beeinflusst wurde. Diese Änderungen betrafen sowohl die Zusammensetzung als auch die Konfiguration der Landschaft. Flächen mit exotischen Forstplantagen nehmen vor allem von 1985 bis 1999 drastisch zu, mit dem höchsten Nettozuwachs von 706% von 1985 bis 2011. Alte Wälder zeigten im Laufe der Zeit einen kontinuierlichen Rückgang, mit der höchsten Entwaldungsrate von 1,2% - Nettoverlust - zwischen 1985 und 1999, diese Entwaldungsrate verlangsamt sich in der letzten Untersuchungsperiode (2010-2011). Darüber hinaus stieg die Fragmentierung des Altwaldes vor allem zwischen 1985-1999 mit dem Rückgang der Flächengröße und einer Erhöhung der gesamten Kantenlänge. Der Sekundärwald zeigte im Laufe der Zeit einen Anstieg, jedoch mit kleinen und fragmentierten Stellen in der Landschaft. Im Falle der Versorgung mit Ökosystemdienstleistungen zeigten die verschiedenen geomorphologischen Einheiten ein unterschiedliches Muster mit höheren Regulierungsleistungen im Anden- und Küstenbereich im Vergleich zu den meist im Zentraltal zugewiesenen Versorgungsleistungen. Die Versorgungswege der Ökosystemdienstleistungen enthüllten einen Rückgang der Kohlenstoffbestände in beiden Gebirgszügen sowie einen Anstieg im Zentraltal. Die Regulierung von Dienstleistungen wie Sediment- und Phosphorrückhaltung zeigte unregelmäßige Trends, die die unterschiedlichen Bewirtschaftungsstrategien in dem Gebiet widerspiegelten, ebenso wie die geringe Einhaltung des Schutzes vor Strompuffern, die die Bedeutung des Schutzes von Ufergebieten hervorhoben. Auch die kulturellen Ökosystemleistungen nahmen ab, wie z.B. der Fall des ästhetischen Werts, der abnahm (Grad der Natürlichkeit). Im Falle von Freizeitdienstleistungen sind diese Gebiete trotz einer Zunahme des Angebots während des Untersuchungszeitraums isoliert und haben einen geringen oder begrenzten Zugang sowie eine geringe Art von Ökosystemen. Bezüglich der im letzten Kapitel bewerteten empfohlenen Landschaftsplanungsstrategie (SCA) berichteten die Ergebnisse über einen positiven Beitrag - Erhaltung und Verbesserung - nicht nur zur Versorgung mit Ökosystemdienstleistungen, sondern auch zum Schutz in dem Gebiet. Die Bewertung ergab das hohe Potenzial von SCA als Schutzstrategie durch die Wiederanbindung dieser fragmentierten Landschaft und den Schutz gefährdeter Gebiete (Uferkorridore); aufgrund der hohen Anzahl von Dienstleistungen, die sie erbringen (mehr als 60%), mit einer ebenfalls höheren Dichte an Begünstigten, auch wenn SCA nur 40% des gesamten Untersuchungsgebietes ausmacht.

Die Dissertation bestätigt die Relevanz der integrierten Forschung durch die Kombination verschiedener Techniken, Disziplinen und die Berücksichtigung verschiedener raumzeitlicher Skalen, um ein besseres funktionales Verständnis von sozial-ökologischen Systemen zu erreichen, indem das Konzept der Ökosystemdienstleistungen als Rahmen verwendet wird. Darüber hinaus verdeutlichen die Ergebnisse die Notwendigkeit einer erhöhten zeitlichen Auflösung bei Landbedeckungs- und Ökosystemleistungsbewertungen, um gezieltere und fundierte Empfehlungen für die Landschaftsplanung zu geben.

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1 Introduction

1.1 Motivation and conceptual background

The increasing global demand for natural resources has led to severe modifications in landscapes and ecosystems at different spatio-temporal scales (Butchart et al., 2010; Foley et al., 2005; Meyfroidt and Lambin, 2011). For example, negative impacts on human and environmental health have intensified until unprecedented limits due to factors such as the increase of the global population, changing consumption patterns, water scarcity, and the unsustainable use of resources (Ellis et al., 2013; Foley et al., 2005; Rockström et al., 2009). Scientific evidence of human alterations is large; for example Ellis and Ramankutty et al. (2008), reported in 2008 that more than 75% of ice-free land has some degree of modification and Haberl et al. (2007) showed that approximately 30% of the global terrestrial net primary production shows signs of human appropriation. One of the most important forces that drive land degradation, biodiversity loss, and the ability of the biological system to support human needs is land use/cover human-influenced transformations (Foley et al., 2005; Lambin and Meyfroidt, 2011). Changes in land use/cover have significant consequences, not only at the local but also at the regional, continental and global scale, affecting climate, biogeochemical cycles, and ecosystems in general (Ellis et al., 2010; Klein Goldewijk and Ramankutty, 2004; Lambin et al., 2001). Correspondingly, modifications on the landscapes structure or spatial pattern, respectively, triggers effects that are sometimes difficult to predict (i.e., deforestation, habitat fragmentation, biodiversity loss), due to the diverse range of functions and process that it supports (Hersperger et al., 2012; Lambin and Meyfroidt, 2010; Verburg et al., 2009). Landscapes are complex interconnected systems characterized by their composition and structure at a specific-spatio-temporal scale (Gergel and Turner, 2002; Mcgarigal, 2001; Turner, 1989a). Landscape planning requires robust base information to meet not only society demands but also to maintain the delicate balance of the different ecosystems (de Groot et al., 2010; Goldstein et al., 2012). For that reason, integrated approaches that combine biophysical and socioeconomic information are highly required, as is the case of the ecosystem services approach, that addresses the interconnection between ecosystem services supply and societal demands. Ecosystem services are defined as the benefits that humans obtain from ecosystems (Millennium Ecosystem Assessment, 2005). Furthermore, the approach seeks for a better understanding of the components and functions of socioecological systems, by separating the multiples services and benefits from ecosystems as well as and linkages between them (Folke, 2006; Olsson et al., 2006; Rockström et al., 2009). Although the concept is not new (Wilson and Matthews 1970, Ehrlich and Mooney 1983), the frame rapidly expanded since the release of the Millennium Ecosystem Assessment (Millennium Ecosystem Assessment, 2005). Ecosystem services supply is divided into three major groups: provisioning (i.e., industrial forest plantations for pulp production), regulating (i.e., carbon storage), and cultural (i.e., forest recreation) and the demand or beneficiaries of these services (Haines-Young and Potschin, 2018). Considerations of the spatial and temporal scale are also crucial since landscape processes, and functions are scale dependent (Grêt-Regamey et al., 2014; Müller et al., 2010; Raudsepp-Hearne and Peterson, 2016). Likewise, other global initiatives (i.e., TEEB, IPCC, IPBES, Natural Capital) helped to put on value the importance of ecosystem services for the human well-being (de Groot et al., 2010; Roy Haines-Young and Potschin, 2013; Turner and Daily, 2008). At the global level governments have agreed to implement the concept of ecosystem services in their national assessments. They get increasingly integrated into their planning and decision-making process (Lele et al., 2013; Wong et al., 2015). For example, Joppa et al., 2016 pointed out that U.S. federal agencies were required to integrate ecosystem services into their planning and decision-making by "conducting decision-relevant and scale-specific ecosystem-services assessments, as well as plans for effective monitoring and evaluation."

Measuring these land cover and ecosystem services modifications requires careful consideration of the spatial and temporal scale at which the landscape process or function occur (N. M. Haddad et al., 2015; Turner, 1989b). Regional land use/cover change processes are often monitored at relatively coarse temporal resolutions (snapshots from 5 to 15 years), reducing considerably the ability to identify and understand regional and local impacts across the landscape (Lunetta et al., 2006; Pflugmacher et al., 2012a; O E Sala et al., 2000). Moreover, it also decreases the potential to link land use change to policies, and market-oriented decision as well as the availability to react to them (Barnett et al., 2016; Hein et al., 2006; Rodríguez et al., 2006). The increasing availability of high spatio-temporal earth observation products, together with the improvement of big data management technologies provide a considerable potential to expand the resolution of information related to land use/cover change research and thus to improve the understanding of related patterns and process at different scales (J Hansen et al., 2000; Zhu and Woodcock, 2014). At the regional scale, the free-release of the Landsat archive opened the potential to continuous landscape dynamics observations (Wulder et al., 2012). The integration of earth observation, ground data with time-series image processing and change detection techniques offer considerable improvements especially in areas with low data accessibility (i.e., cloud, poor quality images), high levels of biodiversity, and rapid land use change processes. Likewise, much more than solely land cover mapping is needed to be able to understand landscape dynamics and manage socio-ecological systems. Considering that human societies highly depend on natural and managed systems to prosper (Pecl et al. 2017), a more inclusive approach is required that accounts for the natural/managed systems and its beneficiaries across the landscape (Balvanera et al., 2015; Bennett et al., 2009; Ellis and Ramankutty, 2008). The previously discussed ecosystem service concept offers this integrated approach by including biophysical or ecological aspects of landscapes (supply) together with the society demands (demand) (Bennett et al., 2015; de Groot et al., 2012; Díaz et al., 2006; Mach et al., 2015). Despite its relevance of including the ecosystem services demand (i.e., beneficiaries of the services) there is a strong bias towards supply ecosystem services assessments (de Groot et al., 2010; Reyers et al., 2013; Seppelt et al., 2011). Bagstad et al., 2014 argue that the conflict mainly remains in difficulty to identify and map the beneficiaries of the different ecosystem services, understanding that often ecosystem services and its beneficiaries are not co-located (spatial mismatch). In many regions, ecosystem services assessments that integrates multi-scale mapping and monitories are still missing or are underrepresented as in the case of Latin America and Southern Chile (Balvanera et al., 2012; Malinga et al., 2015).

This also applies to the unique relict of Temperate Rainforest in South America, the Valdivian Rainforest (73°20’ W-39°25’ S - 71°59’ W-41° 14’ S). The Valdivian Rainforest has been identified as a biodiversity hotspot due it is high level of endemism, with 90% at the species level and 34% at the genes level for woody species (J. Armesto et al., 1998; Arroyo et al., 1996). Despite its ecological relevance the region has experienced rapid deforestation and fragmentations processes (Echeverria et al., 2006c; Locher-Krause et al., 2017b) due to historical and actual economic demands (Armesto et al., 2010; Niklitschek, 2007). Reported land use change in the area provided information regarding deforestation and fragmentation trends but lacks the temporal resolution to understand land use modifications as a dynamic process interlinked to the regional and global demands (Altamirano et al., 2013; Echeverria et al., 2006c; Miranda et al., 2017; Laura Nahuelhual et al., 2013a; Zamorano-Elgueta et al., 2015). In this region due to the high climatic variability — with high precipitation amounts and cloud cover— an automatic multitemporal land cover analysis offers improvements of spectral variation, as well as enhancement of cloud, obscured images (Canty and Nielsen, 2008; Gao and Masek, 2008). The extended temporal information —by using all the available Landsat scenes for the area— allows not only a better understanding of landscape dynamics but also the mapping and modeling of ecosystem services to close the knowledge gap regarding human modifications and state of ecosystems in the area. This is particularly relevant such hotspot biodiversity areas as the Valdivian temperate forest, in which dynamic land use/cover information together with ecosystem services assessments can support landscape planning.

Moreover, management strategies that highlight the benefits of the ecosystem services approach for landscape planning can contribute largely to maintain the flow of ecosystem services in our study area, such as the case of planning strategies oriented to reconnect fragmented landscapes. Pringle, 2017 suggested that landscape strategies that look for expanding and restoring current landscape elements by reconnecting existing protected areas and corridors improve conservation efforts. Research exercises that integrate such management strategies provide valuable information to provide the knowledge needed for the implementation of desirable socioecological future scenarios.

1.2 Knowledge gaps and main research objectives

Land use/cover change is one of the most important drivers of environmental and ecosystem transformations. Under the current global scenario, with unexpected and unprecedented system modifications due to climate change and anthropogenic demand, information on land use/cover and ecosystem services dynamics is highly required (Bennett and Chaplin-Kramer, 2016; Foley et al., 2011; N. M. Haddad et al., 2015). However, while current global initiatives enhance the relevance of scale-oriented research to improve sustainable landscape planning, integrative assessments that include the three main pillars of sustainability (ecological, economic and social) are limited (Burkhard et al., 2014; Clec'h et al., 2016; Costanza et al., 2017; Hicks et al., 2016). Nowadays the increasing free access to datasets, models, and tools represent a considerable potential to understand processes and functions at the different temporal and spatial scales (Cord et al., 2017b; Hansen and Loveland, 2012; Ju and Roy, 2008; Roy et al., 2014; Wulder et al., 2012). At the regional level, these tools and datasets are crucial for providing comprehensive and consistent base information to policymakers and the society in general. This dissertation aims to understand the impacts and effects of human activities on selected ecosystem services in a biodiversity hotspot area, by expanding the temporal resolution of land cover data (composition and configuration) under the umbrella of the ecosystem services approach (supply and beneficiaries assessment). The spatially-explicit approach integrates not only biophysical but also socioeconomic variables providing concrete landscape planning strategies (i.e., connectivity areas), that focus on enhancing and maintaining the benefits and flow of ecosystem services across the landscape. Specifically, this dissertation addresses three main research objectives and its knowledge gaps:

Objective 1. Expand the temporal resolution of landscape transformations in a biodiversity hotspot area. The Valdivian temperate rainforest is located in an area in which previous studies have reported drastic land cover transformations linked to political and economic governmental decisions (Armesto et al., 2010; Echeverria et al., 2006c; Niklitschek, 2007). These decisions have led to changes in composition and configuration of the forest ecosystems (Altamirano and Lara, 2010; Miranda et al., 2017; Zamorano-Elgueta et al., 2015) impacting in a not yet fully understood way the socioecological systems of the area. Despite the relevance of these studies, landscape transformations are reported at a low temporal resolution (described only in two or three snapshots) missing crucial information regarding the periodicity, continuity, and magnitude of these changes over time and across the landscape. To close this gap, an approach that integrates high spatial-temporal density dataset is applied by:

- Revealing patterns of land cover change processes at a high temporal scale based on all suitable Landsat scenes available for the study area in the south of Chile from 1985 to 2011, with a particular focus on forest cover changes.
- Analyzing changes in land cover composition and configuration across a geomorphological gradient –from the Coastal mountain range, over the Central Valley to the Andes mountain range– in southern Chile.

Objective 2. Spatio-temporal assessment to understand the dynamics of multiple ecosystems services supply across the landscape.

Multi-scale ecosystem services mapping and monitoring are still far from being regularly integrated into landscape planning, and this gap is even more significant in Latin America (Balvanera et al., 2012). Regardless of the ecological importance of the area, not only at the national but the global level, there is fragmented information regarding ecosystem services. Spatio-temporal ecosystem services assessments in this area could contribute to understanding functions, process and its benefits to the society. In Central-South Chile, ecosystems services assessments are few and rely only on the comparison of two or three periods that could lead to a misperception about the magnitude of change (Lara et al., 2009; Little et al., 2014, 2009; L. Nahuelhual et al., 2013; Núñez et al., 2006). To fill the knowledge gap on the dynamics of ecosystem services supply in regions under threat such as the study area, different data and techniques are needed. The analysis builds on the integration of land cover information derived from remote sensing data with spatially explicit models, official statistics and field measurements, addressing the following questions:

- How were the supplies of selected individual ecosystem services distributed across the landscape and over time?
- How did the spatial distribution of selected individual ecosystem services supply change over time?
- What are the implications for the ecosystem and landscape management that aim to balance natural resources utilization and conservation in the region.

Objective 3. Ecosystem services beneficiary's assessment integrating landscape management strategies to reconnect fragmented ecosystem.

An integrated ecosystem services assessment should include not only the biophysical aspect of services but also its beneficiaries to make explicit the interconnection among them (Bennett et al., 2015; Liu et al., 2010). Therefore, the third objective of this dissertation integrates the beneficiaries of the selected forest ecosystem services. Furthermore, this objective also includes the evaluation of one specific landscape management strategy oriented to reconnect this fragmented landscape to make visible the impact of the ecosystem services approach on landscape planning. The assessed landscape planning strategy builds on the protection of structural connectivity areas (from now on SCA), quantifying its effects and impacts on ecosystem services supply and its beneficiaries, to address the following questions:

- How much does SCA contribute to forest ecosystem services supply over time?
- Where are the highest density of ecosystem services beneficiaries located across the landscape and which is their linkage with SCA?
- How much the SCA, consider as landscape planning strategy under different protection status, could contribute to maintaining the balance between production and conservation

1.3 Overview of dissertation structure

This dissertation comprises six chapters. Chapters 1 and 2 provide the scientific and methodological frame applied in this accumulative dissertation. Chapters 3, 4 and 5 cover the published papers or manuscripts submitted to ISI-listed journals.

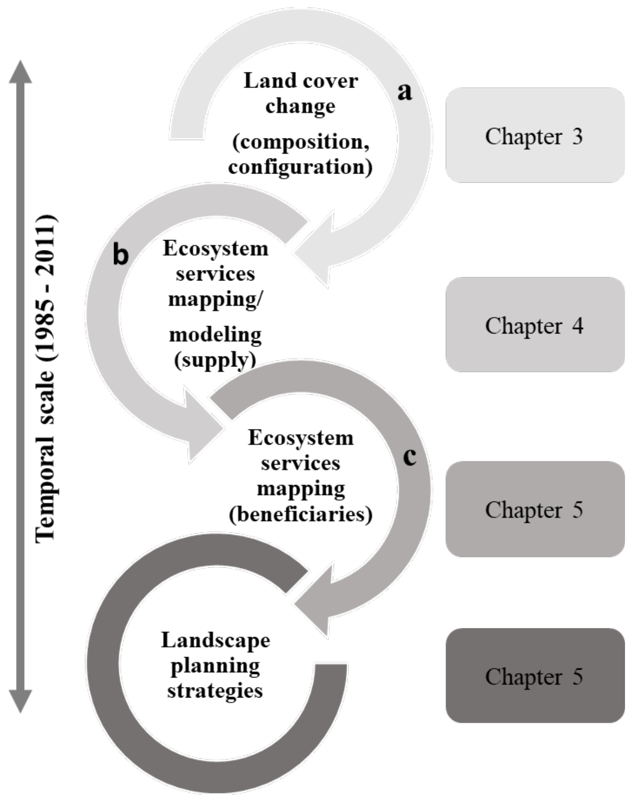

Figure 1.1 Overview of the dissertation structure, showing the chapters that comprise the related journal papers. In the left side of the figure, the temporal resolution of the overall research is included (from 1985 until 2011), the regional spatial resolution is constant within the different chapters (30 meters). The letters a, b, c represents the major research topics developed through the different assessments, described in detail in Figure 2.1. The different grey tones illustrate the interconnection among the chapters and the research topic addressed in this dissertation.

Chapter 3 addresses **objective 1** of this research by using remote sensing and ground data to expand the temporal resolution of land cover change analysis, describing the changes in landscape composition and configuration. Chapter 4 addresses **objective 2.** This study uses dynamic landscape data plus biophysical and socioeconomic data to model multiple ecosystem services supply in the area at different temporal scales. Chapter 5 addresses **objective 3,** by assessing the beneficiaries of the services identified and modeled in chapter 4. In this chapter, the relevance of potential landscape planning strategies oriented to reconnect this fragmented landscape is also assessed.

Finally, chapter 6 discusses the scientific significance of the results of each chapter. Furthermore, conclusions and recommendation for future work are also provided in this chapter.

2 Methods and Data

2.1 Study area

The study was focused on a biodiversity hotspot area located in Southern Chile, Northern Chilean Patagonia (73°20' W-39°25' S - 71°59' W-41° 14' S). The area is characterized by the Valdivian temperate rainforest, one of the largest relicts of temperate rainforest in the world with a high amount of endemic and threatened species (Armesto et al., 1996; Myers et al., 2000; Olson et al., 2001). Geomorphologically the area is divided into three units, the Coastal Mountain range (up to 900 m), (2) the Central Valley (up to 250 m), and Precordillera and the Andes mountain range (up to 2,422m).

The area is home to circa 381.720 inhabitants, distributed mainly in the Central (INE Instituto Nacional de Estadística, 2012). From these inhabitants 68.3% are urban, and 31.3% rural residents; 17% describe themselves as Mapuches, a group of Chilean indigenous inhabitants. Principal economic activities are closely related to natural resources utilization, particularly in the forest industry and the agricultural sector. The forest industry mainly focuses on exotic plantations (e.g., pulp, shipyard, and paper industry) as well as on wood processing factories for wood and paper products. This strong economic focus oriented to natural resources together with historical unsustainable management practices have triggered environmental conflicts in the area (i.e., soil erosion due to poor management standards, river pollution, loss of biodiversity) (Armesto et al., 2010; Echeverria et al., 2006c; Lara et al., 2009).

2.2 Overview of the data and methods used in the dissertation

Studies that aim at a comprehensive and integrative landscape dynamics assessment are requested to evaluate not only composition and configuration elements but also to address function and processes at the suitable spatio-temporal scale (Cord et al., 2017a; Holland et al., 2011; Scholes et al., 2013). In this dissertation, different types of data and techniques are used to evaluate historical landscape dynamics, emphasizing on the use of the ecosystem services framework as a tool for sustainable landscape planning (Balvanera et al., 2015; Bennett et al., 2015; Spake et al., 2017). The approach followed in this dissertation is characterized by methodological steps as is shown in Figure 2.1. The first step (a) is used in chapter 3 and provided a continuous land cover trajectory (composition and configuration) by integrating earth observations (Landsat) and remote sensing techniques (Figure 2.2). The second (b) and third (c) steps include the mapping and modeling of ecosystem services (Figure 2.3 and Figure 2.4). The mapping and modeling of multiple ecosystem services (supply and its beneficiaries) built on the data generated in the first step (a) plus biophysical and socio-economic regional data. In the last stage, data from the previously described analyses (a, b, c) were integrated to exemplify a suggested landscape planning initiative that aims to reconnect this fragmented landscape, balancing conservation, ecosystem services supply and beneficiaries needs. A more detailed description of the data and methods is provided in the following sections.

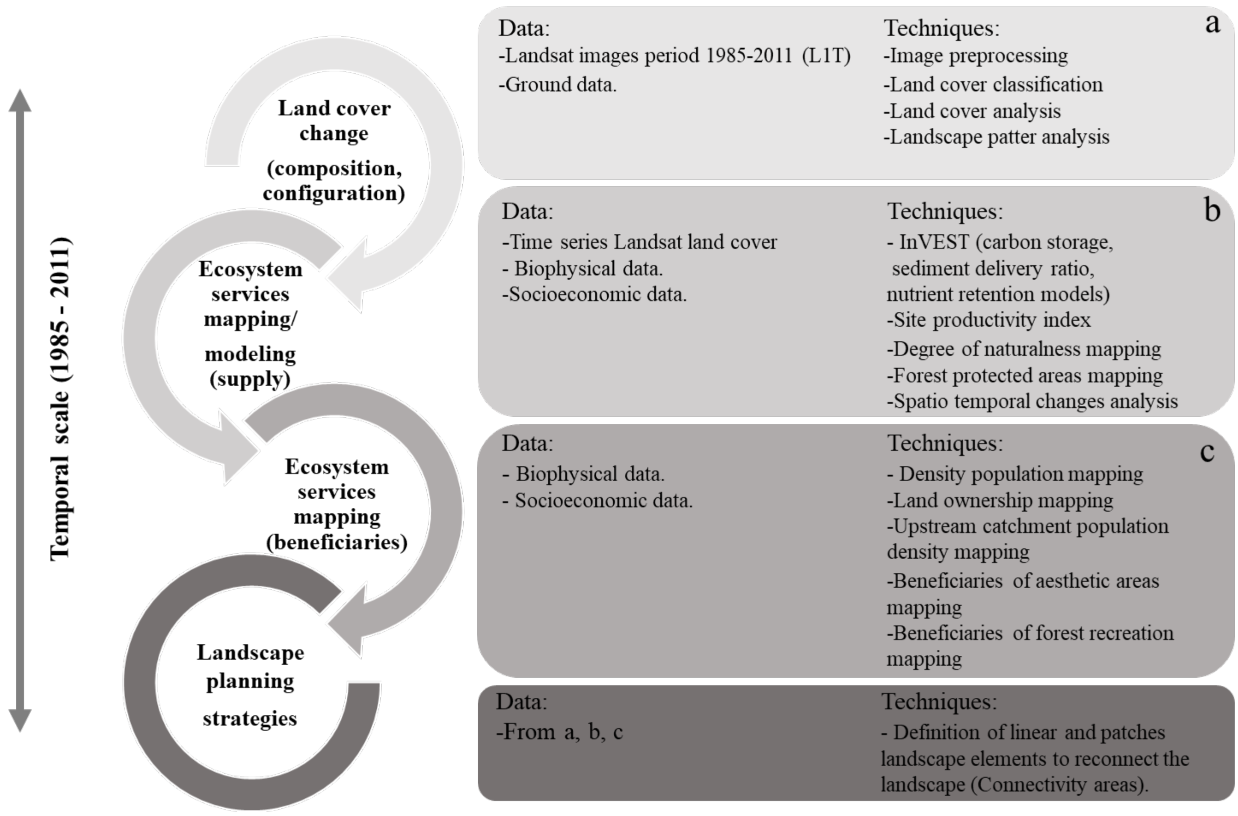

Figure 2.1 Overview of the data and methods used in the dissertation. The landscape dynamics assessment is based on the temporally continuous land cover analysis and the ecosystem services assessment. The ecosystem services mapping and modeling (supply and its beneficiaries) was performed at a regional scale and set the bases for regional landscape planning.

2.2.1 Spatio-temporal land cover transformations

The temporally continuous land cover analysis was based on earth observations and ground data as is shown in Figure 2.2. To reduce and eliminate atmospheric, cloud and radiometric distortions corrections to all selected images were done by using different protocols (LEDAPS, Fmask, and IR-MAD) (Canty and Nielsen, 2008; J. Masek et al., 2006; Zhu and Woodcock, 2012). These protocols are explained in detail in chapter 3. The land cover classification was done using a non-parametric machine-learning classifier to automatize the classification process. The random forest algorithm in R offers a robust alternative to traditional imagine classification methods (Gislason et al., 2006; Pal, 2005; Waske et al., 2012). Post-processing was performed to ensure the quality and to provide error and uncertainties of the classification process.

Based on the continuous land cover dataset, landscape configuration indexes were calculated using the software FRAGSTAT v.4 (Mcgarigal, 2014).

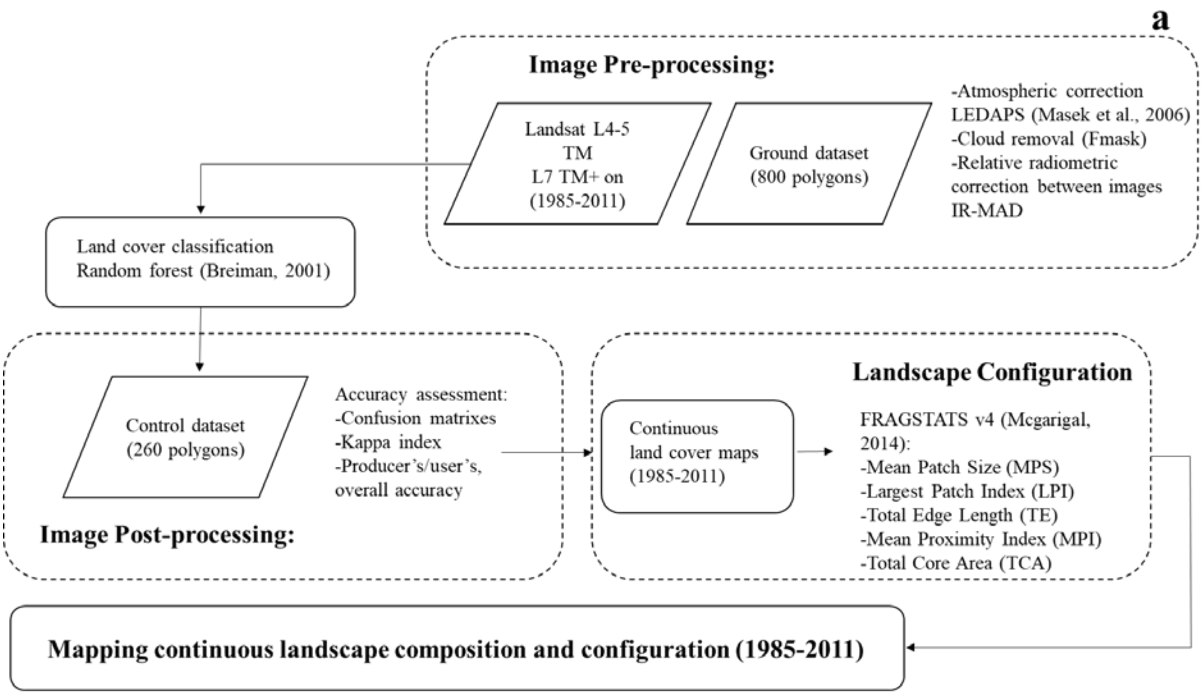

Figure 2.2 Overview of the data and methods used in the first stage (a) of the dissertation. This step (a) integrate remote sensing and data analysis techniques to extend the temporal resolution of landscape transformation in the area.

2.2.2 Ecosystem services assessments (supply and its beneficiaries)

The mapping and quantification of ecosystem services were divided into two parts, one that describes the modeling and analysis of the ecosystem services supply and the second that deals with its beneficiaries (Figure 2.3 and Figure 2.4). Six forest ecosystem services were selected based on a literature review, information obtained from the first stage of this research and also from communication with regional government. The selected ecosystem services were: forest plantation production (provisioning), carbon storage (regulating), nitrogen retention (regulating), phosphorous retention (regulating), aesthetic value (cultural), and forest recreation (cultural). The supply of the six forest ecosystem services selected was quantified at a spatial scale of 30 meters, and for the case of ecosystem, services supply at the same temporal resolution of the land cover analysis. Due to the lack of data the ecosystem services beneficiaries were mapped only for the final period (2010-2011). Also, in the case of beneficiaries, the services nitrogen and phosphorous retention were merged as water regulating services to identify drinking water beneficiaries.

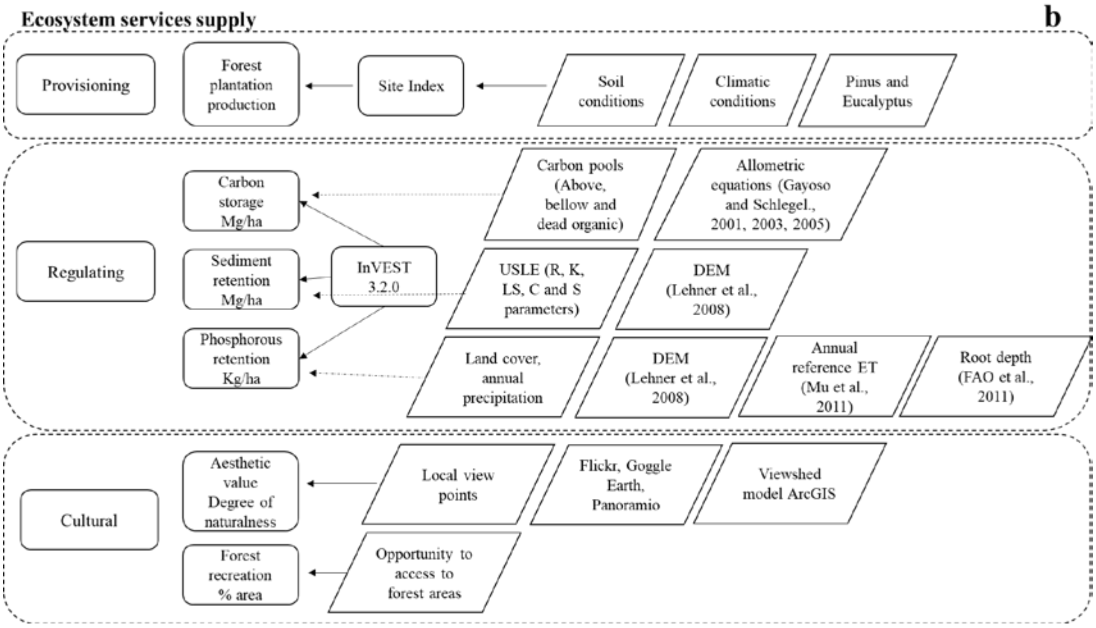

Figure 2.3 Overview of the data and methods used in the second step (b) of this dissertation. This step (b) integrate remote sensing and modeling techniques to quantify and estimate the trajectory of ecosystem services supply in the area.

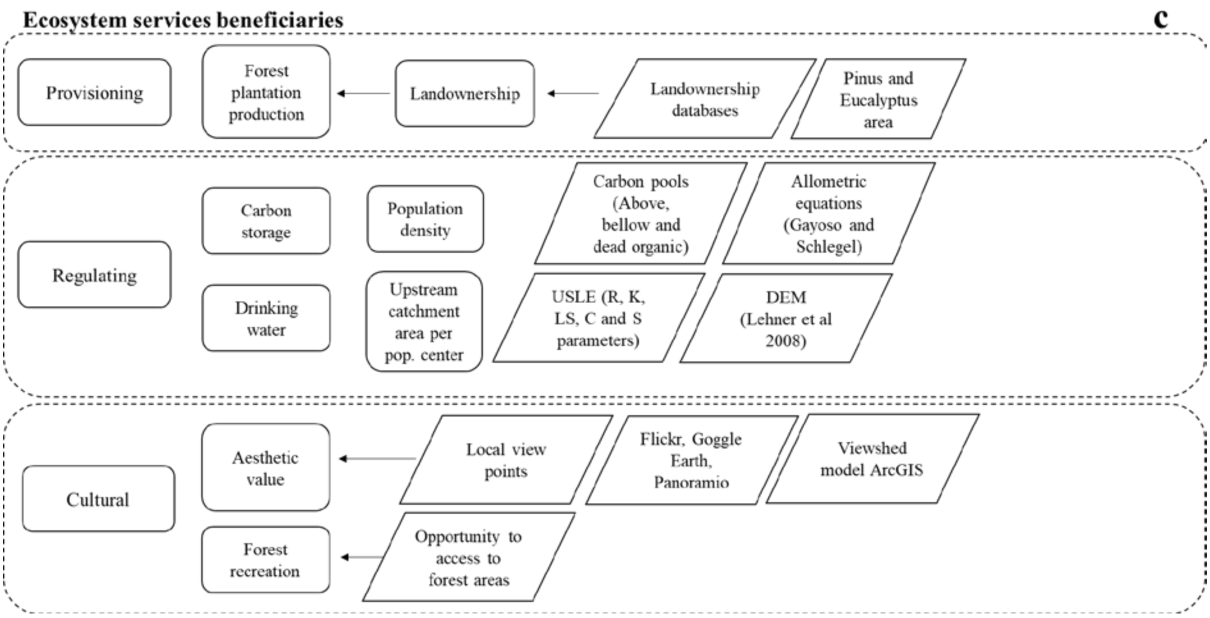

Figure 2.4 Overview of the data and methods used in the third step (c) of this dissertation. This step (c) integrate remote sensing and socioeconomic data to quantify and estimate the ecosystem services beneficiaries in the area.

2.2.3 Landscape planning strategies

In this dissertation, areas that offer solutions to reconnect these fragmented landscapes (Echeverría et al., 2007; Locher-Krause et al., 2017b) were integrated. Structural connectivity areas (SCA) were defined based on different types of landscape elements: linear elements (riparian corridors) and patchy elements (protected areas in the Andes and Coastal range). These SCA were identified based on of the first and second steps of this research and intended to connect the two-main old-growth forest (in the Andes and Coastal range) through riparian areas. We integrated three protected areas located in the Andes (National parks Puyehue, Vicente Perez Rosales and National reserve Mocho Choshuenco) and three in the Coastal Mountain rage (National park Alerce Costero, National reserve Valdivia and the Private reserve Costera Valdiviana). These protected units were connected by a 300 meters buffer area surrounding the main rivers that flow from the Andes range to the sea (San Pedro, Calle Calle, and Rio Bueno rivers). Two areas identified as relevant to support ecological processes at the regional level and considered crucial as the potential habitat of endangered species were also added the SCA. These areas are located in the Central Valley —the San Pedro river valley (658 km^2) and Llollenhue (246.3 km^2).

3 Expanding temporal resolution in landscape transformations: Insights from a Landsat-based case study in Southern Chile.

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3.1 Abstract

Understanding temporal and spatial dimensions of land cover dynamics is a critical factor to link ecosystem transformation to land and environmental management. The trajectory of land cover change is not a simple difference between two conditions, but a continuous process. Therefore, there is a need to integrate multiple periods to identify slow and rapid transformations over time. We mapped land cover composition and configuration changes using time series of Landsat TM/ETM+ images (1985-2011) in Southern Chile to understand the transformation process of a temperate rainforest relict and biodiversity hotspot. Our analysis builds on 28 Landsat scenes from 1985 to 2011 that have been classified using a random forests approach. Base on the high temporal data set we quantify land cover change and fragmentation indices to fully understand landscape transformation in this area. Our results show a high deforestation process for old growth forest strongest at the beginning of the study period (1985-1986-1998-1999) followed by a progressive slowdown until 2011. Within different study periods, deforestation rates were much larger than the average rate over the complete study period (0.65%), with the highest annual deforestation rate of 1.2% in 1998-1999. The deforestation resulted in low connectivity between native forest patches. Old-growth forest was less fragmented but was concentrated mainly in two large regions (the Andes and Coastal mountain range) with almost no connection in between. Secondary forest located in more intensively used areas was highly fragmented. Exotic forest plantation areas, one of the most important economic activities in the area, increased sevenfold (from 12,836 to 103,540 ha), especially during the first periods at the expense of shrubland, secondary forest, grassland/arable land and old grown forest. Our analysis underlines the importance of expanding temporal resolution in land cover/use change studies to guide sustainable ecosystem management strategies as increase landscape connectivity and integrate landscape planning to economic activities. The study is highlighting the key role of remote sensing in the sustainable management of human-influenced ecosystems.

Keywords: Land cover change; deforestation, forest fragmentation; Southern Chile; Landsat.

Highlights

- Expand the temporal scale contributed to identify different intensities transformations within the study period.
- Deforestation process differs in intensity within the study period.
- Old growth forest was converted into more intensive land cover.
- Native forest shows continuous fragmentation patterns over time.

3.2 Introduction

The high anthropogenic pressure on land utilization widely transformed ecosystem patterns and processes across a range of temporal and spatial scales (Ellis et al., 2013; Hietel et al., 2004; Turner, 1989b; Vitousek, 1997). Therefore, land use/cover is recognized as an important driver of global environmental change (Foley et al., 2005; IPCC and Barker, 2007; Lambin and Meyfroidt, 2011; B. L. Turner et al., 2007). Land use/cover transformation has serious impacts on biodiversity as well as on ecosystem goods and services essential for human well-being (Diaz et al., 2007; Millennium Ecosystem Assessment, 2005). Furthermore, the frequency and intensity of these changes influence the current status of ecological systems impacting vegetation composition, biodiversity, biogeochemical cycles, and soil degradation (Lambin et al., 2003; Osvaldo E. Sala et al., 2000; B. L. L. Turner et al., 2007; Vitousek et al., 1997). Understanding landscape change trajectories in a higher temporal frequency enable the identification of natural and anthropogenic disturbances especially in dynamic systems (Lunetta et al., 2006; Pflugmacher et al., 2012b). As land use/cover change is neither uniform nor random remote sensing has become an important and widely used technique for managers and researchers, to map and monitor ecosystem modification over time (James Hansen et al., 2000; Loveland et al., 2000; Zhu and Woodcock, 2012). In order to map the earth's surface, a wide range of satellites is available with different spectral, spatial and temporal characteristics (Kennedy et al., 2009; Stow et al., 2004). The Landsat program provides the largest temporal record of space-based land observations, with more than 40 years of multispectral imagery (Hansen and Loveland, 2012; Ju and Roy, 2008). In 2008, NASA (National Aeronautics and Space Administration) and USGS (U.S. Geological Survey) opened the Landsat achieve revolutionizing the use of remote sensing data (Wulder et al., 2012), offering a huge potential to monitor landscape dynamics by continuous time series analysis (Griffiths et al., 2013; Huang et al., 2010; Kennedy, Yang, & Cohen, 2010; Lambin & Geist, 2006; M. G. Turner et al., 2012, Zhu & Woodcok 2014). The spatial resolution (30 meters) of the Landsat sensors together with its sample frequency of 16 days and the availability of relatively homogeneous measurements over a long period, make it a crucial dataset for monitoring dynamic landscapes (Goodwin et al., 2013; Wulder et al., 2008). A significant problem for land use/cover change detection is clouds and cloud shadow contamination, in particular in areas with high climatic variability. In these areas, the multi-temporal analysis offers several advantages, including improvement of spectral variations (i.e., from topology and phenology) and allows quality enhancement of cloud- obscured images (Canty and Nielsen, 2008; Ju et al., 2012; J. G. Masek et al., 2006). However, the huge amount of information available through time series brings also methodological challenges in image processing and change detection (Rodriguez-Galiano et al., 2012). Non-parametric machine learning methods such as random forests (Breiman, 2001) provide a robust alternative to traditional image classification methods (Gislason et al., 2006; Pal, 2005; Waske and Braun, 2009). Integration of machine-learning classifier offers considerable advantages for land cover analysis, especially in dynamic systems with high cloud covers, low amount of cloud-free scenes as the south of Chile. Southern Chile is a biological hotspot – it is of particular importance to conserve and protect the Valdivian temperate rain forest with its high levels of endemism (90% at species level and 34% at the genus level for woody species) and endangered species (J. J. Armesto et al., 1998; Arroyo et al., 1996). The region also has been characterized by WWF and the World Bank (Myers et al., 2000; Olson et al., 2001) as one of the most threatened eco-regions in the world. Despite its high biological importance, the area has undergone a strong land use/cover transformation similar to most regions in Chile (J. J. Armesto et al., 1998; Echeverria et al., 2006a; Lara et al., 2011, 2009; Rozzi et al., 1994). In Chile - as in entire South America - the colonization period sets the starting point of strong natural resource utilization. Agricultural conversion and the commercial logging of valuable native species were the main processes that shaped the large native ecosystem conversion from mid-1800 (Armesto et al., 2010). This land transformation resulted in one of the most rapid deforestation events in Latin America and led to severe soil erosion especially on the coastal mountain range where 59% of the surface eroded (Armesto et al., 2010; Otero, 2006; Salazar et al., 2015; Siebert, 2003). In an attempt to control the intense erosion processes, exotic forest plantations were introduced in the second half of the 20th century. However, it took until the 1970s until the forestry industry rapidly expanded, focusing on commercial plantations of *Pinus radiata*, *Eucalyptus globulus,* and *Eucalyptus nitens.* Trade reforms implemented in Chile in the 1970s created large economic incentives for exotic forest plantations. Governmental economic subsidies, i.e., Law Decree 701 (DL 701), which reimbursed 75% of the afforestation expenses to the landowners after certifying an adequate rate of survival, resulted in a substitution of native ecosystems due to the economic benefits (Niklitschek, 2007). The increase of planted areas caused a dynamic land use/cover transformation that has been recognized as one of the most important drivers of deforestation in the Central-South of Chile (Díaz et al., 2011; Echeverria et al., 2008, 2006b; Lara et al., 2011; Nahuelhual et al., 2012). This transformation also led to a change in the composition and configuration of native forest ecosystems (Echeverría et al., 2007; Little et al., 2009; Nahuelhual et al., 2012). Even though existing studies created an important base of knowledge with respect to the human utilization of natural resources, they are restricted by their limited temporal resolution with only two or three snapshots over time (Altamirano et al., 2013; Echeverria et al., 2008, 2006b; Miranda et al., 2015; Schulz et al., 2010;

Zamorano-Elgueta et al., 2015). This discrete information does not allow us to identify whether the transformation is continuous or specific, and its magnitude over time. An analysis that integrates data sets of high spatial-temporal density - which could improve the understanding of the historic impact and its influence on the current state of the ecosystem in this region has been missing so far. Our study aims at closing this gap by investigating land cover dynamics at a high temporal scale to better understand the historic transformation of the ecosystem over time. Our objectives are:

- to reveal patterns of land cover change processes at a high temporal scale based on all suitable Landsat scenes available for the study area in the south of Chile from 1985 to 2011, with a special focus on forest cover changes.
- to analyze the changes in land cover composition and land cover configuration across a geomorphological gradient (from the Coastal mountain range, over the Central Valley to the Andes mountain range) in southern Chile.

3.3 Material and Methods

3.3.1 Study area

The study area is located in Southern Chile, Northern Chilean Patagonia (73°20’ W-39°25’ S - 71°59’ W-41° 14’ S) covering 16,625.7 km² (c.f. Figure 3.1). Administratively it belongs mainly to the Los Rios region - the remaining part belongs to the North of Los Lagos administrative region. The climate is temperate oceanic with a Mediterranean influence, showing an annual mean temperature of 11.9ºC. Annual rainfall is about 2,500mm, with the highest values in the winter season (June-September) (CIREN, 1994; di Castri & Hajek, 1976). Altitude ranges from sea level to 2,422 m in the study area. The most important morphological units from west to east are the Coastal mountain range (up to 900 m), the Central Valley (up to 250 m) and the Andes mountain range (up to 2,422m). Vegetation is characterized by Valdivian temperate rainforest, a temperate broadleaf and mixed forest that is subdivided into four vegetation zones: deciduous forests, Valdivian laurel-leaved forests, Northern Patagonian and Evergreen forests (Gajardo, 1994; Veblen et al., 1983).

The area is home to circa 381.720 inhabitants, which represent 2.2% of the population of Chile (INE Instituto Nacional de Estadística, 2012). From these inhabitants 68.3% are urban, and 31.3% rural residents; 17% describe themselves as Mapuches, a group of Chilean indigenous inhabitants. The Los Rios region was established in 2007 as a new administrative region. This resulted in an increase in the urban population, partly caused by internal migration from rural areas. The most important economic activities in the region are related to the forest industry and the agricultural sector. The forest sector mainly focuses on exotic plantations (e.g., pulp, shipyard, and paper industry) as well as on wood processing factories triggered by the high global demand for wood and paper products. These products are targeted to external markets, indicating a spatial decoupling or land teleconnection (Friis et al., 2015), of land use/cover change processes due to global market trends. Agricultural activities are focused on livestock and crop and berries production.

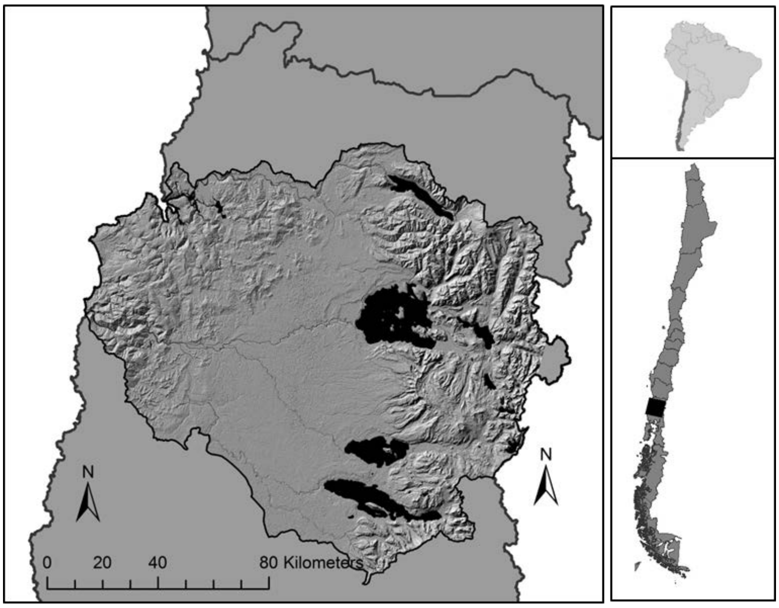

Figure 3.1 Study area location within Southern Chile and South America. Topography shows the three major morphological units: Coastal mountain range, Central Valley and the Andes mountain range (from West to East). Lakes are shown in black.

3.3.2 Data

3.3.2.1 Remote sensing data

We used all available Landsat images (path 233/row 088) that fulfilled our quality criteria for the analysis. We aimed for scenes during the growing season (September-February) with less than 20% cloud cover (cf. Table 3.1). This constraint was an important limitation due to the low amount of cloud-free scenes per year – the non-availability of cloud-free Landsat scenes during the growing season led to a data gap between 1986 and 1998. We downloaded the data from the USGS at an L1T processing level, which includes radiometric correction, systematic geometric correction, precision correction using ground control points, and parallax correction due to local topographic relief.

Table 3.1 Landsat scenes together with training and validation data used for the analysis.

Date	Season	Sensor	Cloud Cover (%)	Train/validate data
1/25/1985	Summer	L4-5 TM	0	Orthophoto
9/25/1986	Spring	L4-5 TM	0	Orthophoto
2/14/1998	Summer	L4-5 TM	12	Aerial photography
11/16/1999	Spring	L4-5 TM	0	Aerial photography
10/25/2000	Spring	L7 ETM+ on	4	Field data
11/29/2001	Spring	L7 ETM+ on	0	Field data
10/10/2003	Spring	L4-5 TM	0	Field data
2/15/2004	Summer	L4-5 TM	8	Field data
2/1/2005	Summer	L4-5 TM	0	Field data
11/19/2006	Spring	L4-5 TM	3	Field data
12/10/2008	Spring	L4-5 TM	8	Aerial photography
2/28/2009	Summer	L4-5 TM	18	Aerial photography
1/14/2010	Summer	L4-5 TM	6	Field data
1/1/2011	Summer	L4-5 TM	10	Field data

3.3.2.2 Land cover reference data

Land cover image classification and posterior accuracy assessment were based on a total of 800 training and 260-validation ground points independently selected and randomly distributed in each scene. Ground points for 2010-2011 were collected during the growing season. Ground points for older dataset were based on the following data: i) land cover information from the national inventory of native resources (Catastro Nacional) together with thematic maps derived from the interpretation of aerial photography from 1994-1996 (CONAF-CONAMA-BIRF, 1999) and its update (2007; 2013); ii) a set of panchromatic orthophotographs, (1:20.000, IGM) from the year 1985-1994, provided by the Laboratory of Geomatics, Universidad Austral de Chile; iii) complemented with field data provided by the Laboratory of Geomatics, Universidad Austral de Chile collected in the same study area (2000 to 2006). Based on this information we created homogenous land cover polygons to train and validate each image during the classification process.

We based our analysis on nine land cover classes, representing the dominant land cover and land cover categories present in the study area (cf. Table 3.2)

Table 3.2 Description of land cover categories defined in the study area.

Category	Description
Urban	Urban and industrial areas, roads and other anthropogenic surfaces
Grassland/Arable land	Natural and artificial annual pastures, agricultural areas with different crops such as wheat or oats.
Shrubland	Areas with vegetation dominated by shrub species with < 10% tree cover, the result of natural succession or native forest logging.
Secondary forest	Areas with secondary growth native forest result of natural succession or native forest logging.
Old-growth forest	Areas with pristine or almost intact mature native forest
Forest exotic plantation	Areas planted with exotic forest species mainly *Pinus radiata* and *Eucalyptus sp*.
Bare land	Areas temporally or permanently without vegetation
Water	Water bodies such as river, lakes, ponds
No information	Areas without land cover information due to clouds and shadows, masked by Fmask.

3.3.3 Data preprocessing and analysis

3.3.3.1 Image preprocessing

The preprocessing chain consists of two steps. First, we performed an atmospheric correction using the radiative transfer based Landsat Ecosystem Disturbance Adaptive Processing System tool (LEDAPS) (Kaufman et al., 1997; J. G. Masek et al., 2006). LEDAPS uses information on the sea level atmospheric pressure, water vapor characterization, ozone level and aerosol optical thickness (AOT). Sea level atmospheric pressure and water vapor characterization were taken from the National Center for Environmental Prediction (NCEP) and the National Center for Atmospheric Research (NCAR). Information on ozone came from the NASA Earth Probe Total Ozone Mapping Spectrometer (EP TOMS). A static global 0.05° digital elevation model was used by LEDAPS to adjust the atmospheric pressure from sea level to surface level (Ju et al., 2012). The estimation of aerosol optical thickness (AOT) was based on the dense dark vegetation (DDV) approach and assuming a fixed “continental” aerosol model.

In the second step, we produced a suitable cloud, cloud shadow and snow mask using Fmask to avoid or to reduce the brightening effect of clouds and their shadows. This two-step masking algorithm has been shown to improve the level of accuracy by avoiding misclassification in the time series (Zhu and Woodcock, 2012).

3.3.3.2 Land cover classification

We performed a supervised image classification using random forest machine learning approach (Breiman, 2001). The algorithm was trained based on the land cover reference data (previously described) to discriminate between different land cover classes. The analysis was carried out using R (R Development Core Team, 2013) and the randomForest package (Liaw and Wiener, 2002). Random forest classifier has been applied successfully in a number of remote sensing studies

(Doktor et al., 2014; Gislason et al., 2006; Pal, 2005; Waske and Braun, 2009), where it showed that the approach is superior to the widely used maximum likelihood approach. The algorithm builds on classification and regression trees but overcomes their sensitivity towards noise in the data instead of relying on a single decision tree, using the majority vote of a forest of decision trees fit to bootstrap samples from the original data. While individual decision trees suffer from a high variance of estimates, the averaging across the bootstrap sample leads to a significant variance reduction (Hastie et al., 2009; James et al., 2013). In addition to bagging approaches, random forests decorrelate the trees by using only a random sample of the variables (i.e., spectral bands) for each split. We trained the random forest classifier algorithm and selected 500 decision trees with two variables for each split.

To improve classification and avoid implausible land cover change events, we include constraints to land cover transformations. These constraints were based on the ecological characteristics of native and exotic species and ecosystems dominant in the study area (i.e., growth rates). Posteriori to the classification process, we validated the accuracy of the classification of each Landsat-derived map individually with an independent set of 260 control polygons. We derived confusion matrixes and calculated indicators such as the producer's and user's accuracy as well as the overall accuracy to evaluate the classification performance (Foody, 2002; Pontius et al., 2011, 2004) (cf. Table 3.3).

3.3.3.3 Land cover change analysis

To perform land cover change analysis, we merged land cover information from two consecutive years to avoid large areas without information due to clouds and cloud shadows. This resulted in seven combined land cover maps for the following years: T1=1985-1986; T2=1998-1999; T3=2000-2001; T4=2003-2004; T5=2005-2006; T6=2008-2009; T7=2010-2011. To analyze the land cover information, we calculated the extent, net change, gain and losses of each land cover class between consecutive periods over time. We generated cross-tabulation matrixes using IDRISI SELVA (Eastman, 2012) to derive and analyze the different land cover trajectories. The frequency of land cover change was identified by combining binary maps of change/no change events between the seven combined land cover maps.

We estimated the annual deforestation for the native forest (old-growth forest and secondary forest) with the compound-interest-rate formula proposed by Puyravaud (2003):

$$r = \ln\,(A_2 / A_1) \times (100 / t_2 - t_1)\,;$$

where A_1 and A_2 are the areas of native forest at the beginning (t_1) and the end of each period (t_2). r is percentage per year.

3.3.3.4 Landscape pattern analysis

In addition to land cover composition, we also identified changes in land cover configuration for native forests and exotic forest plantations. We calculated five landscape pattern indices for secondary forest, old-growth forest and exotic forest plantation, separately. These indices provide crucial information to characterize and monitor the spatial configuration of natural and cultural landscapes (i.e., regrowth due to natural succession, clear-cut logging, afforestation, reforestation) (Gergel and Turner, 2002; Jiang et al., 2014; Mcgarigal, 2014; Walz, 2015). We quantified the following indices using FRAGSTATS v4 (Mcgarigal, 2014):

- mean patch size (MPS), the average size of a patch of a land cover class. An increase in mean patch sizes typically indicates a reduced fragmentation.
- largest patch index (LPI), the percentage of the total landscape area covered by the largest patch of a land cover class. Higher values indicate typically less fragmentation.
- total edge length (TE) [km] for each class as an indicator of patch shape– the larger the total edge length, the higher the complexity of the shape of the patch. Increasing edge length indicates a loss of core area and typically an increase in fragmentation.
- mean proximity index (MPI), a measure of the degree of patch isolation and class fragmentation; it measures for each patch the size of and the distance to all neighboring patches of the same class. A patch with lots of other large patches in close proximity will have a large index value (Gustafson and Parker, 1992).
- total core area (TCA), which is calculated as the sum of the core area of all patches of a land cover class. A decrease in the total core area indicates a reduction of high-quality interior habitat area. We calculated the total core area with a 500m buffer.

3.4 Results

3.4.1 Land cover classification

The land cover classification overall accuracy ranged from 91.4 to 96.4% (cf. Table 3.3), with the lowest values in 2000-2001 and highest value in 1998-1999, respectively. Regarding the individual land cover classes, shrubland shows the lowest producer's accuracy, with intermediate accuracy values in 1998-1999 and 2003-2004. (Cf. Table 3.3 and online appendix). These intermediate accuracy values were presumably caused due to the short successional gradient between secondary forest and shrubland.

Old-growth forest was the land cover class with the highest accuracy for each land cover map with values from 96% to 99% (producer's accuracy) and 85% to 98% (user's accuracy). Exotic forest plantations were also classified with a good performance (producer's accuracy: 65%-92%) with low values only in 1985-1986.

Table 3.3 Performance indicators for each land cover map and land cover category (P: Producer's accuracy, U: User's accuracy, both in percentage).

Time	Overall accuracy	Urban		Grassland		Shrubland		Secondary forest		Old-growth forest		Exotic forest plantation		Bare soil		Water	
	%	P	U	P	U	P	U	P	U	P	U	P	U	P	U	P	U
T1: 1985/86	96	85	98	97	99	76	85	87	66	98	98	65	98	100	97	100	100
T2: 1998/99	94	97	100	98	99	65	71	84	70	98	96	90	100	100	99	100	100
T3: 2000/01	91	99	99	99	98	67	83	65	77	99	85	84	100	99	99	100	100
T4: 2003/04	94	98	99	87	86	65	70	78	70	98	92	92	100	97	98	100	100
T5: 2005/06	95	99	100	89	98	96	66	84	81	97	93	92	99	99	100	100	100
T6: 2008/09	95	100	87	97	99	66	74	81	80	99	90	91	100	98	99	100	100
T7: 2010/11	94	100	97	94	96	70	74	72	74	98	87	87	100	99	99	100	100

3.4.2 Changes in land cover composition over time

The magnitudes and trajectories of change for the different land cover classes changed over time – a finding that would not have been revealed at a coarser temporal resolution.

Old-growth forest decreased continuously, but in different magnitude over the 26 years from 32 (to 27% 5,310.9 km² to 4,624.4 km²), with a 4.8% net loss and an annual deforestation rate of 0.65% for the whole period. Between 1985-1999 results showed the highest deforestation rate with 1.2% net loss, followed by a slowdown to 0.8% in 2003-2004 (compared with the previous period). These rates were remarkably higher than the average loss over the complete period.

Exotic forest plantation covered only a small part (0.7%) of the area in 1985 - this changed drastically after 1998 (4%) when the area covered by plantations expanded, reaching its maximal extension of 6% (1036.7 km²) in 2005-2006. This land cover class also showed the highest net change increase for a single class of 706% in 26 years, more concentrated in 1985-2004 (with a maximum of 80% and 15.2% minimum).

Secondary forest increased over time from 7.5% in 1985-1986 to 13% in 2010-2011 (1,477 km² to 2,603 km²) with an average net loss of 2.1% per year, reaching the highest net change in 1999-2001 (3.9%). Shrubland areas showed mixed trends over time, with a net loss of 0.8% for the total period. As mentioned above, the similarity within shrubland and secondary forest might have led to confusion between the two classes for single periods. Since we enforced transformation rules that prevent unrealistic changes between shrubland and secondary forest the uncertainty of the changes can be considered lower than indicated by the accuracy values. For the whole period, we can assume that the change rates are not affected strongly.

Grassland/arable land and old-growth forest were the predominant land cover classes during the entire analysis period covering around 25% and 30% of the total study area respectively, but with important differences among them. Grasslands/arable land moderately decreased over the whole period from 4,782.7 km² to 4,348.2 km² (28% to 26% of the total area) with an intermediate increase in 2005-2006. The main decrease in grassland/arable land occurred within 2001-2004 and 2006-2009 (Figure 3.2 to Figure 3.3).

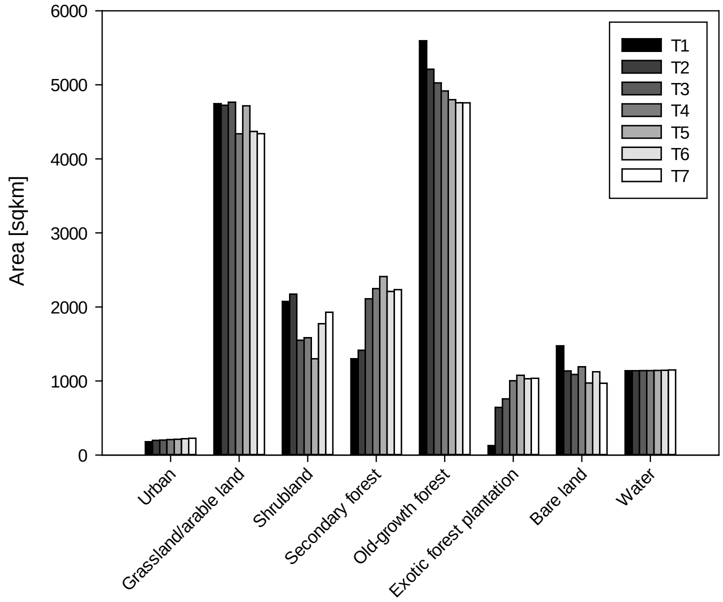

Figure 3.2 Change in land cover composition (T1:1985-1986; T2:1998-1999; T3:2000-2001; T4:2003-2004; T5:2005-2006; T6:2008-2009; T7:2010-2011).

The categories without vegetation such as urban, bare land and water covered about 15% of the study area. Urban areas showed a small increase over time from 1.03% to 1.3%, with an overall net change of 0.3%. The area without vegetation cover showed oscillations, mainly driven by human-related modifications as clear-cut and the regrowth of exotic forest plantations.

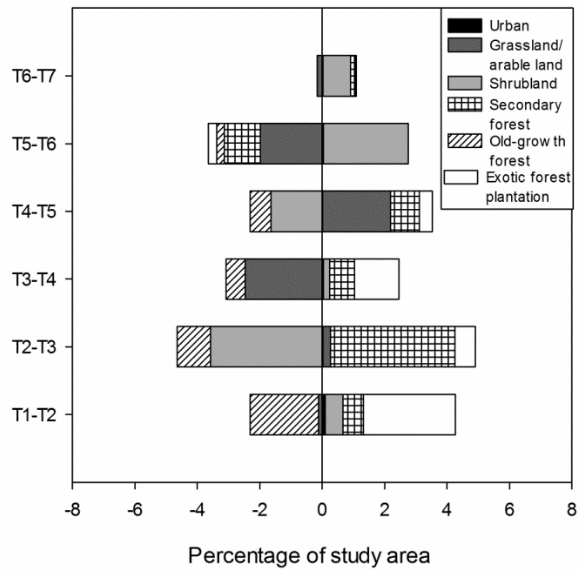

Figure 3.3 Net changes (gain minus losses) for each land cover class over time (T1:1985-1986; T2:1998-1999; T3:2000-2001; T4:2003-2004; T5:2005-2006; T6:2008-2009; T7:2010-2011).

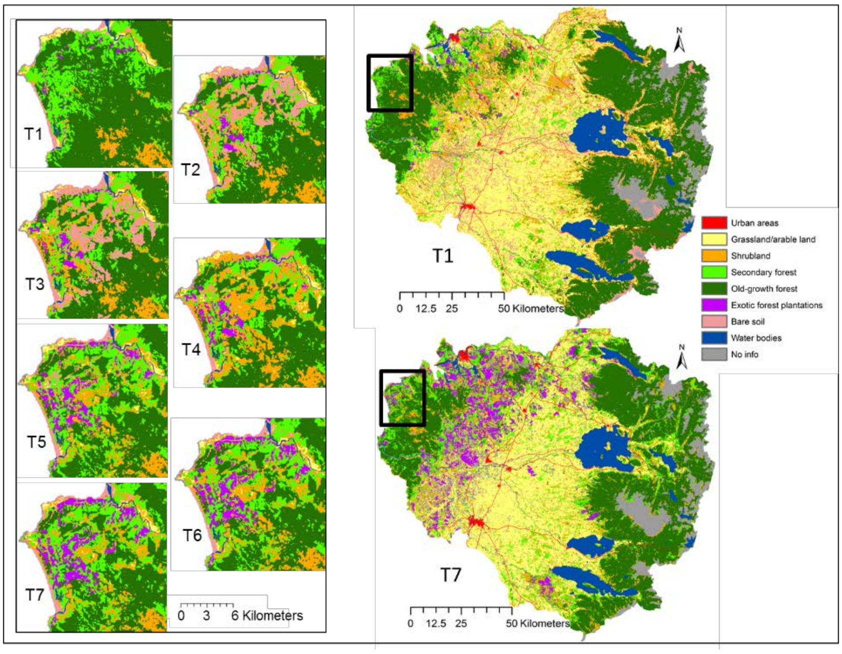

Figure 3.4 Overview of landcover change. Land cover maps for the initial and last period together with a detailed example of land cover transformation over time in the Coastal mountain range.

The areas strongest affected by human activities were located in the Central Valley, the coastal areas and close to water bodies (Figure 3.4) – all areas easily accessible using transport. The Andes Mountains, which are more difficult to access, show fewer changes than the other geographic areas over time. Within the study period (1985-2011), our results reveal that land cover in most areas changed one or twice (14.1% and 13.6% of the total study area), areas with three land cover changes were less frequent (8.9%) while areas with more than 4 land cover changes were rare (Figure 3.5). The areas with no change (56.1%) correspond mainly to old-growth forest in the coastal and the Andes mountain range.

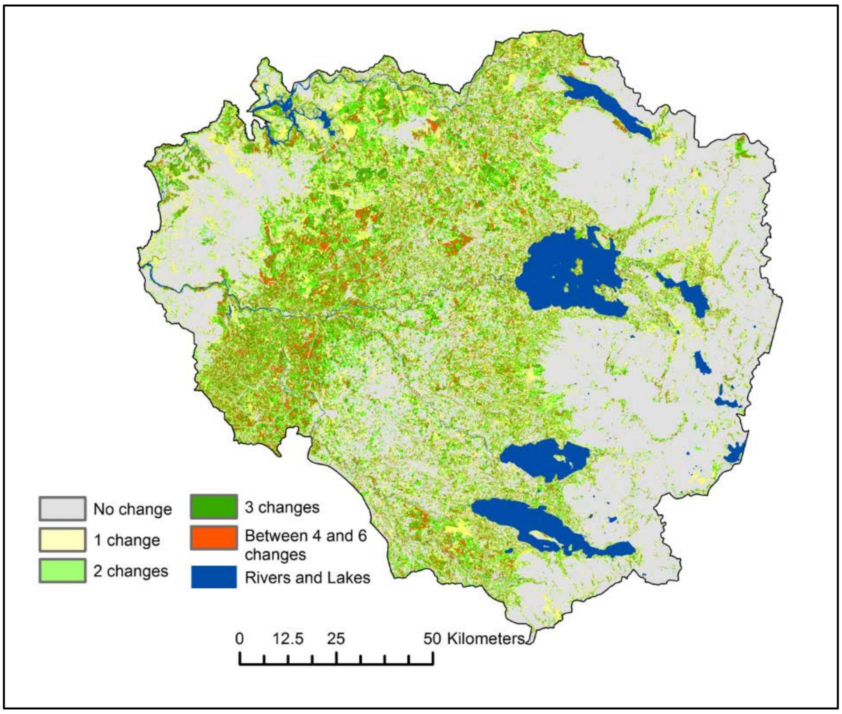

Figure 3.5 Spatial distribution and persistence of land cover changes over the whole study periods (1985-2011). The different colors indicate how often the lands cover pixels changed over time (i.e., one, two, three or more times) while the grey areas indicate the areas that remained without change.

3.4.3 Land cover transitions

Secondary forest and exotic forest plantation were the categories with the highest net gain within periods and for the whole study period - 93,164 ha and 90,676 ha respectively. The largest old-growth forest loss occurred during 1985-1999 with 38,148 ha when the area was converted mainly to shrubland, exotic forest plantation; secondary forest and grassland/arable land (Figure 3.6e).

Grassland/arable land loss and gained areas mainly to and from two land cover classes, bare land, and shrubland (Figure 3.6a). However, in 1985-1986 grassland/arable land was also converted into exotic forest plantations (0.7%). Shrubland gained area from grassland/arable, bare land and old-growth forest and lost area to secondary forest, exotic forest plantation and grassland/arable land (Figure 3.6b).

Between 1985 and 1999, secondary forest gained area mainly from shrubland with the highest conversion rate (3.6%). The area that secondary forest continuously gained from old-growth forest (1985 – 2006) might be attributed to a degradation process. Secondary forest was converted especially to exotic forest plantation (1985 to 2009) and to bare land (1985-1999; 2004-2009).

Transition patterns of old-growth forest changed distinctively across time: between 1985 and 1999: the transformation to shrubland (1.3%), exotic forest plantations (0.2%) and bare land (0.3%) was the dominant process during this period. The conversion to secondary forest gained more importance between 1998 and 2009 (Figure 3.6e).

The transformation rate for exotic forest plantations changed during the study period. Between 1985 and 1999, exotic forest plantations revealed the largest conversion from all vegetation related land cover classes: secondary forest (0.9%), shrubland (0.8%), grassland/arable land (0.7%), bare land (0.3%), and old-growth forest (0.2%). The only losses from exotic forest plantations were conversions to bare land (2008 to 2011) that might be a result of the clear-cutting harvesting process; a common practice in the forest industry.

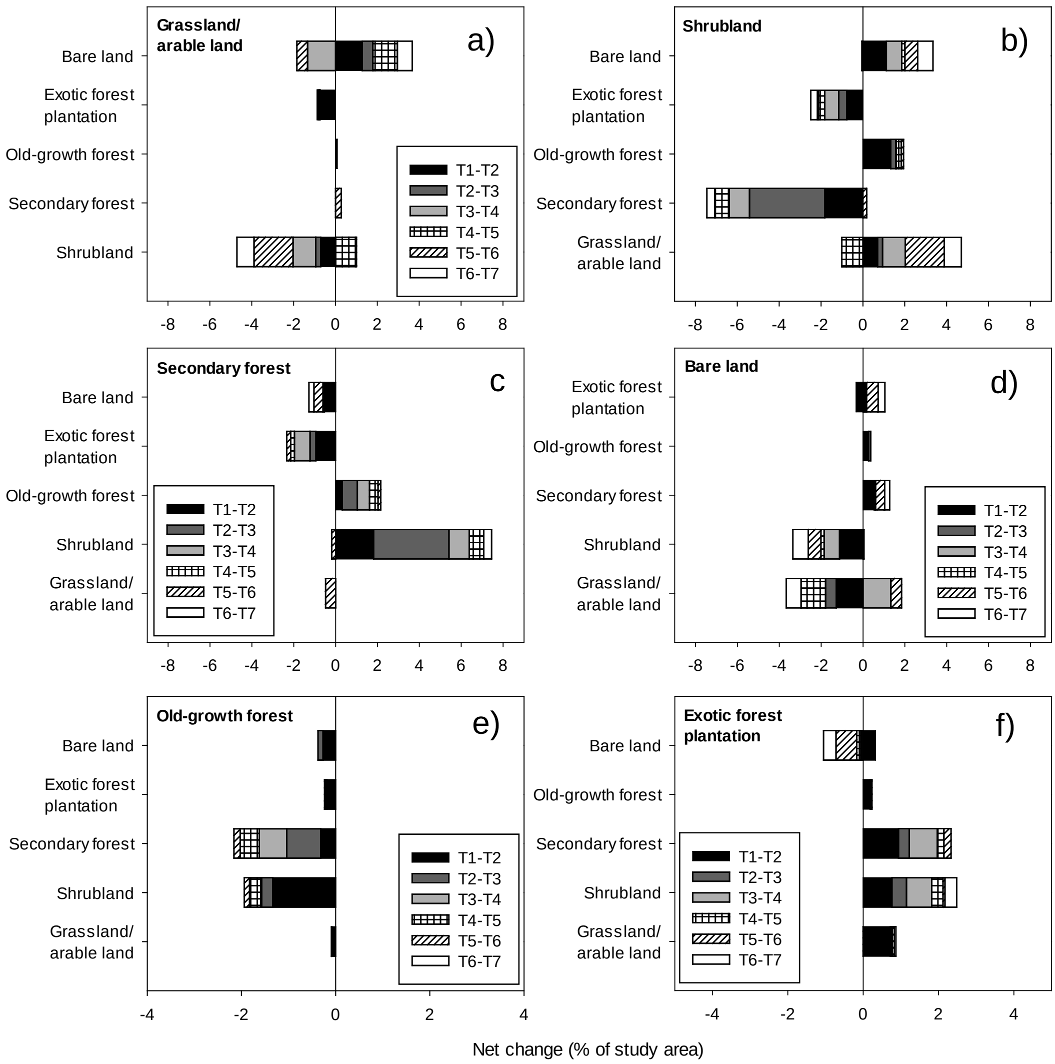

Figure 3.6 Transition between land cover in the different period (percentage of the study area) (T1:1985-1986; T2:1998-1999; T3:2000-2001; T4:2003-2004; T5:2005-2006; T6:2008-2009; T7:2010-2011).

3.4.4 Changes of landscape indices over time

Land cover configuration differed significantly among the three-different forest/plantation land cover classes (Figure 3.7). The high values of old-growth forest for the total core area, largest patch index, mean proximity index and mean patch size, were caused by larger, more connected and compact patches, which remained in more remote areas with low or difficult access (Figure 3.7). Old-growth forest showed a strong decrease in the mean patch size index and total core area between 1985 and 1999 with a relative change of -22.2% and -9.34%, respectively. The total core area and the mean proximity index decreased to a lower degree during the next periods while the total edge length increased slightly. After 1999, landscape indices for old-growth forest remained almost constant, with the exception of total edge length that slightly decreased. This indicates that the largest patches of old-growth forest were fragmented at the edges especially between 1985 and 1999. Afterward, the largest patches seem to have stayed relatively intact while fragmentation at the edges of smaller forest patches leads to a continuous loss of core area, a key element to preserve ecological processes and species (Echeverria et al., 2006a).

In comparison to old growth forest, the secondary forest was characterized by much smaller and more disconnected patches which were mainly allocated in the Coastal and Central Valley (Figure 3.5), in more intensive production areas (where primary forest was substituted during the colonization time, due agricultural production). The mean patch size changed markedly, first decreasing and afterward increasing with a relative change of -31.5% and 42.7%. The same pattern is identifiable in the total core area indicator with a relative change of -12.5% and 75.2%. Total edge length of secondary forest increased strongly from 1985 to 2001 and remained relatively constant afterward. The largest patch index and the mean proximity index barely changed over time, with a small decrease until 1999. This indicates that secondary forest patches were combined, forming larger patches that led to an increase in the core area. Still, compared to old growth-forest patches are smaller and much stronger isolated. Partly this higher fragmentation of secondary forest might be attributed to the fuzzy class boundary towards regrowing shrubland.

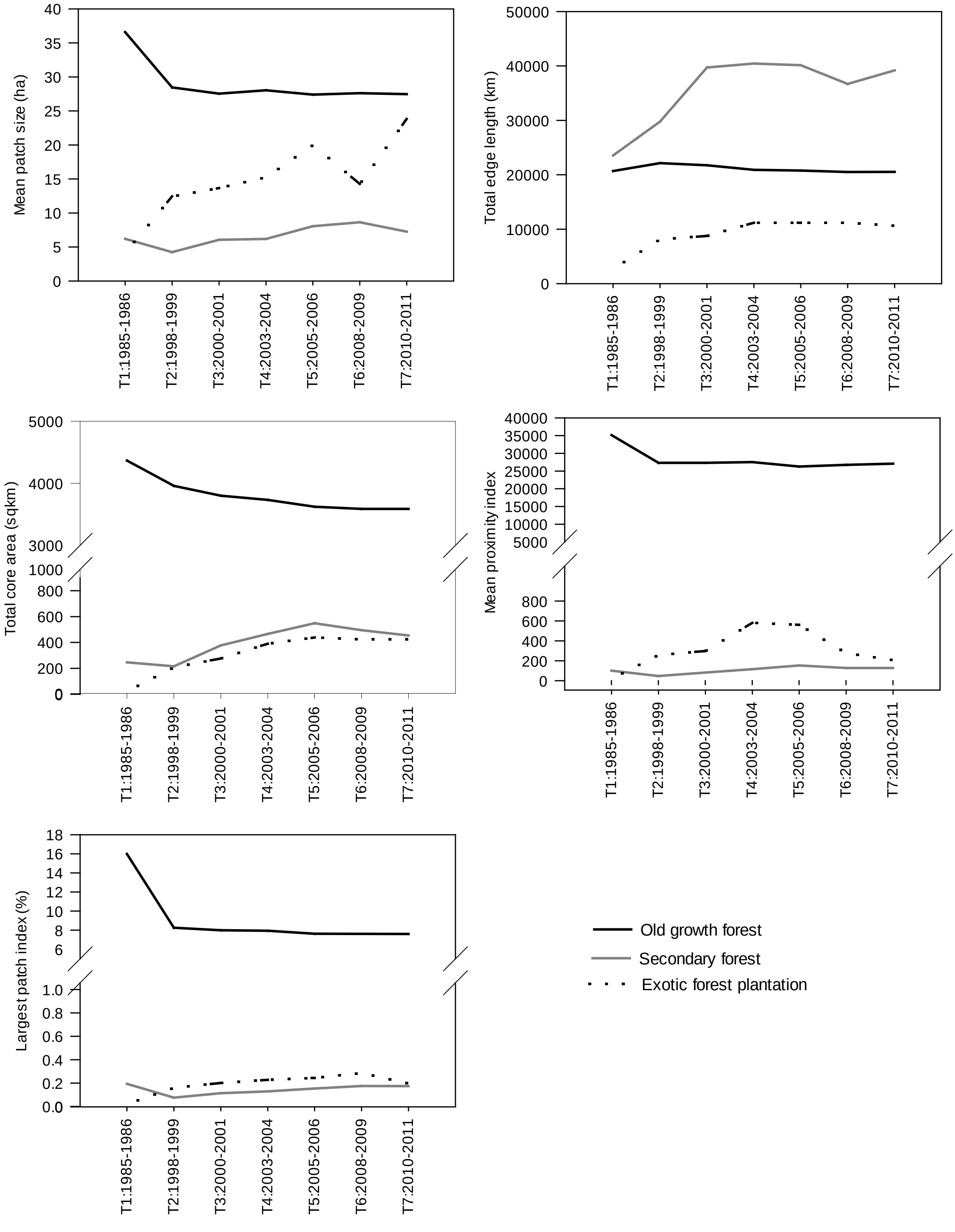

Figure 3.7 Landscape indices change over time for old-growth forest, secondary forest, and exotic plantations. Total core area, total edge, largest patch index and mean proximity index graphs include a scale break to show the three-land cover categories over time.

In contrast to old–growth forest and secondary forest exotic forest plantations mean patch size increased strongly (relative change of 199.4 %) with a slight decrease in 2006 (relative change of -3.3 %). Total core areas, total edge length and the large patch index increased over time. From 1985 to 2006, the mean proximity index increased twofold followed by a continuous decrease afterward.

These indices revealed that exotic forest plantations increased their dominance in the landscape, with larger, more connected and compact patches indicating large homogeneous areas used as plantations.

3.5 Discussion

Identification and reporting errors during the land cover classification are the base for a reliable landscape pattern analysis (Foody, 2010; Kennedy et al., 2009; Shao & Wu, 2008). Our overall accuracy values showed high reliability of classifications in each land cover map, with slight variations ranging from 91.4% (2000-2001) to 96.5% (1998-1999). Such variations in accuracy can be attributed to seasonal differences and characteristics of the input data especially cloud cover. While only summer scenes were available for 2000-2001, a summer and a spring scene had been used in 1998-1999. In addition, we faced methodological difficulties such as high cloud cover and low-quality images. An option for (future) improvement might be the fusion of data from different sensors. Multi-sensor techniques, including, i.e., additional MODIS data (launched 1999) and aerial photography (especially important to extent temporal resolution before satellite era) are interesting options to increase the amount of information to enable the use of multi-seasonal data. (Hilker et al., 2009; Senf et al., 2015; Walker et al., 2006; Xin et al., 2013). The accuracy of the classification of some land use classes were intermediate, especially between shrubland and secondary forest. However, it should be kept in mind that the ecological difference between shrubland misclassified as secondary forest and secondary forest is presumably not too strong since the misclassified shrubland presumably was already in a transition state towards to open secondary forest. Therefore, assessed ecological impacts of land use change in the region should still be reliable after taking this shortcoming into account.

Based on the high temporal land cover information, we identified a highly dynamic system. The area is under a continuous transformation process between land cover categories over the six temporal periods, as well as dominated by an irregular but intense deforestation. The identification of the magnitudes of land cover change trajectories has been recognized as a key element to establish sound management and conservation strategies, which especially applies in threatened and conservation priority areas (Andrew et al., 2015; Vogelmann et al., 2012). Such trends are particularly relevant considering the large magnitude of the deforestation process, especially from 1985 to 1999.

Recent studies also recognized the significant dynamics of these systems, demonstrating the relevance of more dense information to address and fully understand the transformation process (Altamirano et al., 2013). Zamorano-Elgueta (2015) reported a similar trend with 5.1% net loss of native forest and annual net deforestation by 0.2%, for two periods (1985-1999-2011) in the Coastal mountain range. Other studies that were carried out in close areas of the Central-South of Chile reported a native forest annual deforestation rate of 4.5% for 1975-1990-2000 and 1.60% for the period 1986-1999-2008, both in the Coastal mountain range (Altamirano et al., 2013; Echeverria et al., 2006a). Our study helps to understand old-growth forest deforestation trends in the area better. We revealed that the deforestation process was not static over time, but a dynamic process over time - stronger at the beginning of the study period, followed by a slowdown afterward until 2003 and a gradual decrease until 2011. These results stress the importance of quantifying the different magnitudes of land cover transformations across time. The higher the temporal resolution, the easier it gets to spot such dynamics. A high temporal resolution also enables a deeper analysis of the dynamics at a higher temporal resolution and identified areas that were transformed repeatedly. This has been the case for the Coastal mountain range and the Central Valley, where the transformation process was more intense presumably due to the better accessibility and conditions to establish exotic forest plantation (Echeverría et al., 2007; Schlatter and Gerding, 1995).

Old-growth and secondary forest composition and configuration patterns differed remarkably in different periods and over the whole study. Even though it is possible to observe an increase in the total area of old growth-forest and secondary forest combined, it is important to emphasize that they cannot be treated equally because they support diverse ecological functions and processes (Donoso, 1993; Donoso and Lara, 1999; Hobbs et al., 2009, 2006). We observed two main patterns regarding native forest configuration over time: i) old-growth forest shows a low number of bigger patches, more connected and compact than secondary forest. The areas of these patches decreased over time and became less compact and connected, with mainly two remaining populations (one in each mountain range) which have low or no connection between them. ii) Secondary forest was highly fragmented and occurred dominantly in more intensively used areas (Central Valley and Coastal range). This low connectivity across the landscape contributes to the isolation of species (plants, mammals, amphibians, etc.) with the consequent reduction of gene flow between populations and risk for the long-term survival of some species (Andren, 1994; Cushman, 2006; Echeverria et al., 2008; Nick M. Haddad et al., 2015; Lara et al., 2011; Prugh et al., 2008). It is also important to consider that patch size influence the ability to support different ecological functions as, i.e., plant species richness is reported to be dependent on the size of the patch and its core area (shade-tolerant and shade intolerant species) (Donoso and Lara, 1999; Echeverría et al., 2007; Gutiérrez et al., 2009). Even though secondary forest can play an important ecological role, small forest fragments are more threatened to change their species composition and structure due to alien species invasion and loss of species niche (Bustamante and Simonetti, 2005; Pauchard and Alaback, 2004; Torres et al., 2015). We were able to identify a large, but disconnected area of secondary forest in between the exotic forest plantations, especially in the land strip between the Central Valley and the Coastal mountain range. Those areas experienced increases and decreases in almost every period, i.e., buffer areas near to the small streams, and between exotic forest plantation stands. These fragmented areas are under threat of change to more intense land cover with larger profitable economic activities.

This situation also triggers degradation mainly due to exotic forest plantation harvesting process and illegal logging. This makes it difficult to predict the likelihood of these fragments to remain as native forest.

We observed an increase of exotic forest plantations over time that has also been reported in Central and Central South Chile (Altamirano et al., 2013; Echeverria et al., 2006a; Nahuelhual et al., 2012; Zamorano-Elgueta et al., 2015). This increase is not constant in time and takes place mainly between 1985-1999 and 2000-2004. The subsidy for exotic forest plantations (DL 701) together with the market-oriented strategy adopted by the government triggered the substitution of extensive areas of secondary forest and degraded old-growth forest. Native forest substitution was exacerbated mainly because subsidies did not differentiate between exotic forest plantation established on abandoned farmland/degraded forest and the ones on old-growth forest areas (Armesto et al., 2010; Lara and Veblen, 1993; Niklitschek, 2007). The increase of exotic forest plantations in Central and Central-Southern Chile has been linked to changes in hydrological regimes, deforestation, native forest fragmentation, biodiversity loss, soil erosion, decrease in nitrogen and phosphorus retention, and poverty of the adjacent communities (Brockerhoff et al., 2008; Bustamante and Simonetti, 2005; Lara et al., 2009; Little et al., 2009; Nahuelhual et al., 2012; Oyarzun et al., 2007). All these potential transformations affect the way in which the environment supports human existence and wellbeing (Millennium Ecosystem Assessment, 2005). In addition, considering the large economic importance of exotic forest plantations in the country, it is becoming crucial to find a balance between the supply of different ecosystem services. Innovative, improved land use planning is necessary to integrate socioeconomic activities and ecosystem conservation. Future research should focus on integrated ecosystem services assessments in order to provide knowledge that allows policy makers to optimize the utilization and protection of natural resources (Seppelt et al., 2013).

The high temporal resolution also enabled us to observe one of the most important land cover transformation in the period 1985/1986-1998/1999: during that period, a large transition from old-growth forest to shrubland - clustered in the coastal mountain range - took place. These areas were converted to shrubland, fragmented forest or exotic forest plantations (Figure 3.4). The degraded areas were more likely to change further, either by natural succession (abandonment of agricultural areas) or by reforestation with fast-growing tree species (*Pinus radiata, eucalyptus sp.*) (Figure 3.6f). Shrubland was one of the land cover categories that changed frequently, mainly because of its nonproductive importance, being either replaced by more intensive land cover classes or being given the opportunity to regrow triggered by the abandonment of farmlands (Figure 3.6e) (Carmona et al., 2010; Díaz et al., 2011; Izquierdo and Grau, 2009; Schulz et al., 2010). After 2008, shrubland gained area again, mainly from grassland/arable land. This transformation allowed us to recognize a possible pattern of land cover change associated with agricultural land abandonment, taking into account that the old forest cover area remains almost constant during the same period. The land cover dynamics detected in our study area support the idea that the area is in an initial state of forest transition (the shift from net deforestation to net reforestation). The continuous land cover information from our study provides evidence for this process during the last two study periods. Before that, the old-growth forest deforestation rate remained stable over time. This statement is based only on the development of old-growth forest and shrubland. We did not include exotic forest plantation in the assessment of a transition from net deforestation to reforestation as other studies do (Meyfroidt et al., 2010) because they are industrializing and intensely managed and do not provide the same ecological benefits as native forest ecosystems. Secondary forests were also not considered due to the highly fragmented distribution shown in our results. More research is needed to identify the ecological viability of these areas.

The decision to split different successional states (shrubland - secondary forest - old-growth forest) into separate land cover categories is a methodological challenge and adds an additional degree of uncertainty to the classification. But from our perspective the high - and in a few cases - moderate to high accuracies indicate a high-quality representation of the medium scale landscape patterns. An integration of the successional stages in a common forestry class potentially leads to a misinterpretation of pattern and process that took place in the area.

3.6 Conclusions

Our analysis shows the advantages to integrate multi-annual trajectories with automatized remote sensing techniques, contributing to the identification of historic transformations that led to a better understanding of current landscape configuration and composition. This high temporal resolution allowed us to identify strong relations between the different processes that occurred in the landscape (clear-cutting, regrowth, afforestation, reforestation, deforestation), confirming the dynamic transformation due to pressure convert them into more intensive land cover. Even though some landscape processes as deforestation are normally monitored with 5-10-year frequency, we want to highlight the potential to increase the temporal resolution of the landscape studies, especially in highly dynamic areas. These detailed trajectories provide crucial information that would otherwise be lost such as the peaks of conversion from native forest to exotic forest plantations. Our outcomes suggest important progress to conservation and landscape planning, considering the dense amount of Landsat information available and the opportunity to automatize change detection.

Differentiation between successional states (shrubland - secondary forest - old-growth forest) provided important information needed to identify spatial land cover dynamics. An aggregated analysis would fail to achieve important information from the perspective of ecosystem service provisioning and biodiversity. Spatial planning in the region needs to consider that remaining secondary forest patches play an important role in landscape connectivity, especially between the two largest remaining areas of old-growth forest in the Andes but also as step stones in areas dominated by exotic forest plantations. The integration of landscape research into local planning processes adds base knowledge to balance economic, social and environmental dimensions in the area. Future research should aim at assessing the value of different landscape parts for ecosystem service provisioning and will potentially help to prioritize forest areas for conservation under consideration of trade-offs with goods and services provided by exotic forest plantations.

3.7 Acknowledgments

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3.8 Supplementary Material

3 S Table 1 Confusion matrix for all periods

Classified data	Reference data								User's	Commision
	Urban	Grassland	Shrubland	Secondary forest	Old growth forest	Forest plantation	Bare soil	Water	accuracy (%)	error (%)
T1										
Urban	642600	8100	0	0	0	0	0	0	98.8	1.2
Grassland	4500	5121900	18900	35100	0	0	0	0	98.9	1.1
Shrubland	0	65700	716400	64800	0	900	0	0	84.5	15.5
Secondary forest	0	0	202500	1062900	429100	6300	0	0	62.5	37.5
Old growth forest	0	0	0	55800	16985700	258900	0	0	98.2	1.8
Forest plantation	0	0	0	8100	0	498600	0	0	98.4	1.6
Bare soil	108000	60300	0	0	0	0	5056200	0	96.8	3.2
Water	0	0		0	0	0	0	24280	100.0	0.0
Total	755100	5256000	937800	1226700	17414800	764700	5056200	24280		
Producer's accuracy (%)	85.1	97.4	76.4	86.6	97.5	65.2	100.0	100.0		
Omission error (%)	14.9	2.6	23.6	13.4	2.5	34.8	0.0	0.0		
Overall accuraccy (%)	95.8									
T2	Urban	Grassland	Shrubland	Secondary forest	Primary forest	Forest plantation	Bare soil	Water		
Urban	686700	0	0	0	0	0	0	0	100.0	0.0
Grassland	0	2881800	0	31500	0	0	0	0	98.9	1.1
Shrubland	0	3960	293400	104600	12000	0	0	0	70.9	29.1
Secondary forest	0	68400	148700	1371600	362700	12600	0	0	69.8	30.2
Old growth forest	0	0	8100	117900	19648800	657000	0	0	96.2	3.8
Forest plantation	0	0	0	0	0	5786100	0	0	100.0	0.0
Bare soil	21600	900	0	0	0	0	10964700	0	99.8	0.2
Water	0	0	0	0	0	0	0	20189	100.0	0.0
Total	708300	2955060	450200	1625600	20023500	6455700	10964700	20189		
Producer's accuracy (%)	97.0	97.5	65.2	84.4	98.1	89.6	100.0	100.0		
Omission error (%)	3.0	2.5	34.8	15.6	1.9	10.4	0.0	0.0		
Overall accuraccy (%)	96.4									
T3	Urban	Grassland	Shrubland	Secondary forest	Primary forest	Forest plantation	Bare soil	Water		
Urban	696600	0	0	0	0	0	6300	0	99.1	0.9
Grassland	0	3174300	2700	29700	0	0	19800	0	98.4	1.6
Shrubland	0	0	811800	72000	0	1800	93600	0	82.9	17.1
Secondary forest	0	0	279000	995400	12600	900	0	0	77.3	22.7
Old growth forest	0	0	126900	445900	11034000	1367100	0	0	85.0	15.0
Forest plantation	0	0	0	0	0	7105500	0	0	100.0	0.0
Bare soil	6300	28800	0	0	0	0	11098800	0	99.7	0.3
Water	0	0	0	0	0	0	0	243519	100.0	0.0
Total	702900	3203100	1220400	1543000	11046600	8475300	11218500	243519		
Producer's accuracy (%)	99.1	99.1	66.5	64.5	99.9	83.8	98.9	100.0		
Omission error (%)	0.9	0.9	33.5	35.5	0.1	16.2	1.1	0.0		
Overall accuraccy (%)	91.4									
T4	Urban	Grassland	Shrubland	Secondary forest	Primary forest	Forest plantation	Bare soil	Water		
Urban	726300	0		0	0	0	900	0	99.9	0.1
Grassland	2700	2836800	35100	21600	0	0	398700	0	86.1	13.9
Shrubland	0	118800	603900	97200	9900	0	33300	0	70.0	30.0
Secondary forest	0	55800	269100	1215900	182700	10530	0	0	70.1	29.9
Old growth forest	0	0	21600	219600	8809200	558000	0	0	91.7	8.3
Forest plantation	0	0	0	0	0	6786900	0	0	100.0	0.0
Bare soil	11700	259200		0	0	8100	15557400	0	98.2	1.8
Water	0	0		0	0	0	0	243289	100.0	0.0
Total	740700	3270600	929700	1554300	9001800	7363530	15990300	243289		
Producer's accuracy (%)	98.1	86.7	65.0	78.2	97.9	92.2	97.3	100.0		
Omission error (%)	1.9	13.3	35.0	21.8	2.1	7.8	2.7	0.0		
Overall accuraccy (%)	94.0									

Classified data	Reference data								User's	Commision
	Urban	Grassland	Shrubland	Secondary forest	Old growth forest	Forest plantation	Bare soil	Water	accuracy (%)	error (%)
T5	Urban	Grassland	Schrubland	Secondary forest	Primary forest	Forest plantation	Bare soil	Water		
Urban	847800	0	0	0	0	0	0	0	100.0	0.0
Grassland	1800	1918800	1800	6300	0	900	20700	0	98.4	1.6
Shrubland	0	229500	743400	112500	101700	8100	0	0	62.2	37.8
Secondary forest	0	0	25200	1246500	173700	90900	0	0	81.1	18.9
Old growth forest	0	0	8100	108000	8558100	636300	0	0	91.9	8.1
Forest plantation	0	0	0	4500	900	7862400	0	0	99.9	0.1
Bare soil	3600	0	0	0	0	0	9453600	0	100.0	0.0
Water	0	0	0	0	0	0	0	218673	100.0	0.0
Total	853200	2148300	778500	1477800	8834400	8598600	9474300	218673		
Producer's accuracy (%)	99.4	89.3	95.5	84.3	96.9	91.4	99.8	100.0		
Omission error (%)	0.6	10.7	4.5	15.7	3.1	8.6	0.2	0.0		
Overall accuraccy (%)	95.3									
T6	Urban	Grassland	Shrubland	Secondary forest	Primary forest	Forest plantation	Bare soil	Water		
Urban	739800	0		0	0	0	111600	0	86.9	13.1
Grassland	0	2979900	11700	4500	0	0	2700	0	99.4	0.6
Shrubland	0	12600	500400	148500	5400	8100	0	0	74.1	25.9
Secondary forest	0	36000	234000	1161000	7200	9000	0	0	80.2	19.8
Old growth forest	0	44100	7200	123300	8440200	811800	0	0	89.5	10.5
Forest plantation	0	0	0	900	0	8728200	0	0	100.0	0.0
Bare soil	0	5400		0	0	0	7418700	0	99.9	0.1
Water	0	0		0	0	0	0	230944	100.0	0.0
Total	739800	3078000	753300	1438200	8452800	9557100	7533000	230944		
Producer's accuracy (%)	100.0	96.8	66.4	80.7	99.9	91.3	98.5	100.0		
Omission error (%)	0.0	3.2	33.6	19.3	0.1	8.7	1.5	0.0		
Overall accuraccy (%)	95.0									
T7	Urban	Grassland	Shrubland	Secondary forest	Primary forest	Forest plantation	Bare soil	Water		
Urban	770400	0		0	0	0	25200	0	96.8	3.2
Grassland	0	3077100	14400	99000	0	0	0	0	96.4	3.6
Shrubland	0	54900	583200	144900	2700	0	0	0	74.2	25.8
Secondary forest	0	6300	181800	1188900	211500	18900	0	0	74.0	26.0
Old growth forest	0	0	54900	226800	9746100	1161000	0	0	87.1	12.9
Forest plantation	0	0	0	0	0	9055800	0	0	100.0	0.0
Bare soil	900	126900	0	0	0		8734500	0	98.6	1.4
Water	0	0	0	0	0	0	0	242553	100.0	0.0
Total	771300	3265200	834300	1659600	9960300	10235700	8759700	242553		
Producer's accuracy (%)	99.9	94.2	69.9	71.6	97.8	88.5	99.7	100.0		
Omission error (%)	0.1	5.8	30.1	28.4	2.2	11.5	0.3	0.0		
Overall accuraccy (%)	93.5									

3 S Table 2 Transition matrices between land cover in the different period (% of the study area)

Grassland/arable land	T1-T2	T2-T3	T3-T4	T4-T5	T5-T6	T6-T7
Shrubland	-0.705	-0.236	-1.085	1.001	-1.869	-0.814
Secondary forest	0.000	0.000	0.000	0.000	0.277	0.000
Old-growth forest	0.082	0.003	0.000	0.000	0.000	0.001
Forest exotic plantation	-0.725	-0.024	-0.023	0.002	-0.056	-0.044
Bare land	1.279	0.514	-1.333	1.171	-0.509	0.704

Shrubland	T1-T2	T2-T3	T3-T4	T4-T5	T5-T6	T6-T7
Grassland/arable land	0.705	0.236	1.085	-1.001	1.869	0.814
Secondary forest	-1.815	-3.602	-0.966	-0.702	0.185	-0.366
Old-growth forest	1.336	0.235	0.019	0.237	0.113	0.000
Forest exotic plantation	-0.767	-0.386	-0.670	-0.300	-0.051	-0.310
Bare land	1.121	-0.048	0.741	0.148	0.609	0.733

Secondary forest	T1-T2	T2-T3	T3-T4	T4-T5	T5-T6	T6-T7
Grassland/arable land	0.000	0.000	0.000	0.000	-0.477	0.000
Shrubland	1.815	3.602	0.966	0.702	-0.185	0.366
Old-growth forest	0.315	0.723	0.585	0.404	0.132	0.000
Forest exotic plantation	-0.940	-0.284	-0.743	-0.177	-0.188	0.000
Bare land	-0.537	-0.065	0.000	0.000	-0.433	-0.240

Old-growth forest	T1-T2	T2-T3	T3-T4	T4-T5	T5-T6	T6-T7
Grassland/arable land	-0.082	-0.003	0.000	0.000	0.000	-0.001
Shrubland	-1.336	-0.235	-0.019	-0.237	-0.113	0.000
Secondary forest	-0.315	-0.723	-0.585	-0.404	-0.132	0.000
Forest exotic plantation	-0.186	-0.004	-0.016	-0.030	0.000	0.000
Bare land	-0.269	-0.104	0.000	0.000	0.000	-0.001

Forest exotic plantation	T1-T2	T2-T3	T3-T4	T4-T5	T5-T6	T6-T7
Grassland/arable land	0.725	0.024	0.023	-0.002	0.056	0.044
Shrubland	0.767	0.386	0.670	0.300	0.051	0.310
Secondary forest	0.940	0.284	0.743	0.177	0.188	0.000
Old-growth forest	0.186	0.004	0.016	0.030	0.000	0.000
Bare land	0.329	-0.039	-0.043	-0.088	-0.551	-0.328

Bare land	T1-T2	T2-T3	T3-T4	T4-T5	T5-T6	T6-T7
Grassland/arable land	-1.279	-0.514	1.333	-1.171	0.509	-0.704
Shrubland	-1.121	0.048	-0.741	-0.148	-0.609	-0.733
Secondary forest	0.537	0.065	0.000	0.000	0.433	0.240
Old-growth forest	0.269	0.104	0.000	0.000	0.000	0.001
Forest exotic plantation	-0.329	0.039	0.043	0.088	0.551	0.328

4 Spatio-temporal change of ecosystem services as a key to understanding natural resource utilization in Southern Chile.

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4.1 Abstract

The understanding of how ecosystem services are distributed across the landscape and their change over time provides key information to manage multifunctional landscapes. To balance the conflicting demands on multiscale land assessments are highly relevant, especially in biodiversity hotspot areas as the Valdivian temperate rain forest. We quantified six ecosystem services linked to forest ecosystems over six temporal periods (1985 - 2011): three regulating (carbon storage, sediment retention, phosphorous retention), one provisioning (plantation site productivity) and two cultural services (landscape aesthetics, forest recreation). The study area is divided into four geomorphological units (Coastal mountain range, Central Valley, Pre-Andean and Andes mountain range). Our results show a high spatial and temporal variability of ecosystem service supply in these units. We observed a strong increase of plantation production (Coastal range and Central Valley) as well as of forest recreation services over time (Coastal and Andes ranges); remaining services trends varied across units and time. Recommendations for landscape management are: (i) an increase of buffer strips to reduce diffuse emissions into the river network and to enhance ecological connectivity, (ii) an increase of protected areas in the Central Valley and (iii) a rethinking of the role of exotic forest plantations.

Keywords
Landscape multi-functionality, spatio-temporal analysis, mapping of ecosystem services, deforestation, exotic forest plantations, land use change, South America.

4.2 Introduction

Land use change and especially deforestation decrease the ability of many ecosystems to supply services, which are the base to support human needs and well-being (Bennett et al., 2009; Díaz et al., 2006; Millennium Ecosystem Assessment, 2005). The high pressure of natural resource utilization has led to important changes in ecosystem functions and processes at different scales

(Costanza et al., 1997; Ellis et al., 2013; Foley et al., 2011). As many ecosystem services may rely on the same ecosystem process external factors might affect several ecosystem services at the same time (Bennett et al., 2009; Mouchet et al., 2015). Hence, mapping and monitoring ecosystem services dynamics at different scales plays an important role in landscape management and spatial planning, since it aggregates complex information about the effect of ecosystem service utilization on the supply of the services (Dallimer et al., 2015; Renard et al., 2015; Schröter et al., 2015). Several studies have shown that an expansion of the temporal and spatial extent and of the resolution of ecosystem services assessment allows a better representation of the heterogeneity in ecosystem services supply (Birkhofer et al., 2015; Lautenbach et al., 2011; Rodríguez et al., 2006; Syrbe and Walz, 2012). Furthermore, it delivers information about the magnitude and recurrence interval at which ecosystem service supply changes over time which is crucial for landscape planning, environmental management and decision-making (Burkhard et al., 2014; Daily et al., 2009; de Groot et al., 2012; Haines-Young, 2009; Millennium Ecosystem Assessment, 2005; Seppelt et al., 2011).

Land use change processes in Southern Chile – a region recognized as a biodiversity hotspot (Myers et al., 2000; Olson et al., 2001) - deserve special attention: Despite its relevance as a relic of temperate rain forest, the region has experienced a continuous process of land cover transformation and fragmentation (Donoso and Lara, 1999; Echeverría et al., 2012, 2007; Locher-Krause et al., 2017b). These transformations and fragmentation processes have led to an increase in biodiversity loss and ecosystem degradation, which is strongly linked to the fast-growing export-oriented Chilean economy (Balvanera et al., 2012; Díaz et al., 2006; Siebert, 2003; UNEP, 2010). Large landscape transformations in the area date back to forest clearance; first due to the colonization process (16th to 17th century) and second as a result of the boom of wheat crops for domestic use and export in the middle of the 20th century (Armesto et al., 2010; Echeverria et al., 2006). Since the 1970’s exotic forest plantations have been playing an important role in the regional and national economy due to their importance for the forest industry (i.e., the pulp, shipyard, and paper industry). These activities have been recognized as one of the most important drivers of deforestation and biodiversity loss in Southern Chile (Lara et al., 2011; Nahuelhual et al., 2012), with range of annual rate of forest loss between 0 and 5.8% (Miranda et al., 2017) and a net loss of 4.8% for our study area (Locher-Krause et al., 2017b). The forest industry has thereby focused mainly on the production of fast-growing exotic species such as *Pinus radiata* and *Eucalyptus* sp. (Echeverría et al., 2007; Lara and Veblen, 1993), causing enormous pressure on the native ecosystems.

Especially in highly transformed landscapes such as Southern Chile ecosystem services assessments considering several services in a multi-scale approach are highly relevant to understand and implement conservation and management strategies (Echeverria et al., 2006c; Locher-Krause et al., 2017b; Miranda et al., 2015). If important services are missing negative impacts of land use changes such as an increase of forest plantations for the paper industry would be neglected in the analysis. Since land use change, and its effects on ecosystem service supply, is in many cases not unidirectional, assessment and mapping of ecosystem service supply at an adequate temporal resolution is highly relevant. In southern Chile, the clearcut-regrowth cycle of forestry is an important process that leads to a cyclic change of land cover (Patterson and Hoalst-Pullen, 2011). Assessment of ecosystem services at a low temporal resolution might under- or overestimate trends since they rely on a linear interpolation between two or three sample periods.

However, multi-scale mapping and monitoring are still far from being integrated regularly in ecosystem services assessments wide, and this gap is even larger in Latin America (Balvanera et al., 2012). In Chile, existing studies regarding ecosystem services supply have been focused mainly on the effects of forest management on water provision and soil loss and assessed only one or two ecosystem services in combination (Lara et al., 2009; Little et al., 2014, 2009). Núñez et al. (2006) used production functions to evaluate changes in water availability in the context of forest management and conservation. Nahuelhual et al. (2013) mapped the cultural ecosystem services recreation and ecotourism at Chiloe Island to allow their integration in local planning. Even considering the importance of these studies the fact of relying on a comparison of two or three periods might lead to a misperception about the magnitude of change.

We seek to fill this knowledge gap on ecosystem services supply dynamics in the region. We are thereby providing essential information for spatial planning in the region, especially for areas under threat such as the Valdivian temperate rainforest (Myers et al., 2000; Olson et al., 2001). Our study addresses the following research questions: a) how were the supplies of individual ecosystem services distributed across the landscape? b) how did the spatial distribution of individual ecosystem services supply change over time, c) what are the implications for the ecosystem and landscape management seeking to balance natural resources utilization and conservation in the region. Our analysis builds on an integration of remote sensing derived land cover data with spatially explicit models, administrative statistics and field measurements. Based on this information we mapped and analyzed the spatio-temporal changes in ecosystem services supply.

4.3 Material and Methods

4.3.1 Study area

The study area is located in southern Chile (Northern Chilean Patagonia - 73°20’ W-39°25’ S - 71°59’ W-41° 14’ S) and covers 16,625.7 km², belonging administratively to the Los Rios and Los Lagos regions (c.f. Figure 4.1). This area is home to 2.2% of the population of Chile, circa 380.700 inhabitants, which live mainly in urban areas (68.3%) (INE Instituto Nacional de Estadística, 2012). Around 17% of the inhabitants describe themselves as Mapuches, Chilean indigenous inhabitants. The annual mean temperature is around 11.9 ºC, and rainfall is about 2,500 mm per year, concentrated during the winter season (June-September), with a temperate oceanic climate with Mediterranean influence (CIREN, 1994; Di Castri and Hajek, 1976).

The area is recognized as biodiversity hotspot due to the number of species associated with the temperate rain forest (Armesto et al., 1996; Myers et al., 2000; Olson et al., 2001). This temperate broadleaf and mixed forest is subdivided into four vegetation zones: deciduous forests, Valdivian laurel-leaved forests, Northern Patagonian and Evergreen forests (Gajardo, 1994; Veblen et al., 1983). The area is divided into four geomorphological units from the West to the East: the Coastal mountain range (up to 900 m), the Central Valley (up to 250 m), the Precordillera (up to 1,000 m) and the Andes mountain range (up to 2,422 m). These units show a different degree of human utilization and degradation: the majority of the agricultural areas are located in the Central Valley while the Valdivian temperate rain forest is mainly distributed across the Andes and the coastal range.

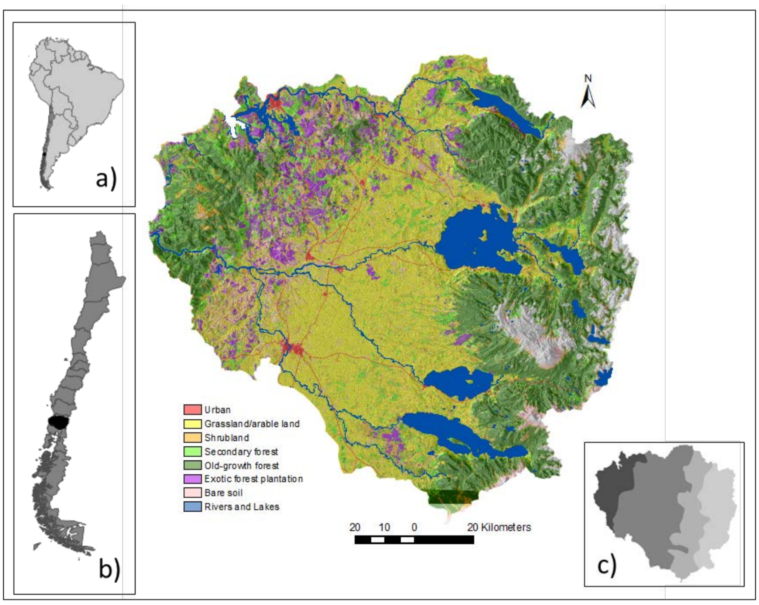


Figure 4.1 Study area location in South America (a) and Southern Chile (b). The map shows land cover information for 2011 as well as the four most important geomorphological units(c): Coastal mountain range, Central Valley, Pre-Andean and Andes mountain range (from west to east); shown in the inset to the lower right.

4.3.2 Methods

We selected six ecosystem services for our assessment based on the following criteria: (1) the service was related to managed or unmanaged forest ecosystems (natural or planted), (2) the service had a recognized importance for the case study region, we based this statement on the most documented and critical ecosystem services for the mid-southern Chile (Balvanera et al., 2012; Lara et al., 2009; Little et al., 2014; Laura Nahuelhual et al., 2013a; Oyarzún and Hervé-Fernandez, 2015); and (3) data to estimate the service indicators were available in the desired spatial and temporal resolution, extent and scale. The following six services were selected: three regulating (carbon storage, sediment, and phosphorous retention), one provisioning (plantation production index) and two cultural (landscape aesthetic and forest recreation) services. The calculation and mapping of the services were based on different InVEST 3.2.0 (Integrated Valuation of Environmental Services and Tradeoffs, www.naturalcapital.org) models (Sharp et al., 2015), productivity functions, field data, and biophysical data. After the ecosystem services mapping, we quantified the spatio-temporal change of the selected ecosystem services. Additionally, we analyzed the relationship between ecosystem services and the land cover classes in the case study area.

4.3.2.1 Ecosystem services mapping

To map and assess the ecosystem services remote sensing data, model results and field data were integrated at a 30mx30m resolution for six periods in time. Land cover information was obtained from Landsat satellite images (L4-5 TM, L7 ETM+ on; 30 m x 30 m). We derived a time series of land cover data from 1985 to 2011 based on all the images available for the study area with L1T processing level (USGS). These scenes were atmospherically corrected using radiative transfer based Landsat Ecosystem Disturbance Adaptive Processing System tool (LEDAPS) (Kaufman et al., 1997; J. G. Masek et al., 2006). Clouds and cloud shadows were masked by the two-step algorithm Fmask to improve the levels of accuracy, avoiding misclassification in the time series (Zhu et al., 2012). Land cover was classified by a random forest classifier (Breiman, 2001) implemented in R (Liaw and Wiener, 2002; R Development Core Team, 2013), using a total of 800 control ground polygons to train the model and 260 to validate the result of the classification. The classifier was trained and applied to nine land cover categories: urban, grassland/arable land, shrubland, secondary forest, old-growth forest, exotic forest plantation, bare land, water bodies and areas without information (4 S-A Table 1). To obtain land cover maps not affected by the frequent cloud cover in the region, we merged land cover information from two consecutive years. This resulted in seven combined land cover maps for the following years: T1=1985-1986; T2=1998-1999; T3=2000-2001; T4=2003-2004; T5=2005-2006; T6=2008-2009; T7=2010-2011 (4 S-B Table 1). More details on the analysis can be found in Locher-Krause et al. (2017).

Table 4.1 summarizes the selected ecosystem service, units and the biophysical indicator or model used to quantify it for each period for which land cover information was available.

Table 4.1 Ecosystem services, biophysical indicator or model and units for the six ecosystem services mapped and quantified for the six periods (1985-2011) for each geomorphological unit.

Selected Ecosystem service	Biophysical indicator/model	Unit
Provisioning services		
Plantation site productivity	Site productivity index	Index
Regulation services		
Carbon storage	InVEST carbon storage model	Mg/ha
Sediment retention	InVEST sediment delivery ratio model	Mg/ha
Phosphorous retention	InVEST nutrient retention model	Kg/ha
Cultural services		
Landscape aesthetics	Areas high aesthetic Value/recreational interest points/degree of naturalness	Degree of naturalness
Forest recreation	Forest protected areas	Area %

4.3.2.1.1 Provisioning services

4.3.2.1.1.1 Plantation site productivity

The estimation of plantation productivity was based on a plantation production site index. The site index is a quantitative measure of the potential of an area to produce plant biomass, in terms of the capacity of a particular tree species to produce aboveground timber volume under specific soil and climatic conditions (Skovsgaard and Vanclay, 2008). The index integrates information regarding the specific species as well as physical and climatic characteristics data (soil depth, texture, nutrient load, precipitation, temperature, slope, elevation, and aspect) to estimate the potential growth rate of each species (Nyland, 2002). The plantation production site index was obtained from the National Forest Research Institute (García, 1970; INFOR, 1999; Pinilla S., 1998; Schlatter and Gerding, 1995). The national forest research agency developed the index from an extensive (national level) and intensive (frequency of measurement) forest inventory. Based on the inventory results a regression model was used to derive a consistent index based on an empirical height - age relationship in predefined growing zones (stratified based on homogenous soil and climatic zones area characteristics) - more information is provided in the Supplementary material S1. The index was index especially developed for the most relevant economic tree species used in forest plantations (*Pinus radiata* and *Eucalyptus* sp). Through the site productivity index, we mapped and assessed the spatial variation of the productivity in the study area over time.

4.3.2.1.2 Regulation services

4.3.2.1.2.1 Carbon storage (Mg/ha)

The amount of carbon stored in the landscape was estimated with the InVEST 3.2.0 carbon module[1]. The module is based on a simplified carbon cycle, which integrates land cover information and the

[1] http://www.naturalcapitalproject.org/invest-releases/documentation/3_2_0/carbonstorage

amount of carbon stored on four main carbon pools: aboveground biomass, belowground biomass, soil, and dead organic matter. We calculated the carbon stored in 30 m x 30 m cells based on land cover data together with estimates for the four major carbon pools. Values for the major carbon pools were obtained from the region and species-specific allometric equations developed by local studies (4 S-A Table 2). These equations were developed on the base of a carbon inventory to estimate the carbon stocks for the main native species by destructive sampling (tree harvesting) and tree roots extraction (A and C, 2005; Gayoso, 2001; Gayoso and Schlegel, 2003; Gonzalez and Gayoso, 2005; Keith et al., 2009; Schlegel, 2001; Schlegel and Donoso, 2008) (4 S-A Table 2).

4.3.2.1.2.2 Sediment retention (Mg/ha)

We quantified sediment export and retention with the InVEST 3.2.0 sediment delivery ratio module[2]. The module is based on the Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1978) and uses geomorphological and climatic information together with information on the hydrological connectivity of the landscape (Hamel et al., 2015). The main inputs are land cover, a digital elevation model (DEM) and the USLE parameters (R, K, LS, C, and S, see explanation below). We used the digital elevation model by Lehner et al. (2008) (3 sec), which we resampled to 30 m x 30 m resolution. The rainfall erosivity R (MJ mm ha^{-1} yr^{-1}) was calculated by the following formula for locations with mean annual precipitation greater than 850 mm (Renard and Freimund, 1994) applicable for the case study region:

$$R = 587.8 - 1.219P + 0.004105\, P^2$$

where P is the annual precipitation in mm, which we obtained from the WorldClim database (Hijmans et al., 2005). We estimated the soil erodibility parameter K (tons ha^{-1} MJ^{-1} mm^{-1}) based on the HWSD (FAO/IIASA/ISRIC/ISSCAS/JRC, 2012) and a local soils database (SINIA, 2015). The land cover management factor (C) and the supporting practice factor (S) for the different land cover categories were obtained from Bonilla et al. (2010) and Olivares et al. (2011). We validated the simulation results against suspended sediment concentration data from the Chilean monitoring agency gauging stations in the study area (DGA-SINIA) (4 S-A Table 3).

4.3.2.1.2.3 Phosphorous retention (tons/ha)

We estimated phosphorus export and retention with InVEST 3.2.0, nutrient retention module[3]. This model calculates export and retention capacity per pixel based on biophysical variables. The phosphorous retention module is built on the assumption that phosphorous retention depends on the potential inflow of phosphorus. Based on this, the module assumes that forest areas around streams provide much more retention than large forested continuous upstream areas (Sharp et al., 2015).

[2] http://www.naturalcapitalproject.org/invest-releases/documentation/3_2_0/sdr
[3] http://www.naturalcapitalproject.org/invest-releases/documentation/3_2_0/waterpurification

The same land cover, annual precipitation, and DEM data as used for modeling sediment retention were used for phosphorus retention modeling. Additionally, annual average reference evapotranspiration, obtained from Mu et al. (2011, 2007) and Zomer et al. (2008, 2007), as well as information on maximal root depth and plant available water fraction, obtained from FAO/IIASA/ISRIC/ISSCAS/JRC, (2012) were used. We applied region specific parameter values for the evapotranspiration index, root depth, phosphorus export and phosphorus retention efficiency in each land cover class (values taken from Cuevas et al., 2006; Donoso and Lara, 1999; Jackson et al., 1996; Little et al., 2014; Oyarzun et al., 2007, 2015) (4 S-A Table 4 and 4 S-A Table 5).

To check the performance of the model, we contrasted predicted values with literature values (Alvarez-Cobelas et al., 2009; Cuevas et al., 2006; Huygens et al., 2011; Iroume, 2003; Little et al., 2014; Oyarzun et al., 2015, 1997, Oyarzún et al., 2007, 2004) due to a lack of applicable phosphorous concentration measurements in the study area.

4.3.2.1.3 Cultural services

4.3.2.1.3.1 Landscape aesthetics (degree of naturalness)

Landscape aesthetics was estimated in three steps. First, we selected areas with high aesthetic value based on a viewpoint database of the tourist agency (SERNATUR and GORE Region de los Rios, 2014). In a second step, this dataset was augmented with information on recreational use based on the number of geo-tagged digital images in a cell. Flickr (www.flickr.com), Google Earth (earth.google.com) and Panoramio (www.panoramio.com) were used for that analysis. This procedure – using images from these hosting websites - has been reported to provide good proxies for recreational interest points (Grêt-Regamey et al., 2014; Martínez Pastur et al., 2016; Laura Nahuelhual et al., 2013a; Sharp et al., 2015). In a third step, the visible area around each viewpoint was derived by a viewshed analysis performed in ArcGIS (ESRI, 2011) using the digital elevation model described above as auxiliary information. In order to understand how much valuable recreational areas were affected by human activities, we included information on the degree of naturalness based on land cover information (Machado, 2004; Walz and Stein, 2014) classified into artificial, semi-natural and natural landscapes (4 S-A Table 6). This last step was used to determine how strong the visual qualities of the landscape were influenced by human utilization over time.

4.3.2.1.3.2 Forest recreation (protected area in each geomorphological unit)

In Chile, it is not allowed to enter natural private areas without permission by the owner. Protected areas provide therefore the only opportunity for forest recreation activities such as outdoor sports activities or birdwatching. Our estimation of the forest recreation service was therefore based on the opportunity to access and use forest areas which was estimated by the area covered by protected forest. This proxy for forest recreation has already been used in other studies (Qiu and Turner, 2013; Raudsepp-Hearne et al., 2010).

We used all areas that were registered in the protected areas national database: national parks, national reserves, national monuments as well as private protected areas. Historic and current data on the development of the protected area network were available from the National Environmental Information System (SINIA). The number of visitors could not be taken into account since no long-term information was available for the region.

4.3.2.2 Spatio-temporal changes

We identified and quantified the supply of the six individual ecosystem services over 26 years (1985 to 2011) for six periods in the four geomorphological units. We calculated the arithmetic mean for each ecosystem service in each geomorphological unit and period to identify the changes in their individual supply. We furthermore calculated crosstables for the supply of each ecosystem service for each geomorphological unit and period against the different land cover classes. This allowed us to follow the importance of the different land cover classes for each service and unit over time. This information is the base towards establishing a connection between the ecosystem services supply trajectory and economic/political decisions in the study area.

4.4 Results and discussion

Ecosystem services supply differed considerably over the six periods analyzed, both across time and across the geomorphological units (Figure 4.2). Plantation production increased strongly over time in two geomorphological regions (with a relative change of 377 % and 917% for the Coastal range and Central Valley, respectively). Forest recreation showed the highest relative increase over time in two of the four regions (Figure 4.2). Sediment retention peaked in 2000-2001 followed by a decrease. The speed and magnitude of changes in the mean supply of individual ecosystem services differed both over time and across the geomorphological units.

4.4.1 Spatio-temporal analysis of individual ecosystem services

Trends in carbon stocks differed markedly across the geomorphological units. The Andes mountain range, with the largest relict of old-growth forest, always stored the highest total amount of carbon. Carbon stocks in the Andes mountain range showed a slight decrease by 2.1% from 1985 to 2011 while for the Coastal and Pre-Andean areas the values decreased by 6.4% and 3.1% per year respectively (Figure 4.2). The reduction of carbon stocks in the Coastal and Pre-Andean areas was linked to the intense deforestation process that is also reported by other studies (Echeverria et al., 2006c; Locher-Krause et al., 2017b; Miranda et al., 2017; Zamorano-Elgueta et al., 2015). An opposite trend was observed in the Central Valley, where carbon stocks increased on average by 6.2% between 1985 and 2011, but also showed an uneven pattern over time. This increase can be explained by the rapid development of the forest industry and the related increase of exotic forest plantations (*Pinus radiata* and *Eucalyptus* sp.) as of one of the main economic activities in the study area - as reported by Armesto et al. (2010), Lara et al. (2011) and Nahuelhual et al. (2012). The uneven pattern over time can be explained on the one hand by a continuous increase in forest cover over the first two periods from 1985 to 2001 due to the ongoing establishment of exotic forest plantations (artificial afforestation) (4 S-B Table 1). Afterward, carbon stocks in the Central Valley fluctuated due to the harvesting practices for exotic forest plantations which consist of clear-cuts every 10 to 20 years.

The plantation productivity index showed the highest service values in the Coastal range and the Central Valley (Figure 4.2b). The index increased strongly particularly in these units, with a relative change of 370% and 900% between 1985 and 2011, respectively. This increase is clearly continuous until 2005-2006 and when levels off (last two periods). In the Andes and Pre-Andean range the same trend is visible - less pronounced but nevertheless important, suggesting that conversion of natural forests to exotic forest plantations still play a role in the area (Figure 4.2, 4 S-B Table 1) (Locher-Krause et al., 2017b). Our results were in line with high rates of land cover transformation to exotic forest plantations reported by different studies in the south of Chile, all in the context of the rapid development of the forest industry (Echeverria et al., 2006c; Echeverría et al., 2012; Miranda et al., 2015; Nahuelhual et al., 2012). This is also supported by results from local studies, such as by Zamorano-Elgueta et al. (2015) which reported an increment of 168% of the area covered by exotic forest plantations for a smaller region in the Coastal. This large and rapid transformation to exotic forest plantations has been explained by the success of the Chilean forest policy, especially by a subsidy passed by the military regime in 1975, the Law Decree 701 (Armesto et al., 2010; Lara and Veblen, 1993; Niklitschek, 2007). The development of the forest industry has been crucial for the economic growth of Chile, leading to the second largest export commodities after the mining sector (Central Bank of Chile, 2016). However, exotic forest plantations have long been under scientific and public scrutiny due to their environmental and socioeconomic impacts (Andersson et al., 2016; Salas et al., 2016). Global and local studies in different ecosystems have already reported the effect of fast-growing trees that leads to the decrease of water yields, especially related to eucalyptus species (Albaugh et al., 2013; Farley et al., 2005; Huber et al., 2008; Jackson et al., 2005). Little et al. (2009) documented a decreasing trend of summer runoff in landscapes dominated by *Pinus radiate* plantations compared to native forest in South-Central Chile.

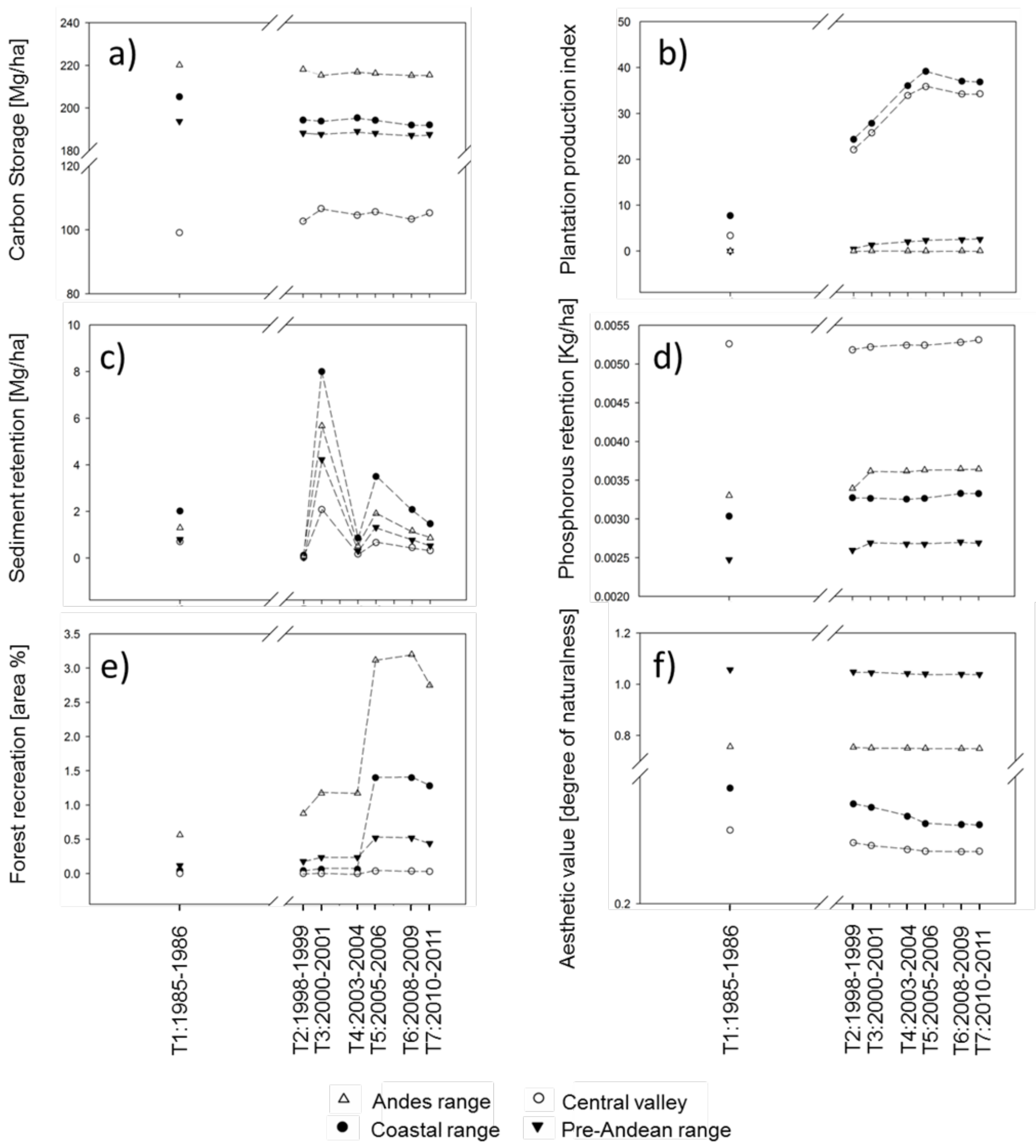


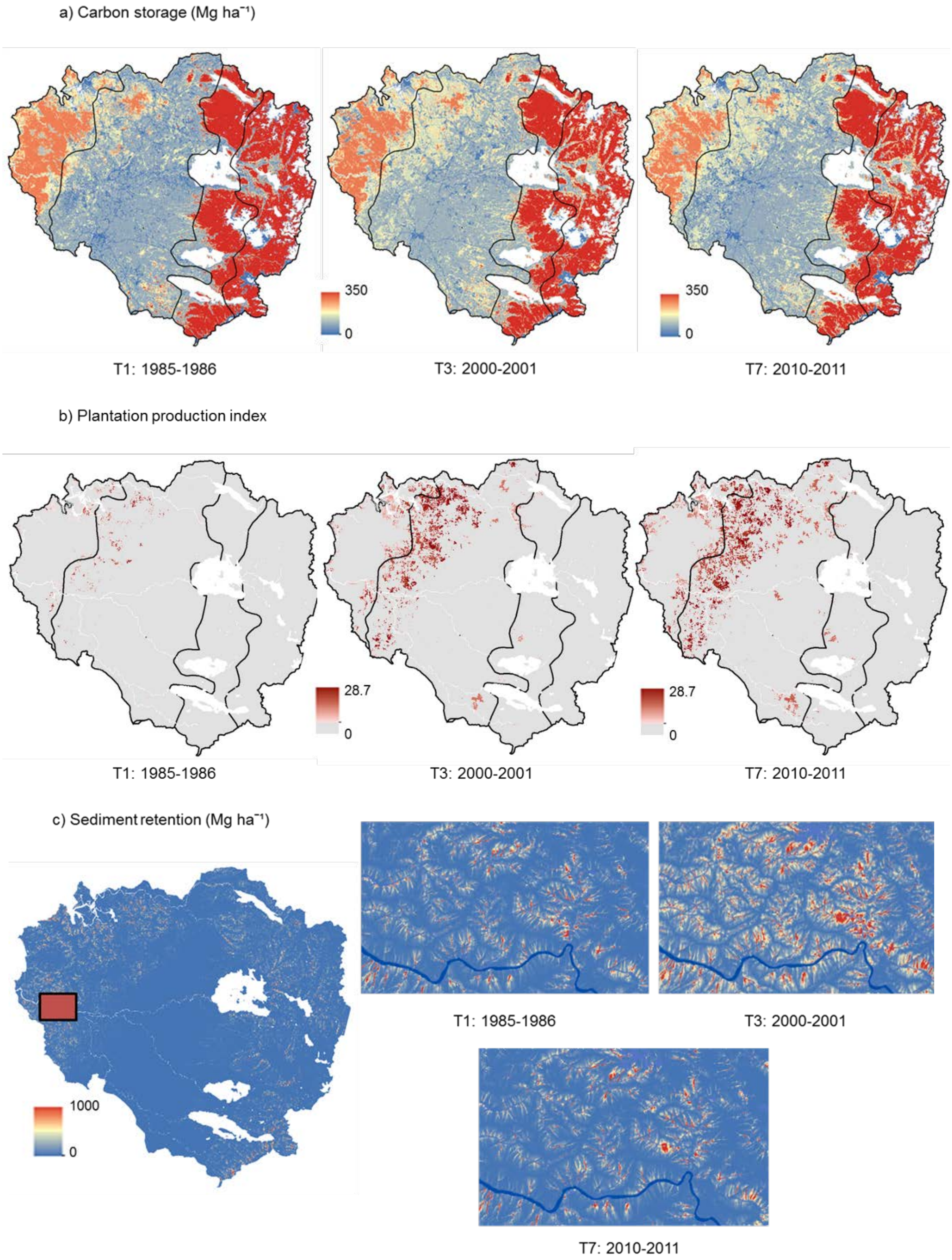
Figure 4.2 Change in the supply of six ecosystem services over time. The values represent their arithmetic mean values for each geomorphological unit in the study area (Coastal range, Central Valley, Pre-Andean range and Andes range).

Overall sediment retention (Figure 4.2c) decreased between 1985 and 2011 by 27%, 53%, 36% and 34% for the Coastal range, Central Valley, Pre-Andean and Andes range, respectively. However, development has been highly dynamic. The development of sediment retention is characterized by a strong peak in 2000-2001 followed by a strong decrease and a smaller peak in 2005-2006. Sediment transportation and thereby sediment retention as well is influenced by strong rainfall events and differs therefore between the years the rivers especially for pluvial flow regimes such as in the case study region (Oyarzún and Hervé-Fernandez, 2015). This highlights the importance of a multi-temporal analysis of ecosystem service provisioning in the case study region. Areas covered by dense vegetation - especially natural forest and exotic plantations - had the highest retention values (Figure 4.3), particularly important in areas with high elevation and steep slopes. Land use change led to a decrease in sediment retention, particularly caused by the transition from tree-based vegetation systems to more open land cover as shrubland, grassland and bare land (Figure 4.4). The importance of maintaining a continuous vegetation cover for erosion control has been already reported in several studies (Lara et al., 2009; León-Muñoz et al., 2013; Little et al., 2009), with the highest relevance in areas with high pluviometry and steep slopes, as in our study area (Bathurst et al., 2011; Cuevas et al., 2006). Furthermore, the combination of medium to high deforestation rates in the region (Locher-Krause et al., 2017b; Zamorano-Elgueta et al., 2015) with the large clear-cuts used in regular exotic forest plantation management led to the decrease in the ability of the ecosystem to retain sediment (Figure 4.3). The lowest retention values were located in areas with low vegetation cover.

For phosphorous retention, the highest values were estimated in the Central Valley especially in downstream areas close to water bodies. The retention values of these areas increased from the beginning of the study period, with the exception of the Central Valley that showed a slight decrease. The lowest phosphorous retention values were reported for the Pre-Andean areas. As high values of phosphorus retention are associated with smaller vegetation patches downstream near the water bodies, the low retention in the Central Valley (Figure 4.3d) is related to the high landscape fragmentation in this region (Locher-Krause et al. 2017). Overall the decrease in phosphorus retention indicates an increasing fragmentation of the landscape which also has been reported in other studies in Southern Chile (Echeverría et al., 2007; Locher-Krause et al., 2017b; Miranda et al., 2017). This fragmentation process might lead to a strong and non-linear effect on ecosystem services supply (both negative and positive), increasing the complexity of the ecological system (Andrieu et al., 2015; Mitchell et al., 2014). While this service is similar to sediment retention driven by strong precipitation events strong peaks such as for sediment retention were missing. This is due to the location of arable fields on flat areas downstream as the main source of diffuse phosphorus retention.

Landscape aesthetic values showed an important decrease over time in the Coastal range and Central Valley, especially from 1985 to 2006. Presumably, the decrease of landscape aesthetics in these areas (Coastal range and the Central Valley) (Figure 4.3e) was related to the increase of roads, urban areas and other artificial landscape elements in the areas with higher population density. In the Andes and Pre-Andean, the values remained more and less constant over the study period, indicating a lower impact of human activities (Figure 4.2).

Protected areas in Chile were historically established in non-productive remote areas with high scenic beauty (Armesto et al., 2010). Over time, recreational forest areas showed a drastic increase especially between 2003-2004 and 2005-2006 (Figure 4.2 and Figure 4.4). This increase reflects an arising concern about conservation in the region, triggered by NGO’s, universities and the local government. However, as Figure 4.3 indicates, protected areas were unevenly distributed across the different geomorphological units: these areas were mainly located in remote locations with low accessibility in the Andes and Coastal range instead of the more productive areas in the Central Valley. The Central Valley had no protected areas until 2005, when private protected areas were introduced due to the cooperation between public institutions and private associations, leading to an increase of 283 km^2.



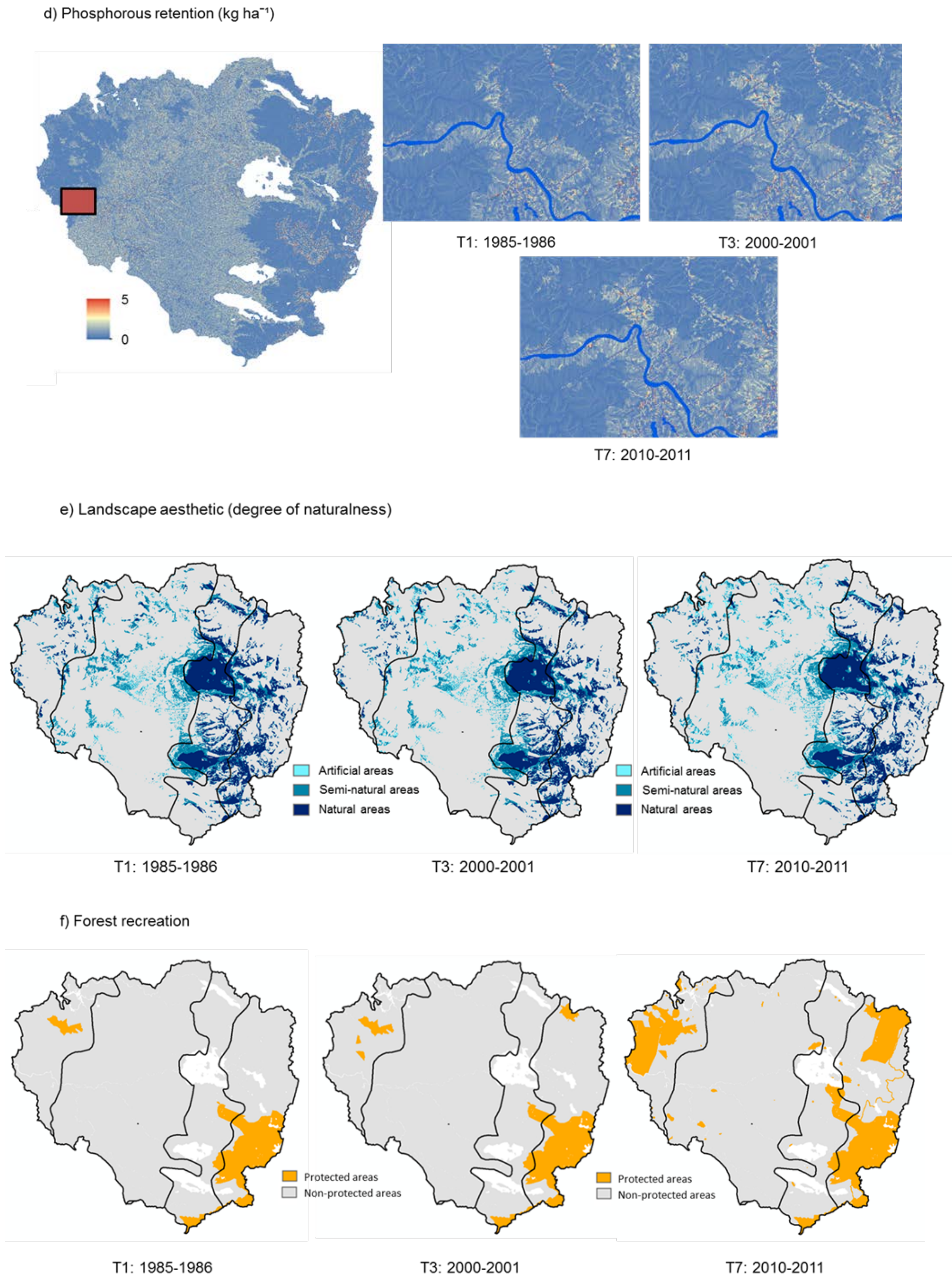


Figure 4.3 Map of selected ecosystem services in three-time periods (T1: 1985-1986, T4:2003-2004, T7:2010-2011). Only three-time slots are depicted due to space restriction; all the maps are available in the supplementary material S2). For phosphorous and sediment retention results are shown for a zoom-in area. The location of the zoom-in area is shown in the main figure (The zoom-in area has been placed in a region with highest service values/high variability of service values, located in between the coastal range and the central valley).

4.4.2 Effects of land use change in space and time

The importance of considering landscape heterogeneity in landscape planning and management has been pointed out by Lambin et al., (2001) and Turner et al., (2012). In our case study region, geomorphological units strongly differ in soil and climatic properties, which also influence vegetation composition and diversity. Based on these differences, we would like to highlight the importance of considering these different characteristics in an adequate landscape planning and management.

Figure 4.3 shows that regulating services fluctuated in time and space overall geomorphological units. This pattern was closely related to the variation in land cover composition (Figure 4.4) (4 S-B Table 1). Changes in carbon stocks for example clearly differed among geomorphological units and over time, especially in the Coastal range and the central valley (relative change from 1985/86 to 2010/11 of 6.4% and 6.2%, respectively). In the case of the plantation production index (Figure 4.3b), dissimilar productivity rates were identified due to the different climatic, soil and species characteristics. High productivity areas were mostly located in the Central Valley, mainly caused by deeper soils in addition to soil textures and climatic conditions that are more favorable for the development of *Pinus radiata*. Landscape aesthetics – as a cultural ecosystem service – also showed an uneven distribution across the different geomorphological units. Higher aesthetic values were mainly identified next to water bodies (lakes, rivers and the ocean) together with areas surrounded by volcanoes, especially in the Andes and Pre-Andean mountain range areas (Figure 4.3).

Overall, the relationship between the cover classes and the selected ecosystem services varied due to land use change (Figure 4.4). However, the sensitivity of the landscape towards land use change differed in space. The same amount of land cover change triggered different changes in ecosystem services supply in the four geomorphological units (Figure 4.4). Changes in carbon stocks were mainly linked to old-growth forest in the Andes ranges which provided more than 52% of the total changes in carbon stocks in 1985-1986. The importance of the old-growth forest in the Andes ranges for carbon storage decreased over time to 46% in 2010-2011. Old-growth forest in the Coastal range provided 15% of the changes in carbon stocks in 1985-1986. The contribution of changes in carbon stocks by the Coastal range decreased by about 11% in 2010-2011. In the case of the plantation productivity index, exotic forest plantation is the only land cover integrated into this index.

Shrubland and the Andes old-growth forest provided the highest sediment retention supply. The importance of these land cover classes slightly decreased over time in contrast to secondary forest and exotic forest plantation that increased in area over the study period. For phosphorous retention grassland/arable land were of most importance, followed by shrubland.

The Andean old-growth forest and water bodies provided the largest share of landscape aesthetic value for all points in time. Forest recreational services were mainly provided by protected forest areas. Other land cover classes such as water areas inside the protected areas also contributed to the provisioning – however, their contribution was relatively low. This is shown in Figure 4.4, were the importance of this service is reflected across the different forest classes, also important for shrubland and bare land especially in high mountainous areas.

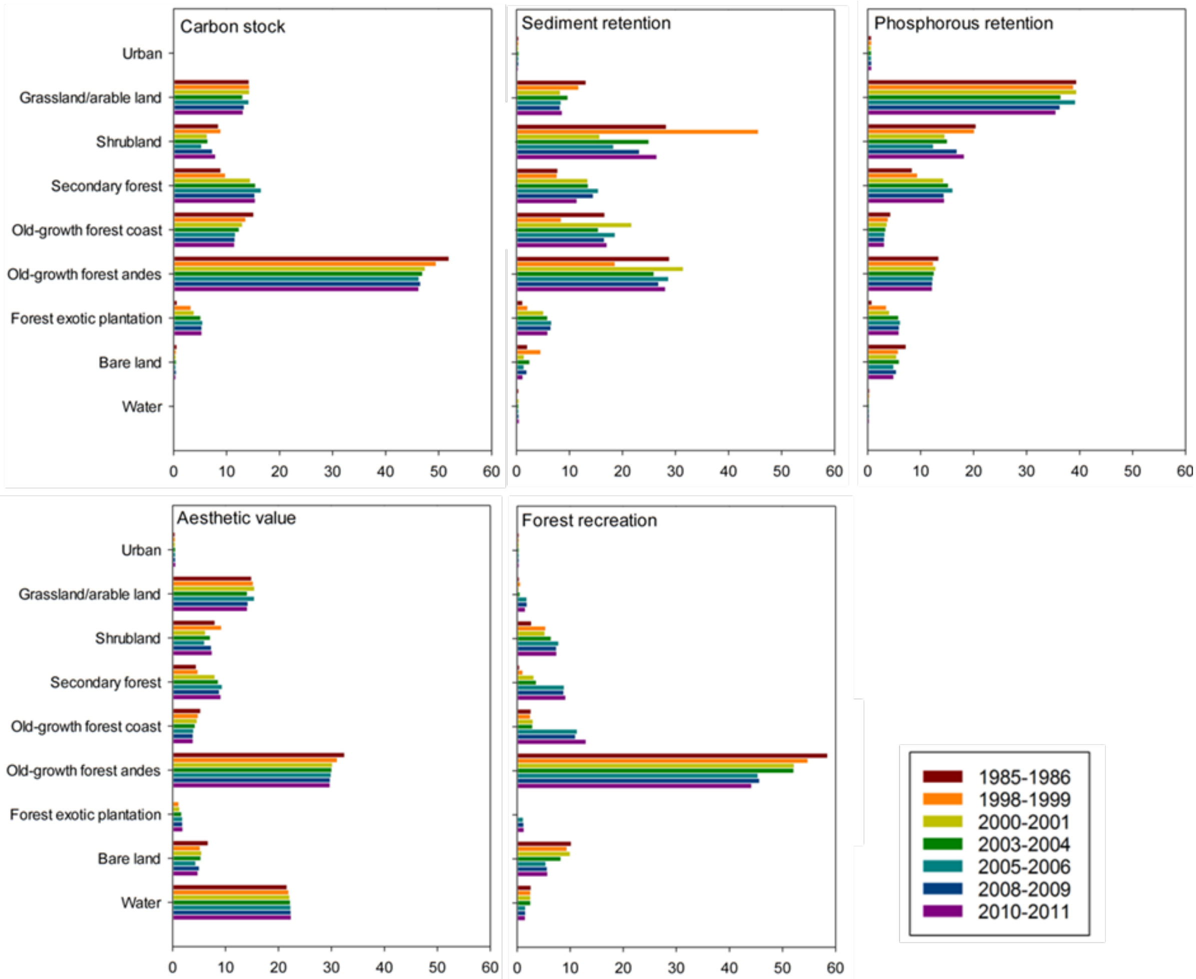


Figure 4.4 Proportion of selected ecosystem services in each land cover class over time. Each graph shows for a one-time period the relationship between land cover classes and ecosystem service. Plantation index was not included because this service corresponds hundred percent to the land cover Exotic forest plantations.

4.4.3 Implications for management

Managing multiple ecosystem services at a landscape scale is a key challenge in natural resources utilization (Bennett et al., 2015; Foley et al., 2011; Rodríguez et al., 2006). Multi-temporal and spatial ecosystem services analysis provide the understanding needed to propose management guides in line with the local requirements.

Our results indicate that sediment and phosphorous retention supply were strongly influenced by the presence of riparian forests. Based on their change over time, we conclude that prioritization of buffer protection zones is highly needed in the case study region- particularly forest buffer stripes downstream of disturbed human areas such as in large parts of the Central Valley. Figure 4.3 and Figure 4.4 show that even small areas of secondary forest helped to retain a large amount of sediment and phosphorus – they reveal furthermore that service supply changed strongly over time due to land cover change. These buffer strip zones are important to ensure an adequate water quality over time (Echeverria et al., 2008; Nick M. Haddad et al., 2015; Little et al., 2014; Prugh et al., 2008). Hence, we suggest that stream/river buffers should be kept as natural forest corridors to ensure landscape connectivity and ecosystem services supply. Landscape connectivity has not only a positive influence on regulating ecosystem services but is also important to cultural services that could enhance forest recreational services, especially in the Central Valley. Likewise, a decrease of connectivity is particularly relevant in these disturbed human areas, with complex topography and high pluviometry, due to the high erosion risk (López-Vicente et al., 2013) and the higher pressure for endangered species (Mitchell et al., 2013; Prugh et al., 2008).

Changes in carbon stocks is one of the most visible ecosystem services related to climate change mitigation mechanisms (i.e., carbon market and payment for ecosystem services) (IPCC and Barker, 2007). These carbon markets treat forest composed of natural and exotic species similarly without considering ecological processes related to the function of the different ecosystems. Even though our results show the high relevance of native forest for carbon stocks, the service supply summed over the whole region increased mainly due to the increase of exotic forest plantation areas especially in the Central Valley. As mentioned before, plantations have high economic relevance for the region and also provide some ecological benefits due to sediment and nutrient retention. However, the delivery of this service showed a discontinuous pattern in time mainly due to clear-cuts (Figure 4.3). Plantation logging practices such as large clear-cuts lead to trade-offs with other ecosystem services such as sediment and nutrient retention, water availability (quality and quantity) and asthenic beauty (Andersson et al., 2016; Armesto et al., 2010; Echeverría et al., 2007; Little et al., 2009; Manuschevich, 2016; Nahuelhual et al., 2012). These trade-offs and their effects in the ecosystem functions need to be taken into account to be able to balance the benefits of exotic forest plantation in the landscape (Cunningham et al., 2015).

The observed temporal and spatial fluctuations in the supply of ecosystem services in the case study region indicate that the socio-ecological system is highly dynamic at spatial and temporal scales (Locher-Krause et al., 2017b). This variability needs to be addressed by regional planning strategies – it is not sufficient to look only at the current state to be able to understand the dynamics of ecosystem services supply. Instead, it is important to integrate space and time perspectives to assess the effects of human activities on the landscapes, adding information about pattern/process frequency and magnitude (Bennett et al., 2015; Dallimer et al., 2015; Renard et al., 2015; Tomscha et al., 2016). Our analysis revealed differences in ecosystem services supply among the geomorphological units and indicated the sensitivity of certain areas to management decisions which varied over time. The consideration of the time dimension in the areas under pressure (such as the Central Valley and the Coastal Range) could be used to reconfigure landscapes to balance ecosystem services supply and demand at a regional scale. This information can be used to improve the type and location of management practices such as native species plantation, mixed forest plantation, smaller clear-cut areas, etc., to reduce negative effects of current exotic plantation management. However, future research regarding trade-offs between land use, ecosystem services, and biodiversity is further needed in order to understand and analyze the system as a whole (Seppelt et al., 2011; Volk, 2013).

4.4.4 Limitations and uncertainties

One of the major challenges of mapping ecosystem services supply in the area is the scarcity of spatially explicit and temporally concordant data. Mapping ecosystem services based on Landsat scenes and land cover maps bring several advantages to overcome long term data availability and reliability limitations (Tomscha et al., 2016). However, the use of purely land cover based proxies for ecosystem service mapping leads potentially to simplifications (Clec'h et al., 2016; Eigenbrod et al., 2010). The InVEST models are spatially explicit and require an intermediate level of input data that has been recognized as suitable for regional planning and management (Bagstad et al., 2013b; Goldstein et al., 2008; Nelson et al., 2009).

Results are influenced both by uncertainties of data and of the model structure - that simplifies the description of ecological and physical processes (Sharp et al., 2015). Calibration of the models to local conditions and validation with independent data increase the reliability of model predictions but is limited by data availability. Based on the data obtained from the water national monitoring program we were able to validate the modeled sediment retention. We compared available observations on Total Suspended Solids (TSS) at the catchment level with simulations with InVEST. Therefore the TSS data had to be converted to annual loads with the Load Estimator LOADEST (Runkel et al., 2004) to the units of the InVEST model (Mg per year). The model results explained approximately 90 % of the observed sediment loads across the case study region and all periods (4 S-A Figure 1).

For the other ecosystem services an exhaustive literature review as well as an analysis of available administrative data was performed (for more information see Supplementary material 1) to obtain data for comparison with the modeled values. Changes in carbon stocks and plantation site productivity modeled values were discussed with local experts (Professor Gayoso and Sandoval, Personal communication) to corroborate plausibility of the modeled results. Our modeled results were in line with the results from local measurements. Phosphorous export values predicted by InVEST were in the range of local measurements (Cuevas et al., 2006; Oyarzun et al., 1997) - more information is provided in the Supplementary material S1.

Due to the presence of extended cloud cover, we were unfortunately not able to cover the period between 1986 and 1998. Therefore, temporal dynamics could only be assessed properly for the periods after 1998. We can only speculate about the dynamics between 1986 and 1998 at a higher temporal resolution. For the periods after 1998 the temporal resolution seems to be adequate to capture the processes of land use change by forest management, expansion of plantations and to a lower degree by urban expansion gave the temporal scale at which these processes manifest at the landscape.

4.5 Conclusions

The presented analysis has demonstrated the relevance of quantifying and mapping ecosystem services over time, integrating knowledge about its frequency and magnitude of change. Our results support the statement that more multi-scale ecosystem service assessment is needed to understand this dynamic process. This is clear for the fast changes in plantation production and forest recreation. The omission of dynamic periods would have led to a different perception about the speed and extent of the changes in carbon stocks, plantation production, and forest recreation service supply. The strong temporal differences in sediment retention highlight that average conditions across a longer period do not capture the importance of regulating services with high sensitivity towards peak events well. For phosphorus retention and landscape aesthetics services, an omission of single periods would have been less dramatic – but the analysis still would have missed important dynamics in the trajectory of ecosystem services supply (i.e., a discontinuous increase of phosphorous retention and landscape aesthetic). Thereby, our results also highlight the potential of using the freely available Landsat time series for ecosystem service mapping.

Our findings further underline the importance of deforestation and exotic forest plantations expansion for ecosystem services supply in the South of Chile. Furthermore, the importance of distinguishing the processes in different regions became clear. While agricultural and exotic forest plantations expansion has been of important in the Central Valley, natural forest logging, as well as exotic forest plantations, have been of central concern in the other regions. We found that while changes in carbon stocks in the Central Valley increased - mainly due to the exotic forest plantations- changes in carbon stocks in other regions such as the Coastal range decreased due to the loss of old growth and secondary forest. The changes in sediment and phosphorous retention supply over time underline existing findings of the high relevance of forest vegetation, especially riparian buffers. This highlights the huge importance to protect and enhance buffer areas close to streams and the relevance of further research to identify areas to maintain a balance between ecosystem services supply and demand at a regional scale. Land use planning in the region should, therefore, pay attention to conserve and enhance these valuable ecosystems. Such a strategy could be combined with the goal of biodiversity conservation if riparian forest could be established as corridors to enhance landscape connectivity. While the number of protected forest areas increased over time, landscape aesthetics showed a contrasting pattern – with a decline in its natural degree over time. This decrease is due to the fact that the new protected areas were created in remote areas far from the anthropic pressure of recreational interest points. Especially the lack of protected areas in the Central Valley should be of concern for land use planners since this has negative effects on the supply of regulating and cultural services. Consequently, we want to highlight the importance of our results for further research on ecosystem services trade-offs and synergies by using spatio-temporal approaches to support management strategies in multi-functional landscapes.

4.6 Acknowledgments

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4.7 Supplementary Material

4.7.1 Supplementary Material A

4 S-A Table 1 Land use land cover classifications product.

LULC Code	LULC Description
1	Artificial surfaces and associated areas (Urban areas >50%)
2	Closed to open (>15%) herbaceous vegetation (natural and artificial annual pastures, agricultural areas with different crops such as wheat or oats)
3	Closed to open (>15%) shrubland (< 10% tree cover, result of natural succession or native forest logging).
4	Closed to open secondary forest, (>15%) broadleaved evergreen or semi-deciduous forest (>5m) secondary growth native.
5	Closed to open old-growth forest (>15%) broadleaved evergreen or semi-deciduous forest from Coastal areas.
6	Closed to open old-growth forest (>15%) broadleaved evergreen or semi-deciduous forest from Andes areas.
7	Productive exotic forest plantation (*Pinus radiata* and *Eucalyptus sp*.)
8	Bare areas
9	Water bodies
10	Areas without land cover information due to clouds and shadows, masked by Fmask

4 S-A Table 2 Carbon model input for aboveground biomass, belowground biomass, soil organic carbon and dead organic matter carbon by LULC type (A and C, 2005; Gayoso, 2001; Gayoso and Schlegel, 2003; Gonzalez and Gayoso, 2005; Keith et al., 2009; Schlegel, 2001; Schlegel and Donoso, 2008)

LULC Description	Aboveground Biomass	Belowground Biomass	Soil Organic Carbon	Dead Organic Matter
	Mg ha^{-1}			
Artificial surfaces and associated areas (Urban areas >50%)	5.5	0.6	15.0	0.5
Closed to open (>15%) herbaceous vegetation (natural and artificial annual pastures, agricultural areas with different crops such as wheat or oats)	4.0	5.0	166.0	2.0
Closed to open (>15%) shrubland (< 10% tree cover, result of natural succession or native forest logging).	8.0	48.0	164.0	20.0
Closed to open secondary forest, (>15%) broadleaved evergreen or semi-deciduous forest (>5m) secondary growth native.	161.2	43.6	148.5	50.5
Closed to open old-growth forest (>15%) broadleaved evergreen or semi-deciduous forest from Coastal areas.	284.9	79.5	180.91	59.4
Closed to open old-growth forest (>15%) broadleaved evergreen or semi-deciduous forest from Andes areas.	387.3	103.8	149.7	102.8
Productive exotic forest plantation (*Pinus radiata* and *Eucalyptus sp.*)	133.7	33.7	117.9	14.7
Bare areas	2.3	0.5	21.7	0.5
Water bodies	0.0	0.0	0.0	0.0
Areas without land cover information due clouds and shadows	0.0	0.0	0.0	0.0

4 S-A Table 3 Sediment retention input table. USLE cover and management factor (C) and USLE support practice factor (P) and Sediment filtration by LULC type (Bonilla et al., 2010; Bonilla and Johnson, 2012; Renard and Freimund, 1994).

LULC Description	USLE C	USLE P
Artificial surfaces and associated areas (Urban areas >50%)	0.01	1
Closed to open (>15%) herbaceous vegetation (natural and artificial annual pastures, agricultural areas with different crops such as wheat or oats)	0.038	1
Closed to open (>15%) shrubland (< 10% tree cover, result of natural succession or native forest logging).	0.13	1
Closed to open secondary forest, (>15%) broadleaved evergreen or semi-deciduous forest (>5m) secondary growth native.	0.004	1
Closed to open old-growth forest (>15%) broadleaved evergreen or semi-deciduous forest from Coastal areas.	0.003	1
Closed to open old-growth forest (>15%) broadleaved evergreen or semi-deciduous forest from Andes areas.	0.003	1
Productive exotic forest plantation (*Pinus radiata* and *Eucalyptus sp*.)	0.006	1
Bare areas	1	1
Water bodies	0.0	1
Areas without land cover information due to clouds and shadows	0.0	1

4 S-A Table 4 Phosphorous retention input table. Evapotranspiration coefficient (ETcoeff), Rooting depth, P export coefficients and vegetation filtration by LULC type (Cuevas et al., 2014, 2006, Oyarzun et al., 2015, 1997; Oyarzún et al., 2007).

LULC Description	ETcoef	Root depth (mm)	Load P (kg ha $^{-1}$ yr^{-1})	Filtering Capacity (%)
Artificial surfaces and associated areas (Urban areas >50%)	0.3	1	0.1	5
Closed to open (>15%) herbaceous vegetation (natural and artificial annual pastures, agricultural areas with different crops such as wheat or oats)	0.65	2000	0.05	40
Closed to open (>15%) shrubland (< 10% tree cover, result of natural succession or native forest logging).	0.7	2500	0.05	75
Closed to open secondary forest, (>15%) broadleaved evergreen or semi-deciduous forest (>5m) secondary growth native.	0.8	6000	0.05	75
Closed to open old-growth forest (>15%) broadleaved evergreen or semi-deciduous forest from Coastal areas.	1	7000	0.011	80
Closed to open old-growth forest (>15%) broadleaved evergreen or semi-deciduous forest from Andes areas.	1	7000	0.011	80
Productive exotic forest plantation (*Pinus radiata* and *Eucalyptus sp.*)	1	7000	0.011	75
Bare areas	0.2	1	0.001	5
Water bodies	1	1	0.001	5
Areas without land cover information due clouds and shadows	0.3	1	0.0	0.0

4 S-A Table 5 Annual export (sediment + solution) of phosphorous (mg m^{-2} yr^{-1}) from watersheds of the Rupanco lake (1994-1995) (modified from Oyarzun et al. 1997)

Watershed	Land use in the watershed (%)					PO_4-P	P_{tot}
	Forest	Grassland	Agricultural	Shrubland	Lake		
Forest	98.4	0.0	0.0	1.6	0.0	16.3	64.8
Forest	100.0	0.0	0.0	0.0	0.0	14.3	104.5
Shrubland-grassland	0.0	54.8	2.0	27.5	15.7	12.6	66.0
Grassland	17.0	72.5	6.9	3.6	0.0	12.3	117.9
Grassland	31.0	67.5	1.5	0.0	0.0	19.3	118.5
Agricultural grassland	23.3	56.2	18.5	1.8	0.0	12.9	93.5

4 S-A Table 6 Description of the degree of naturalness used no assess Aesthetic value (modified from (Machado, 2004).

Degree of naturalness code	Degree of naturalness description	LULC code
1	Artificial surfaces and associated areas were the vegetation has been deliberately determined by humans with loss of the previous habitat	1-7
2	Semi-natural areas with changes but the structure of the vegetation is basically the same	2-3-4
3	Natural areas with no or minimal disturbance by human activities (maintain structure and species composition)	5-6-9
no apply		8-10

4.7.1.1 Site index description

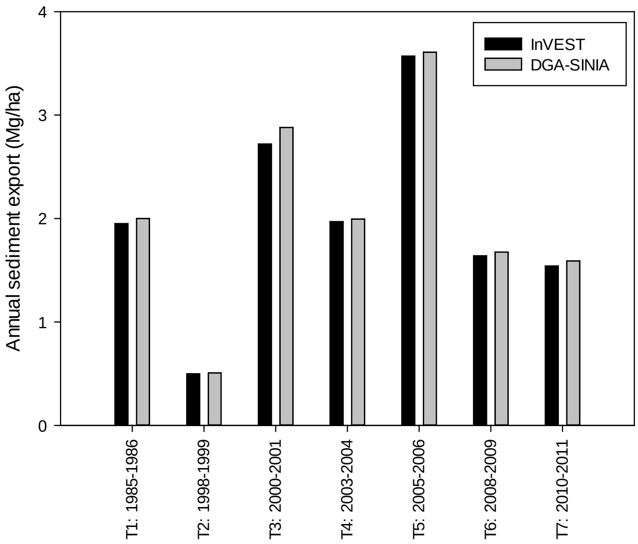
We use the site index developed by the Forest Research Institute in Chile. The index was obtained from extensive and intensive forest inventory were basic data from height and age were measured in different geographic areas. The geographical areas were stratified considering homogeneous soil and climatic areas to avoid distortions on estimators due to different age distributions. The empirical equations for the site index were fitted base on a regression model (height and age variables) for the species (García, 1970; INFOR, 1999; Pinilla S., 1998; Schlatter and Gerding, 1995). The equation obtains for the index curves were:

$\mathrm{Ln}\,A = a + b/E,$

Where A is the mean height of dominant and codominant trees and E is the age, for *Pinus radiata* and

$H=75.3\,(l-e^{-bt})^{1/0.863}$

Where H is the height dominant in meters T is the age in years, b depends on the site quality, for *Eucalyptus* sp. (Garcia, 1995).

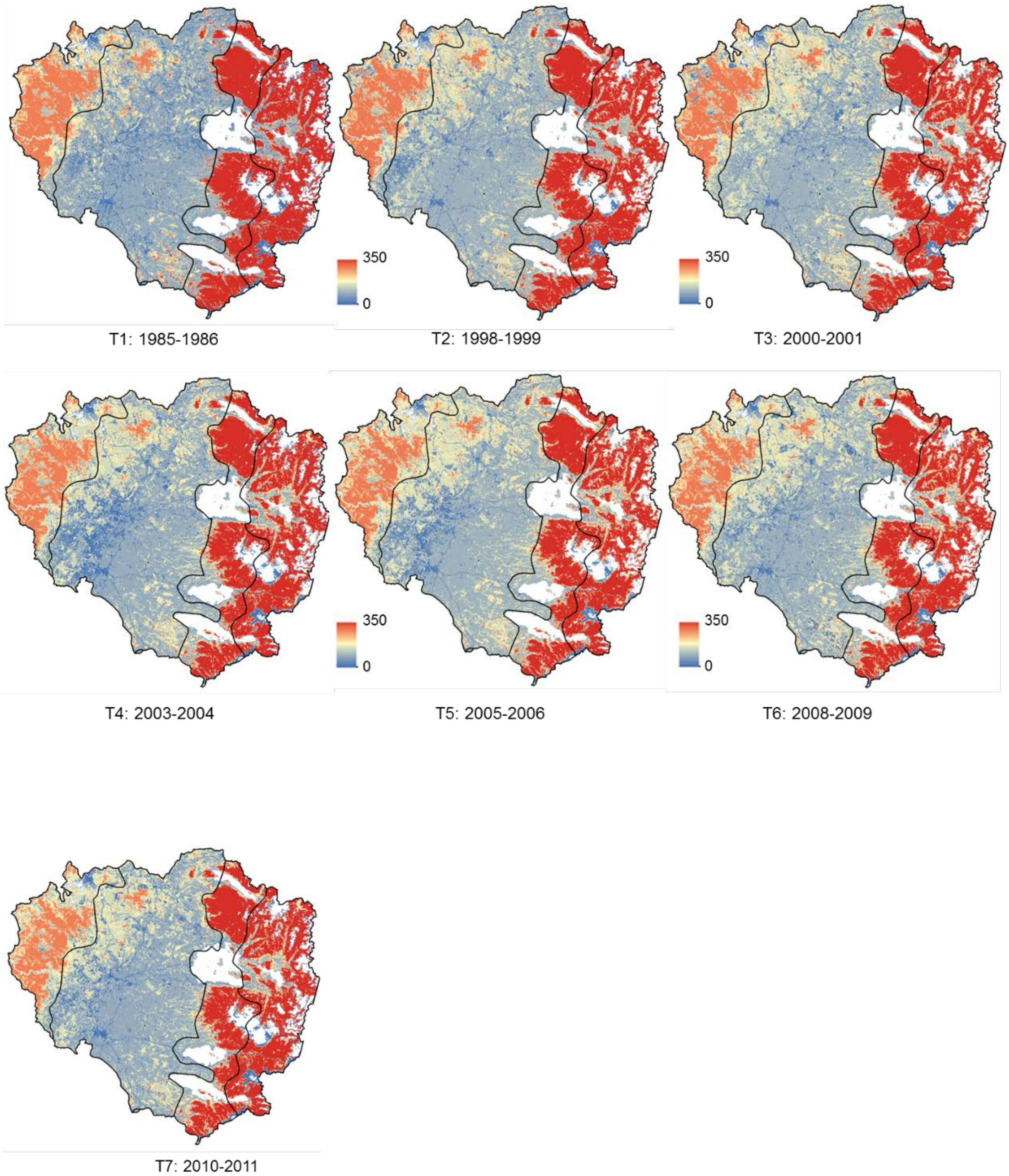


4 S-A Figure 1 Validation of modeled sediment export InVEST (black bars) versus field data obtained from the Chilean national water agency (DGA, Dirección Nacional de Aguas, grey bars).

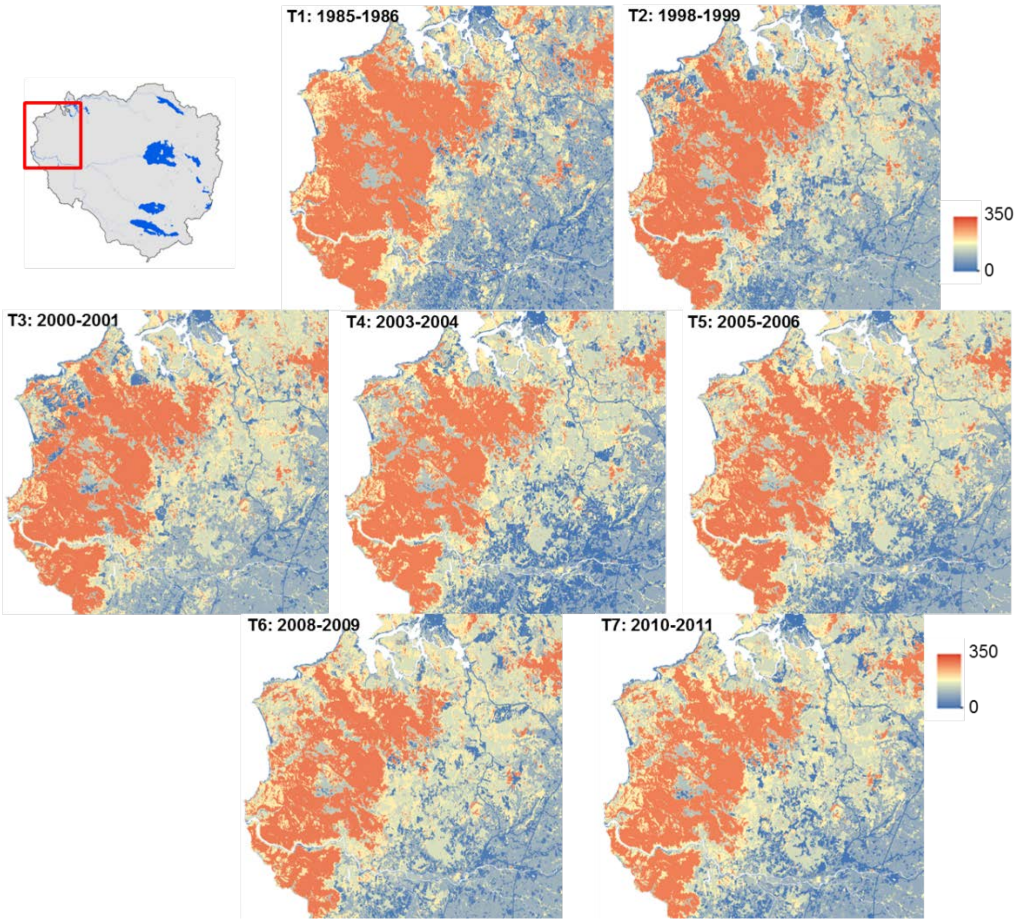
4.7.2 Supplementary Material B

4.7.2.1 Maps of selected ecosystem services and its zoom to clearly visualize changes on ecosystem services supply over time.

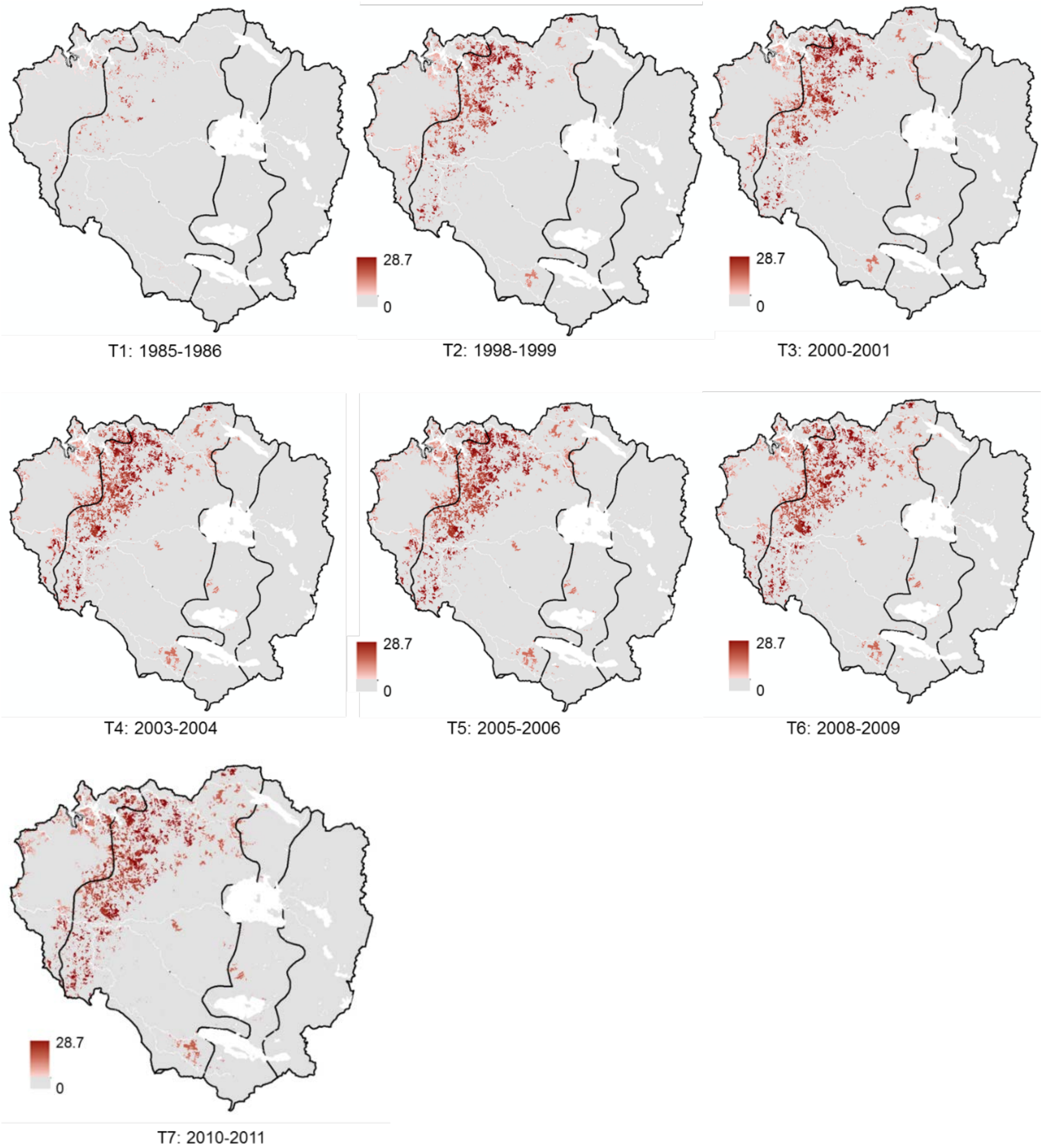
4 S-B Figure 1 Carbon storage (Mg ha^{-1})



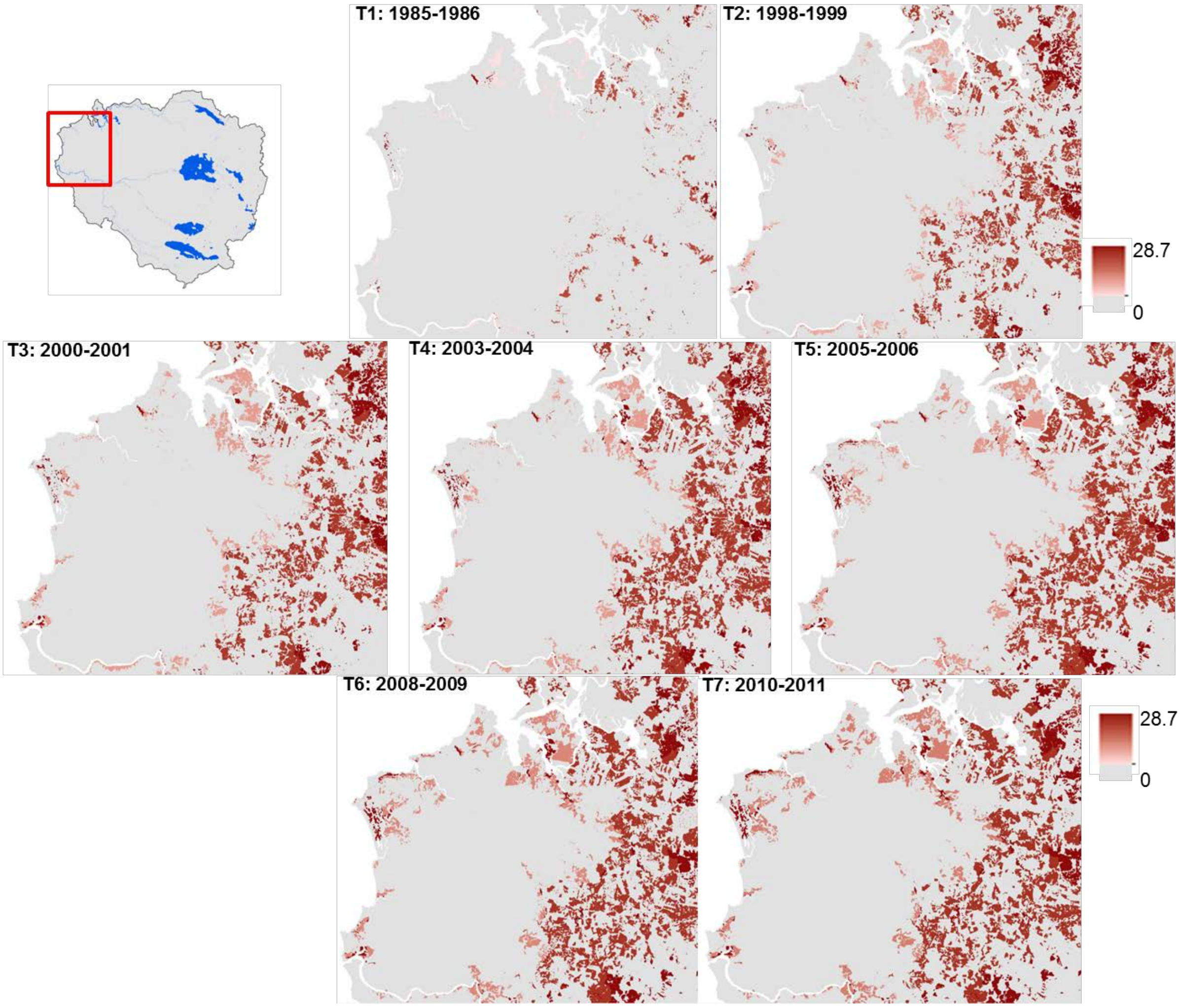
4 S-B Figure 2 Zoom area, carbon storage (Mg ha⁻¹)



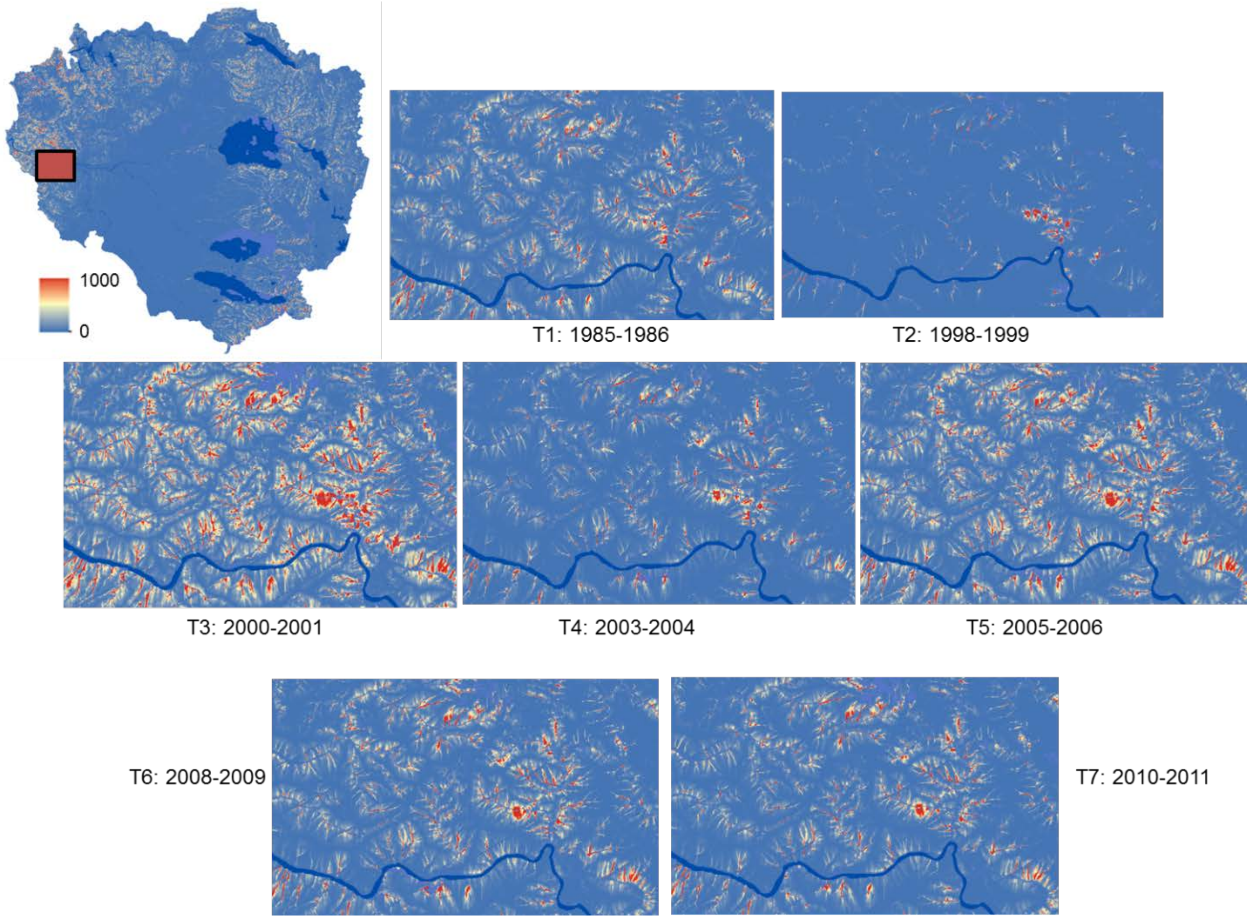
4 S-B Figure 3 Plantation production index



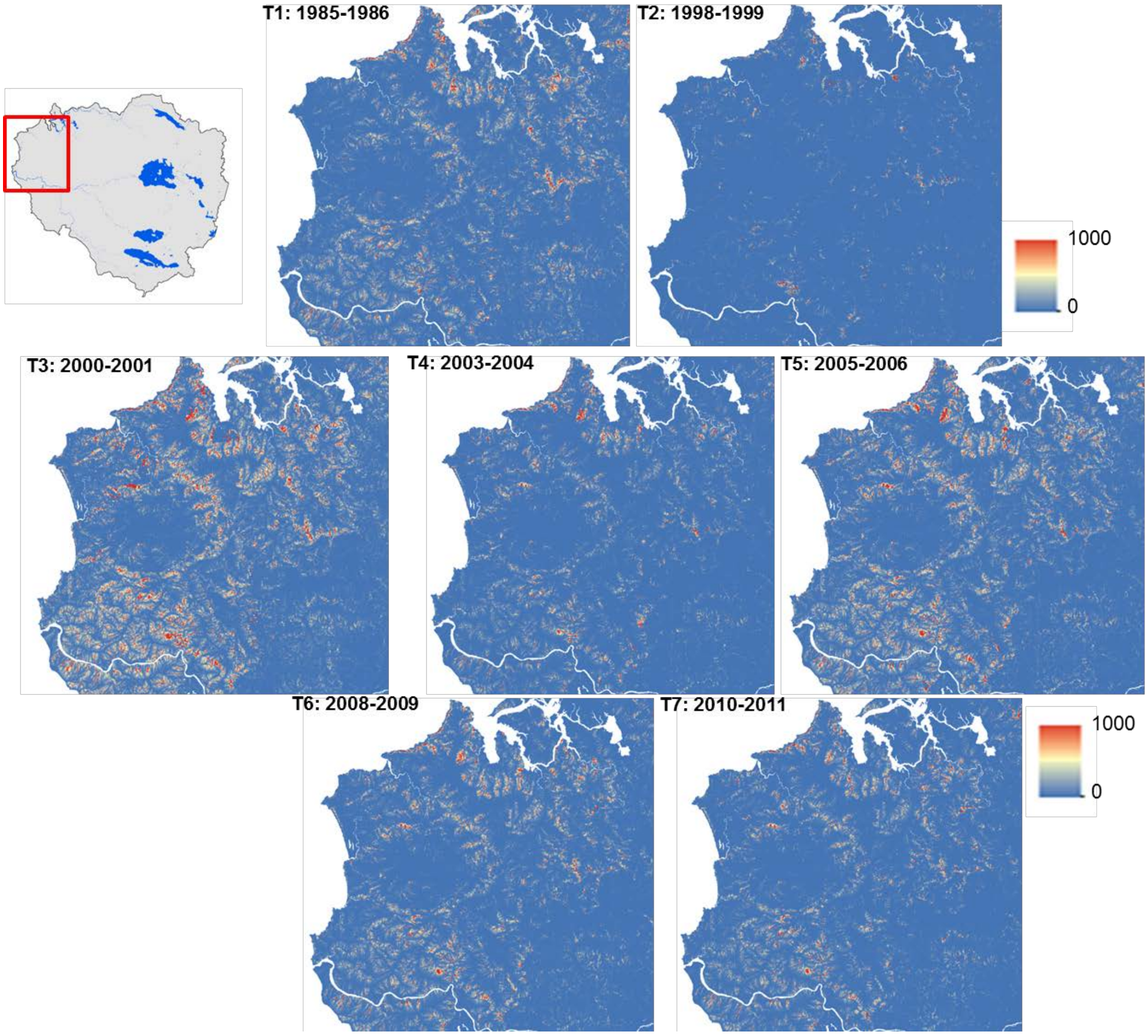
4 S-B Figure 4 Zoom area, the Plantation production index



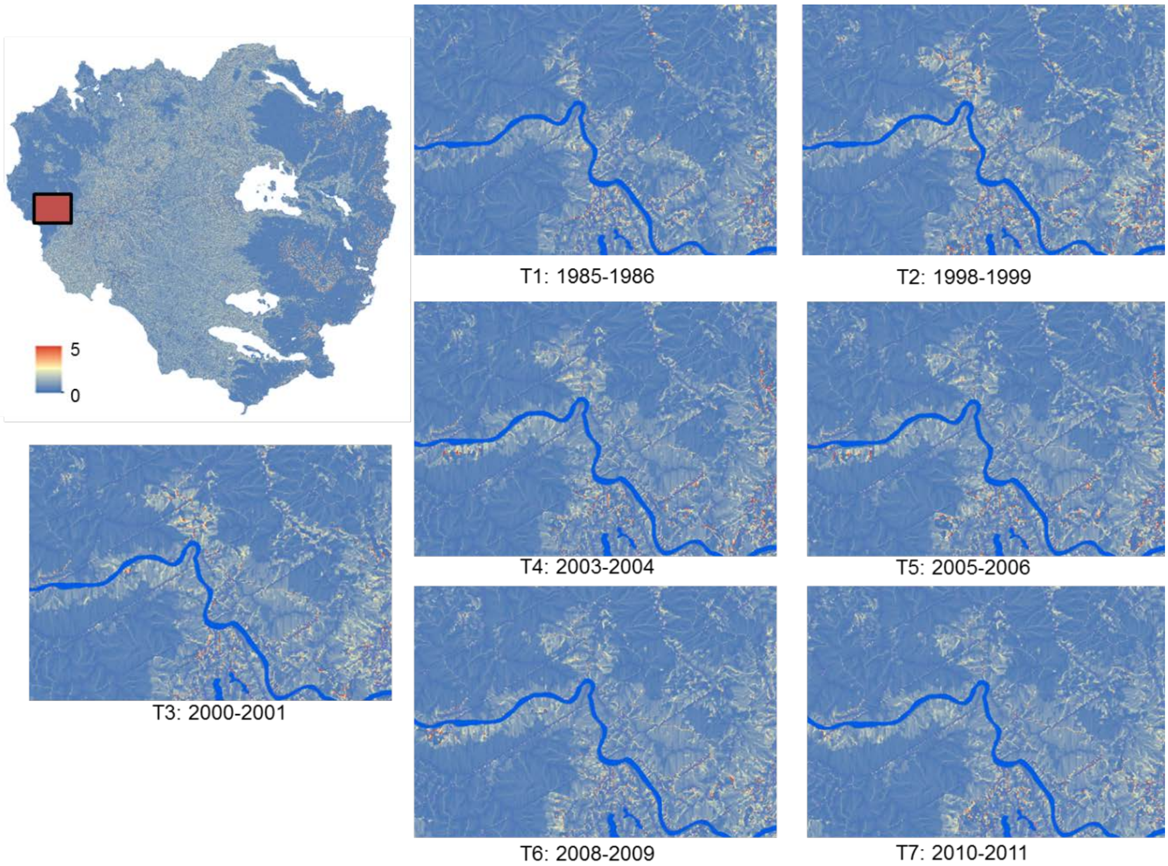
4 S-B Figure 5 Sediment retention (Mg ha^{-1})



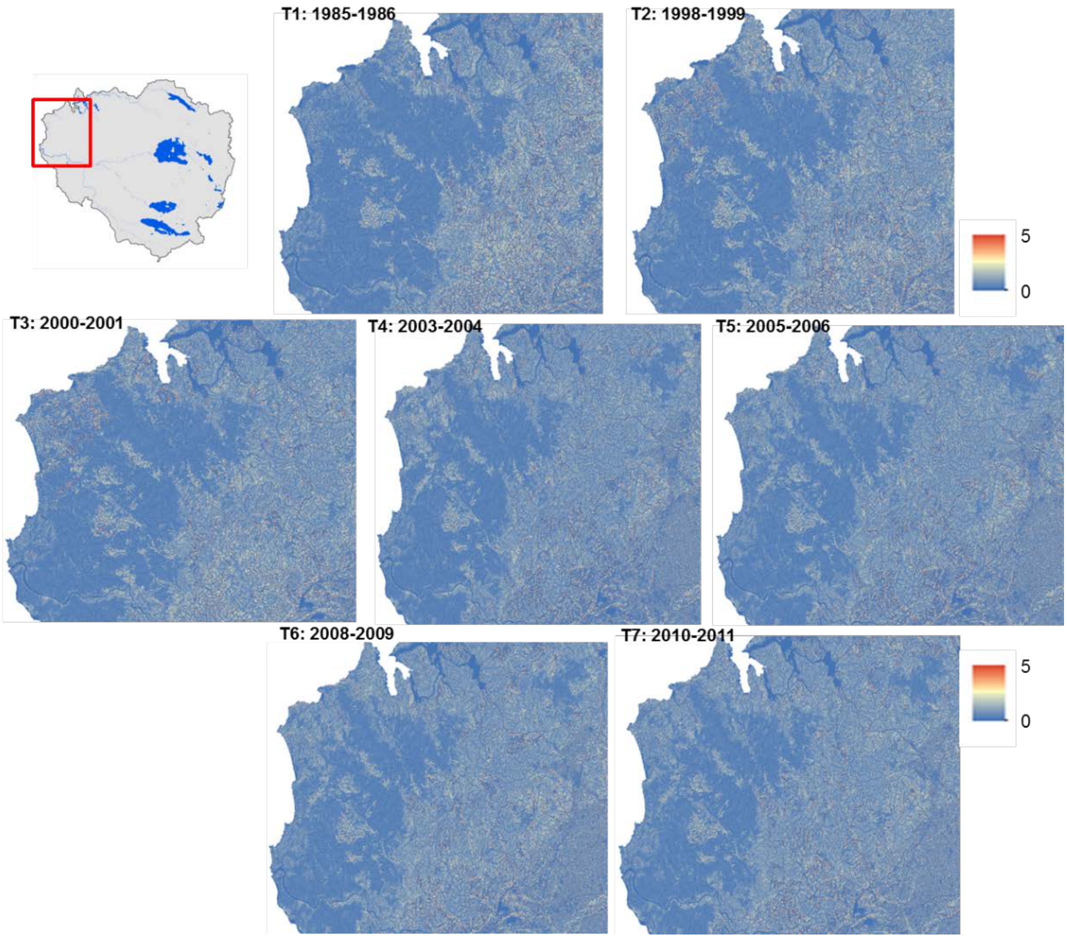
4 S-B Figure 6 Zoom area, sediment retention (Mg ha^{-1})



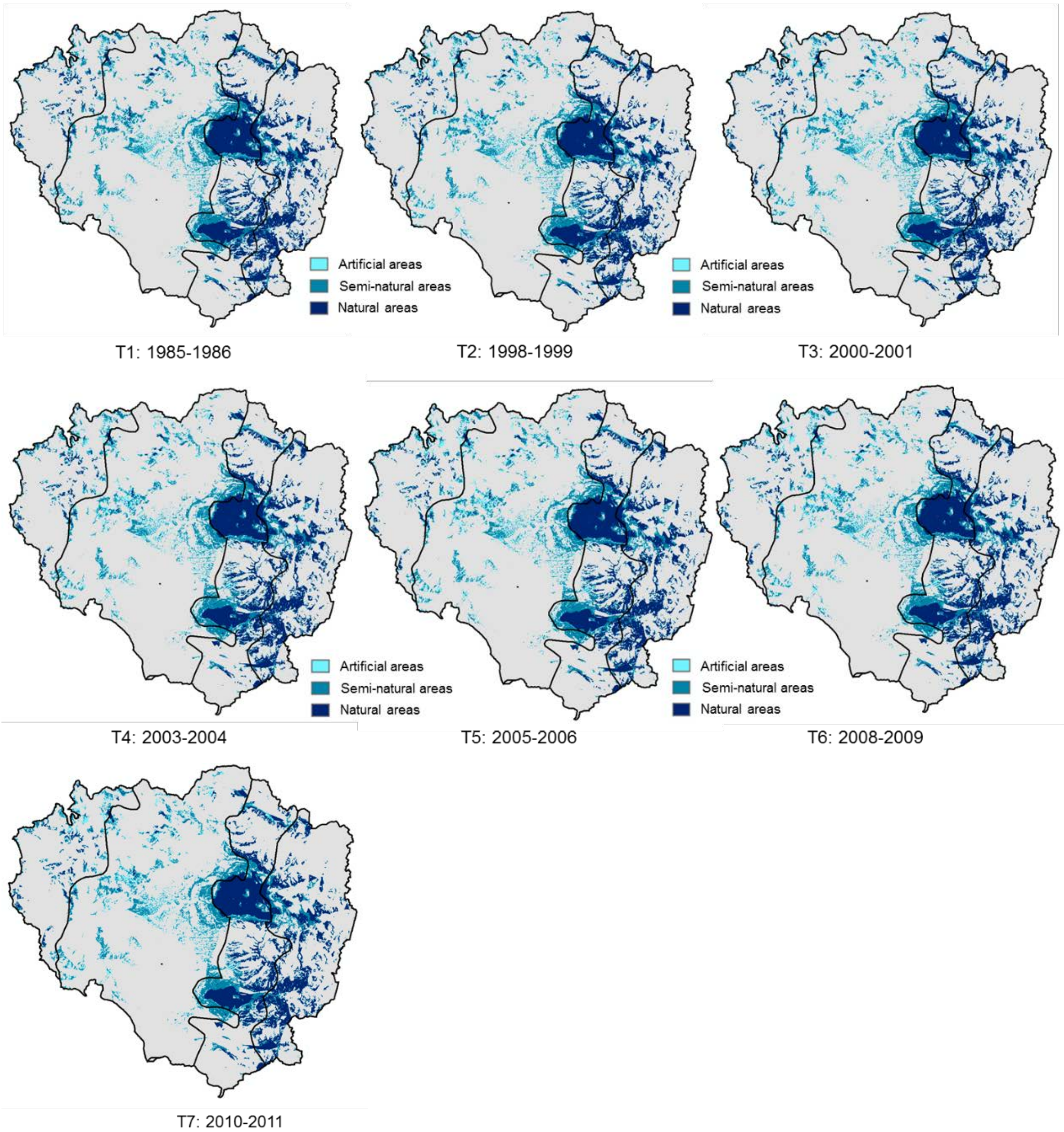
4 S-B Figure 7 Phosphorous retention (Kg ha^{-1})



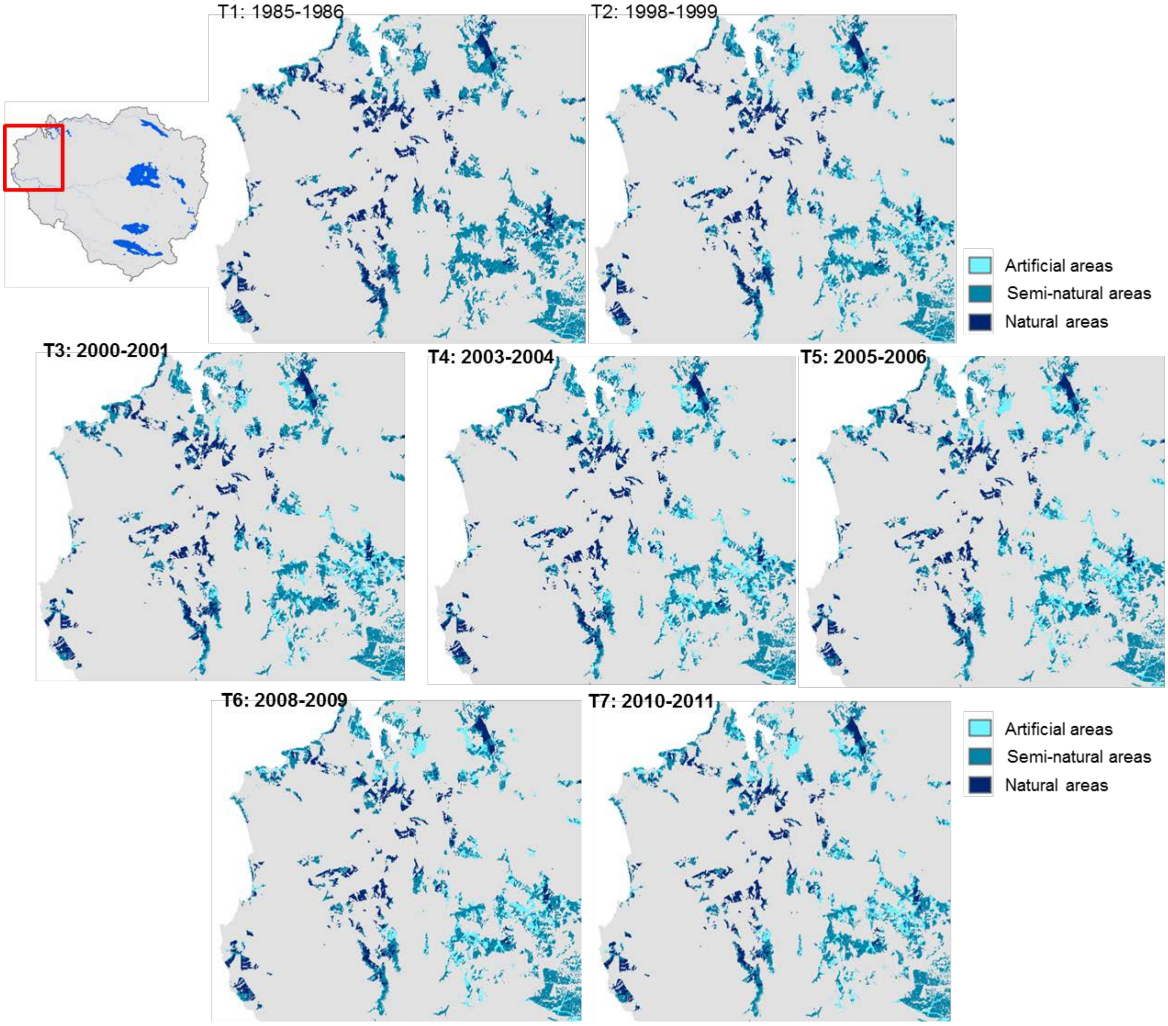
4 S-B Figure 8 Zoom area, phosphorous retention (Kg ha⁻¹)



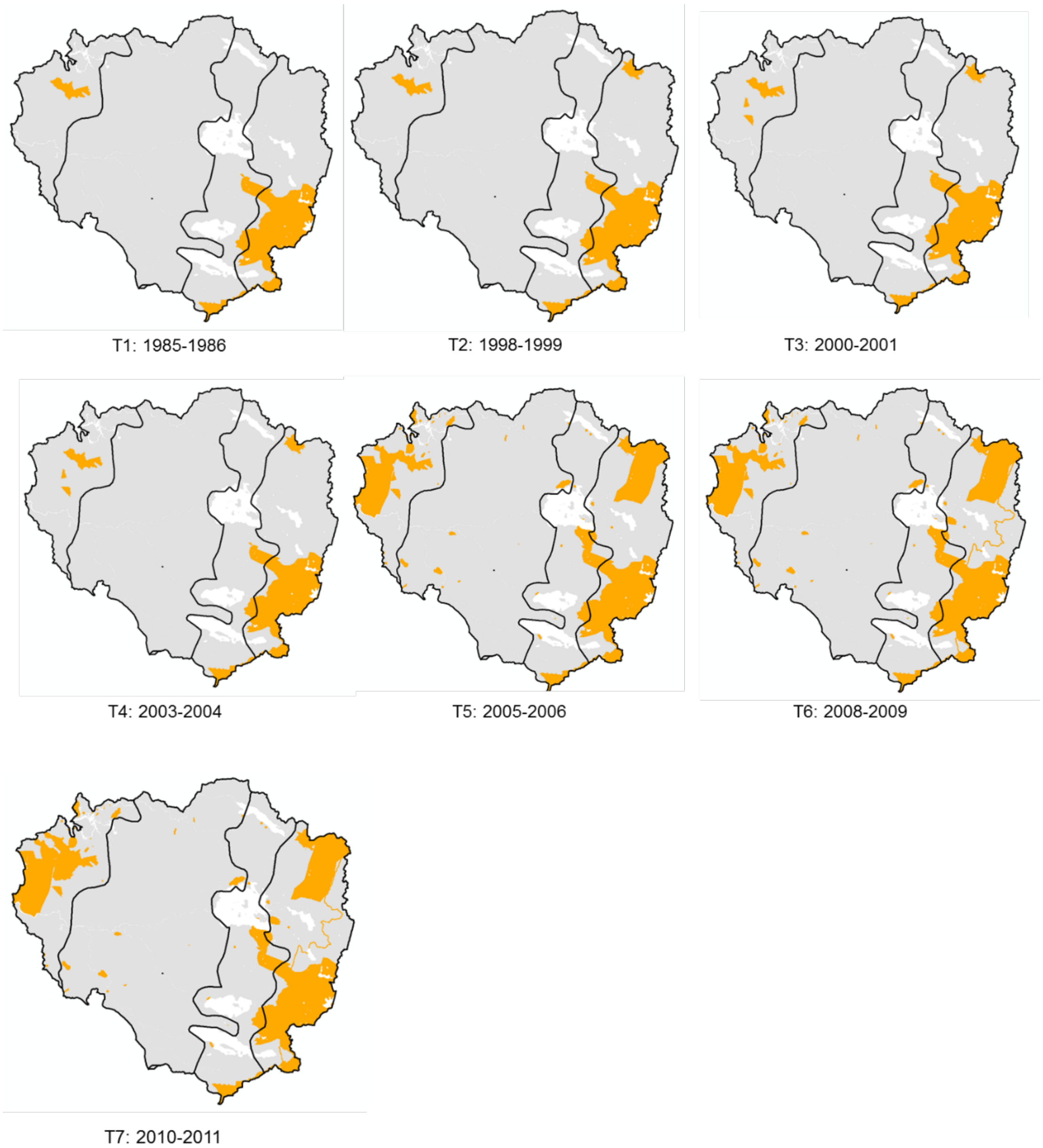
4 S-B Figure 9 Landscape aesthetic (unitless)



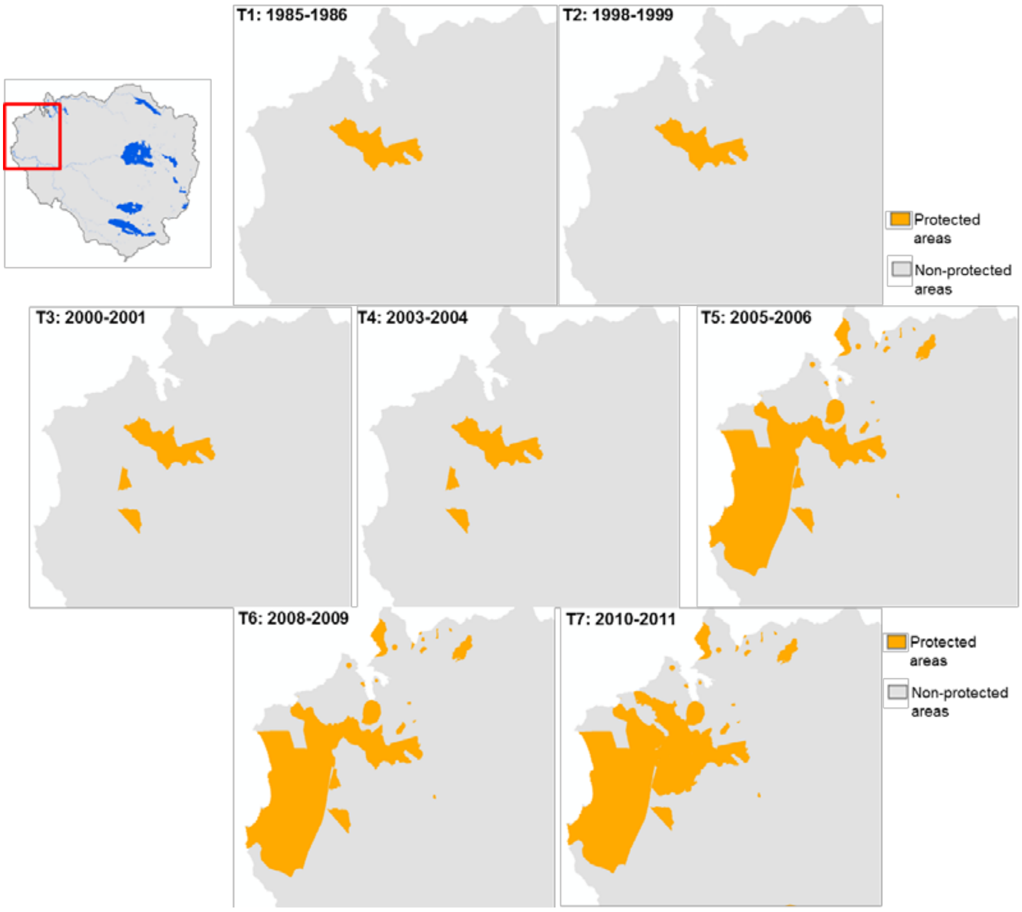
4 S-B Figure 10 Zoom area, Landscape aesthetic (unitless)



4 S-B Figure 11 Forest recreation (% of the area)



4 S-B Figure 12 Zoom area, forest recreation (% of the area)



4 S-B Table 1 Land cover composition. Percentage of the four geomorphological units covered by the different land covers classes. The area of the four regions was constant across time.

	T1: 1985-1986				T2: 1998-1999				T3: 2000-2001			
Land cover	Andes range	Pre-andean	Central valley	Coastal range	Andes range	Pre-andean	Central valley	Coastal range	Andes range	Pre-andean	Central valley	Coastal range
Urban	0.3	0.5	1.7	0.7	0.3	0.5	1.8	0.9	0.3	0.5	1.9	0.9
Grassland/arable land	3.7	9.5	51.3	7.0	4.1	10.7	51.1	4.5	4.0	10.4	51.6	5.2
Shrubland	5.9	7.3	16.6	11.8	9.0	9.2	14.7	14.9	6.2	6.4	10.0	13.0
Secondary	0.9	1.2	10.1	17.9	1.8	2.3	11.5	14.7	6.0	5.5	16.4	16.7
Old-growth forest	60.2	50.7	5.3	53.0	58.1	47.6	4.3	48.1	56.0	46.5	3.8	46.7
Plantations	0.0	0.0	0.9	2.4	0.0	0.2	5.8	7.4	0.0	0.5	6.8	8.4
Bare land	6.8	3.4	12.7	4.0	4.4	2.2	9.4	6.3	5.4	2.8	8.2	5.9
Water	4.4	23.4	1.1	2.8	4.4	23.4	1.1	2.8	4.4	23.5	1.1	2.8
Areas without information	17.7	4.0	0.2	0.3	17.7	4.0	0.2	0.3	17.7	4.0	0.2	0.3

	T4: 2003-2004				T5: 2005-2006				T6: 2008-2009			
Land cover	Andes range	Pre-andean	Central valley	Coastal range	Andes range	Pre-andean	Central valley	Coastal range	Andes range	Pre-andean	Central valley	Coastal range
Urban	0.3	0.5	2.0	1.0	0.3	0.6	2.0	1.0	0.3	0.6	2.0	1.1
Grassland/arable land	3.4	9.5	47.2	4.4	3.6	10.4	51.3	4.8	3.6	8.4	48.0	4.3
Shrubland	7.8	7.6	9.2	13.0	7.7	6.5	6.7	10.9	7.3	8.1	11.7	12.5
Secondary	6.7	6.6	16.6	19.2	7.3	7.5	16.9	22.5	7.3	7.3	14.7	22.0
Old-growth forest	55.7	46.1	3.3	44.8	55.1	45.4	3.1	42.4	55.0	45.1	3.0	41.7
Plantations	0.0	0.8	9.0	10.7	0.0	0.9	9.6	11.6	0.0	1.0	9.2	11.0
Bare land	3.9	1.5	11.4	3.8	3.6	1.2	9.0	3.5	4.2	2.0	10.0	4.2
Water	4.4	23.4	1.1	2.8	4.4	23.5	1.2	2.8	4.4	23.5	1.2	2.9
Areas without information	17.8	4.0	0.2	0.3	17.8	4.0	0.3	0.4	17.8	4.0	0.3	0.4

	T7: 2010-2011			
Land cover	Andes range	Pre-andean	Central valley	Coastal range
Urban	0.3	0.6	2.1	1.1
Grassland/arable land	3.2	8.6	47.9	3.8
Shrubland	7.6	7.6	13.4	13.3
Secondary	7.4	7.7	14.8	21.8
Old-growth forest	55.0	45.1	3.0	41.7
Plantations	0.0	1.1	9.2	11.0
Bare land	4.3	1.9	8.2	4.0
Water	4.5	23.5	1.2	2.9
Areas without information	17.7	3.9	0.2	0.3

5 Reconnecting fragmented Patagonian landscapes: Insights from an ecosystem services supply and beneficiaries' assessment.

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Journal Ecosystem services
Submitted: Mach 2019
In Review

5.1 Abstract

Landscape management strategies oriented to reconnect highly fragmented landscapes play a crucial role in maintaining ecosystem services supply and materials flow in socio-ecological systems. At a regional scale, the integration of different structural landscape elements and types of protected areas show a huge potential to improve ecosystem services and conservation efforts. We analyzed the contribution of structural connectivity areas (SCA), defined as the integration of riparian corridors (linear elements) and national conservation units (patch elements) at the regional level, on ecosystem service supply and beneficiaries in a highly fragmented area hotspot of biodiversity in Southern Chile. Based on a spatially explicit approach, we mapped and identified the beneficiaries of selected forest ecosystem services (carbon storage, sediment retention, phosphorous retention, plantation site productivity, landscape aesthetics, and forest recreation). Our results show that the SCA provided more than 60% of the total ecosystem service supply of the whole the study area in the region —especially for regulating services, even when SCA accounts only for the 40% of the total study area. Furthermore, the highest density of beneficiaries was located in or surrounding the SCA. The spatial identification of beneficiaries revealed a spatial mismatch between ecosystem services supply and their beneficiaries: beneficiaries were mainly allocated in the Central Valley while high ecosystem services provision was located in the Andes and Pre-Andes range. Our research highlight the need for accounting for the interconnection between ecosystem services and its beneficiaries to improve landscape management. Moreover, SCA reveals its potential as a landscape planning strategy enhancing ecosystem services supply in areas with a high density of beneficiaries (especially in the Central Valley).

Keywords
Forest ecosystem services, modeling, structural connectivity areas (SCA), ecosystem service supply, beneficiaries, Southern Chile.

5.2 Introduction

Landscape connectivity is based on the complex interaction between functional and structural elements, playing a crucial role in maintaining ecosystem functions and services at different spatio-temporal scales (Beier and Noss, 1998; Fischer and Lindenmayer, 2007; Mitchell et al., 2013; Taylor et al., 1993). Spatio-temporal changes in landscape composition and configuration drastically impair the ability of ecosystems to provide multiple services (Lamy et al., 2016; Mitchell et al., 2015). Nowadays, habitat fragmentation is a pressing concern in many socio-ecological systems because it leads to an increased vulnerability of the landscape systems to external influences such as invasive species (Kawasaki et al., 2017; Yates et al., 2004), modifications of the local climate (i.e., evapotranspiration, wind) (Agosta et al., 2017; Laurance et al., 2018), alter water flow (Brauman et al., 2007), among others impacts. Despite the relevance of landscape planning strategies oriented to reconnect strongly fragmented landscapes –especially in high biodiversity regions–conservation areas are still commonly isolated between each other (Watson et al., 2014). Pringle (2017) highlighted the huge potential to improve conservation efforts by overhaul protected areas current management strategies through the expansion and restoration of current landscape elements (i.e., degraded riparian corridors). Likewise, several studies have shown that the implementation of landscape elements such as corridors can balance and mitigate landscape fragmentation impacts and thus support landscape planning (Garcia et al., 2017; Hofman et al., 2018; Naidoo et al., 2018; Newmark et al., 2017). Corridors, defined as structural linear landscape features, promote conservation and at the same time maintain system complexity (Hess and Fisher, 2001; Hobbs, 1992; Lindenmayer et al., 2006). Furthermore, riparian corridors are widely recognized as an interface between terrestrial and aquatic systems, controlling diverse ecosystem functions and services (Lyon and Gross, 2005; Naiman et al., 1993). These areas might impact the supply of several ecosystem services such as water and flood regulation, habitat provision for pollinators, birds, fish and mammals, carbon sequestration, and recreation (Cole et al., 2015; Díaz et al., 2006; Gillies and St. Clair, 2008; Turner et al., 2012). Additionally, landscape management programs oriented to combine linear (i.e., riparian corridors) and patchy landscape elements such as conservation areas in highly fragmented landscapes offer a potential solution to maintain/ensure ecosystem services provisioning and biodiversity protection over time.

In this study, we analyze the relevance of combining linear and patchy landscape elements (from now on structural connectivity areas—SCA) in a highly fragmented region in Southern Chile based on its contribution to the provisioning of forest ecosystems services to different beneficiaries. We defined SCA as the integration of riparian corridors (surrounding the main rivers that flow from the Andes to the Coastal range) and the national conservation units at the regional level. These conservation areas protect the Valdivian temperate rainforest ecosystem, one of the largest relict of the temperate forest in the world (J. Armesto et al., 1998) with high rates of endemic flora and fauna

(Armesto et al., 1996). One important characteristic of the national conservation areas is that they are highly disconnected. Even if we consider other private conservation initiatives in the area the main conservation areas are located in the mountain ranges (the Andes and Coastal range), representing only a few ecosystems and using remote areas with poor accessibility (Durán et al., 2013). Furthermore, the region has undergone a large landscape transformation process (Echeverría et al., 2007; Locher-Krause et al., 2017b; Miranda et al., 2015) that altered the ability of the system to provide benefits to society (Locher-Krause et al., 2017a, 2017b).

It has been stated that the ecosystem services approach has a great potential to understand socio-ecological systems in a multidisciplinary way, allowing the development of better planning in landscapes that provide multiple services (Bennett et al., 2009; Renard et al., 2015). Despite the importance of identifying ecosystem services and its beneficiaries –and the complex spatio-temporal mismatches between supply-demand— a clear knowledge gap exists in how to integrate beneficiaries needs and demands in landscape planning strategies (Bagstad et al., 2013a; Fisher et al., 2009; García-Nieto et al., 2013; Mach et al., 2015; Rodríguez et al., 2006). We seek to close this gap by combining an ecosystem service assessment with an analysis of beneficiaries across the landscape, which has been rarely done so far (Syrbe and Grunewald, 2017; Wei et al., 2017). Thereby, we want to answer the following questions: (1) how much do SCA contribute to forest ecosystem services supply, (2) where is the highest density of ecosystem services beneficiaries located across the landscape and which is their linkage with SCA?, (3) how much beneficiaries would profit if the SCA would be designated with a different protection status for further restoration.

5.3 Material and Methods

5.3.1 Study area

The study region covers an area of 16,625.7 km² and is located in Southern Chile, Northern Chilean Patagonia (73°20’ W-39°25’ S - 71°59’ W-41° 14’ S) (c.f. Figure 5.1). The region is characterized by high precipitation, with an average of 2,500 mm per year concentrated mainly in the winter season (June-September). The region shows a (temperate oceanic climate with a Mediterranean influence with an annual mean temperature of around 11.9ºC (CIREN, 1994; Di Castri and Hajek, 1976). The altitudes range between sea level in the West and 2500 m in the East, in the different geomorphological units: (1) the Coastal Mountain range (up to 900 m), (2) the Central Valley (up to 250 m), and (3) Precordillera and the Andes mountain range (up to 2,422m). Historically, the area went through an intense and diverse process of landscape transformation. Different type of historical and economic processes triggered these modifications as for example the convention of native forest to agricultural land during the European colonization back in the XVI-XVII century. In the XX century, the clearing of the native forest continued due to the boom of crop production (Armesto et al., 2010; Echeverria et al., 2006) and later on because of the fast expansion of exotic forest plantations since the 70s. All these processes led to a significant deforestation and reduction of biodiversity in Southern Chile (Lara et al., 2011; Nahuelhual et al., 2012).

The area has a population of circa 356.000 inhabitants (2.4% of the total national population) with a density of 19.3 inhabit/km^2. The population is mainly concentrated in urban areas (68.3%) compared to 31.7% of the inhabitants that live in rural areas. However, the study region has a high proportion of rural population compared to the national rate of 13%. From the total population of the region, 11.3% recognized themselves ethnically as Mapuches, Chilean indigenous inhabitants, considerably higher than the national average of 4.6%, living mainly in areas near to the Andes and the Coastal range. The poverty rate in the study area reaches 16.8% with an illiteracy rate of 6.9% (Casen, 2015). The primary economic activities are based on natural resources utilization from the forestry sector (pulp, shipyard and paper industry), the agricultural sector (livestock, crop, and berries production) and from the non-industrial fishing in the areas near to the seaside (INE Instituto Nacional de Estadística, 2012). The primary economic focus, oriented to natural resources extraction, have led to a degradation of the natural ecosystems and with it a decrease in the supply of benefits for the local society which depend on them. This economic focus has caused severe conflicts between production and nature conservation in the area (i.e., soil erosion due to poor management standards, river pollution, loss of biodiversity) (Locher-Krause et al., 2017b, 2017a).

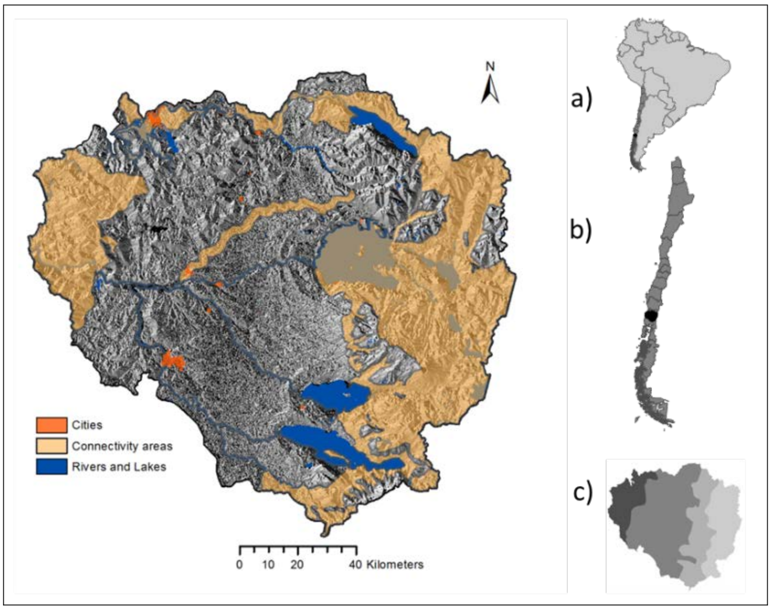

Figure 5.1 Study area location in South America (a) and Southern Chile (b). The map shows the SCA (see section 2.2.1) defined for the study as well as the most important geomorphological units (c): Coastal mountain range, Central Valley, Pre-Andes and Andes mountain range (from west to east).

5.3.2 Methods

We selected six forest ecosystem services that are recognized as critical in several studies in the area (Balvanera et al., 2012; Lara et al., 2009; Little et al., 2014; Locher-Krause et al., 2017a; Laura Nahuelhual et al., 2013a; Oyarzún and Hervé-Fernandez, 2015). The selected forest ecosystem services were: (1) carbon storage (regulation service); (2) plantation production (provisioning service); (3) phosphorous retention (regulation service); (4) nitrogen retention (regulation service); (5) aesthetic value (cultural service); and (6) forest recreation (cultural service). We use information derived in previous study Locher-Krause et al., 2017a to we estimate the contribution of ecosystem services supply to the SCA. Details of the approach used to model the ecosystem service supply can be found in Locher-Krause et al., 2017a.

In the case of ecosystem services beneficiaries, they were mapped to identify the demand side of forest ecosystem services at the regional scale (2.2.2). We used the ecosystems services previously selected for supply, except for nitrogen and phosphorous retention services which were merged as water regulation services. These water regulation services were the base to identify the drinking water beneficiaries in the area.

5.3.2.1 Defining structural connectivity areas (SCA)

We defined areas that potentially contribute to the structural and functional reconnection of the landscape based on vulnerable areas identified in previous research (Locher-Krause et al., 2017a, 2017b). We identified and assessed SCA as a first attempt to reconnect the highly fragmented landscapes in the case study region. The SCA include different types of landscape elements such as linear elements (riparian corridors) as well as patches of protected areas located in each mountain range, the Andes and Central Valley. Following this approach, we were able to connect the two largest native forest fragments effectively through the riparian zones. We integrated three protected areas located in the Andes (National parks Puyehue, Vicente Perez Rosales and National reserve Mocho Choshuenco) and three in the Coastal Mountain rage (National park Alerce Costero, National reserve Valdivia and the Private reserve Costera Valdiviana). To connect these protected units, we defined a 300 meters buffer area surrounding the main rivers that flow from the Andes range to the sea (San Pedro, Calle Calle, and Rio Bueno rivers). Also, two areas located in the Central Valley —the San Pedro river valley (658 km^2) and Llollenhue (246.3 km^2) — were integrated (c.f. Figure 5.1). These areas have been recognized as relevant to support ecological processes at the regional level and considered crucial as the potential habitat of endangered species (Centro de Estudios Agrarios y Ambientales, 2010, and Patricio Romero, personal communication). For the previously defined areas, we estimated the proportion of the different land cover classes and their trajectory over time (from 1985 until 2011) using the land cover information generated from a supervised Random forest classification of Landsat scenes (Locher-Krause et al., 2017b). To evaluate the relevance of protecting the SCA to allow regrowth of natural vegetation, we calculated the ecosystem services supply trajectory for the selected forest ecosystem services in both areas — SCA and remaining areas—, based on Locher-Krause et al. (2017a).

5.3.2.2 Mapping ecosystem services beneficiaries

We used different proxies and data sources to map potential ecosystem services beneficiaries as presented in (Table A.1 S1). These proxies were: (1) population density for carbon sequestration, (2) land ownership for timber/pulp provisioning, (3) upstream catchment area of each population center for water regulating services, (4) beneficiaries of the aesthetic value service identified based on a touristic viewshed analysis (Locher-Krause et al., 2017a), and (5) forest recreation beneficiaries (have been calculated based on the number of visitors).

For carbon sequestration, the population of the whole planet could be considered as service beneficiaries. From a regional perspective, all inhabitants of the region benefit from that service. Therefore, we estimated the beneficiaries by generating a local population density raster based on the urban and rural statistics from national and regional population census (INE Instituto Nacional de Estadística, 2012). For timber/pulp provisioning areas, we evaluated the number of exotic forest plantations by landholding ownership to identify the related main beneficiaries at the regional scale (Locher-Krause et al., 2017a; SII 2000). The upstream catchment area of each population center that supplies water to the river or streams nearest to each settlement in rural and urban areas was used as a proxy of the drinking water provisioning service (MOP, 2012). To be able to identify the locations of beneficiaries we integrated information on drinking water extraction points in the watershed (MOP, 2012) with the population density map. The ArcSWAT watershed delineation tool (ArcSWAT 2012.10.19) was used together with a digital elevation model (Lehner et al., 2008) to delineate the watersheds of the drinking water extraction points. The delineated areas were intersected with spatial information of rural and urban population density to address the number of people that benefit from each water extraction point per watershed. For our analysis, we excluded coastal areas (Figure 5.4) due to a lack of official data regarding rural drinking water sources.

The beneficiaries of the aesthetic value service were identified based on Locher-Krause et al. (2017a). This dataset aggregated spatial information of a viewpoint database from the national tourist agency (SERNATUR and GORE Region de Los Rios, 2014) and geo-tagged digital images from different photo-sharing platforms (Flickr, www.flickr.com, Google Earth, earth.google.com and Panoramio, www.panoramio.com) into a viewshed analysis in ArcGIS (ESRI, 2011). The forest recreation beneficiaries were estimated by the number of local and foreign visitors per year in each protected area (SERNATUR, 2004-2013). We considered only the national protected areas (national parks, national reserves, national monuments) due to the lack of public and continuous information about visitors of private protected areas.

5.3.2.3 Matching SCA and its beneficiaries

After mapping the distribution of ecosystem services beneficiaries across the landscape, we calculated the proportion of beneficiaries that profit from one of the selected ecosystem services in the total area, the SCA and remaining areas. For the forest exotic plantation and forest recreation services, we were able to distinguish different types of beneficiaries: private owners, companies, communities/cooperatives, and governmental institutions.

Areas in the proximity of densely populated or intensively production-oriented areas are at increased risk of being transform. Therefore, we also calculated service provisioning separately of SCA in a 300m buffer in the proximity of the SCA or around densely populated areas.

We identified hot and cold spots of ecosystem services supply and number of beneficiaries by a weighted overlay of the ecosystem services or beneficiaries' layers. First, each ecosystem service supply and each beneficiaries layer were scaled between 0 and 1. Percentiles were calculated for each normalized ecosystem service, and beneficiaries layer and all areas below the 20th percentile and above the 80th percentile were marked. Afterward, all ecosystem service layers and all beneficiaries' layers were intersected. For the hotspot analysis, the presence of individual services or beneficiaries above the 80th percentile in each of the remaining polygons were counted while for the cold spot analysis the presence of services or beneficiaries below the 20th percentile was counted.

5.4 Results

5.4.1 Spatial land cover distribution

The land cover distribution varied substantially across the SCA and remaining areas (Figure 5.2). The SCA were dominated by old growth forest (in the Andes and Coastal mountain ranges) followed by secondary forest. The situation was different in the remaining areas, in which grassland / arable land, shrubland, secondary forest, and exotic forest plantations were the most frequent land cover classes. Furthermore, large differences in land cover trajectory were identified in both areas (Figure 5.2). In the SCA, secondary forest increased over time, in contrast to the remaining areas where secondary forest primarily increased, and then decreased during the last two periods. Old growth forest declined in both SCA and remaining areas over time, but in the SCA, the magnitude of the loss was higher than in the remaining areas, especially between 1985 and 2004. Exotic forest plantations showed a steep increase over time, especially in the remaining areas.

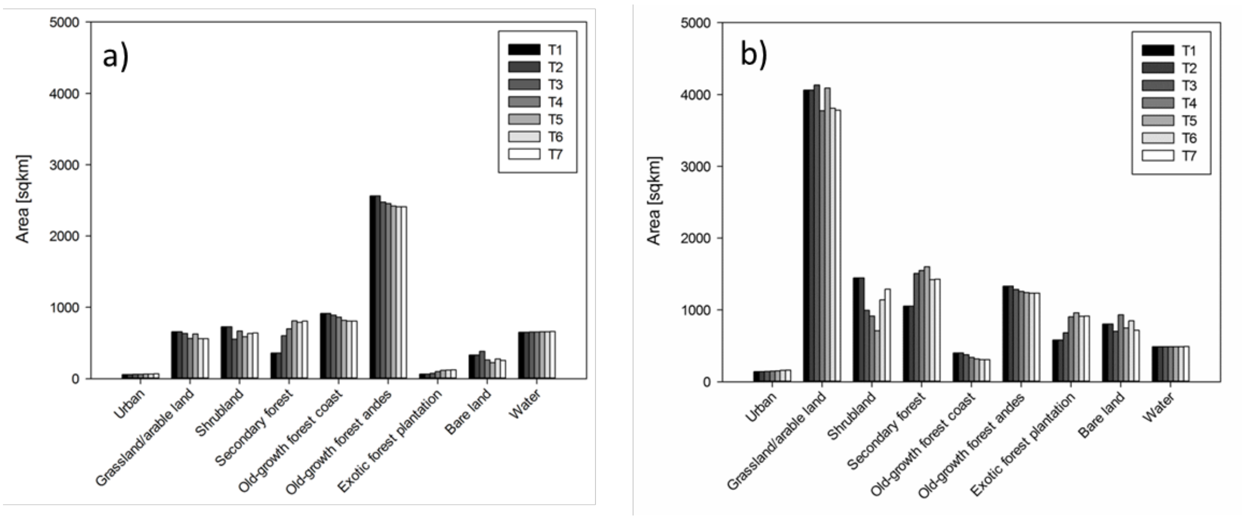

Figure 5.2 Distribution of land cover classes in each area: a) SCA, b) remaining area, and in the different periods: T1:1985-1986, T2: 1998-1999, T3: 2000-2001, T4: 2003-2004, T5: 2005-2006, T6:2008-2009, T7:2010-2011.

5.4.2 Ecosystem services supply in the SCA and remaining areas

The ecosystem services supply showed, both spatially and over time, an uneven trajectory in the SCA and remaining areas. Areas of high ecosystem services supply were not concordant between SCA and remaining areas. SCA showed higher ecosystem services supply values for carbon storage, sediment retention, aesthetic value, and forest recreation. In contrast, the plantation production index and phosphorous retention were higher in the remaining areas than in the SCA (Figure 5.3).

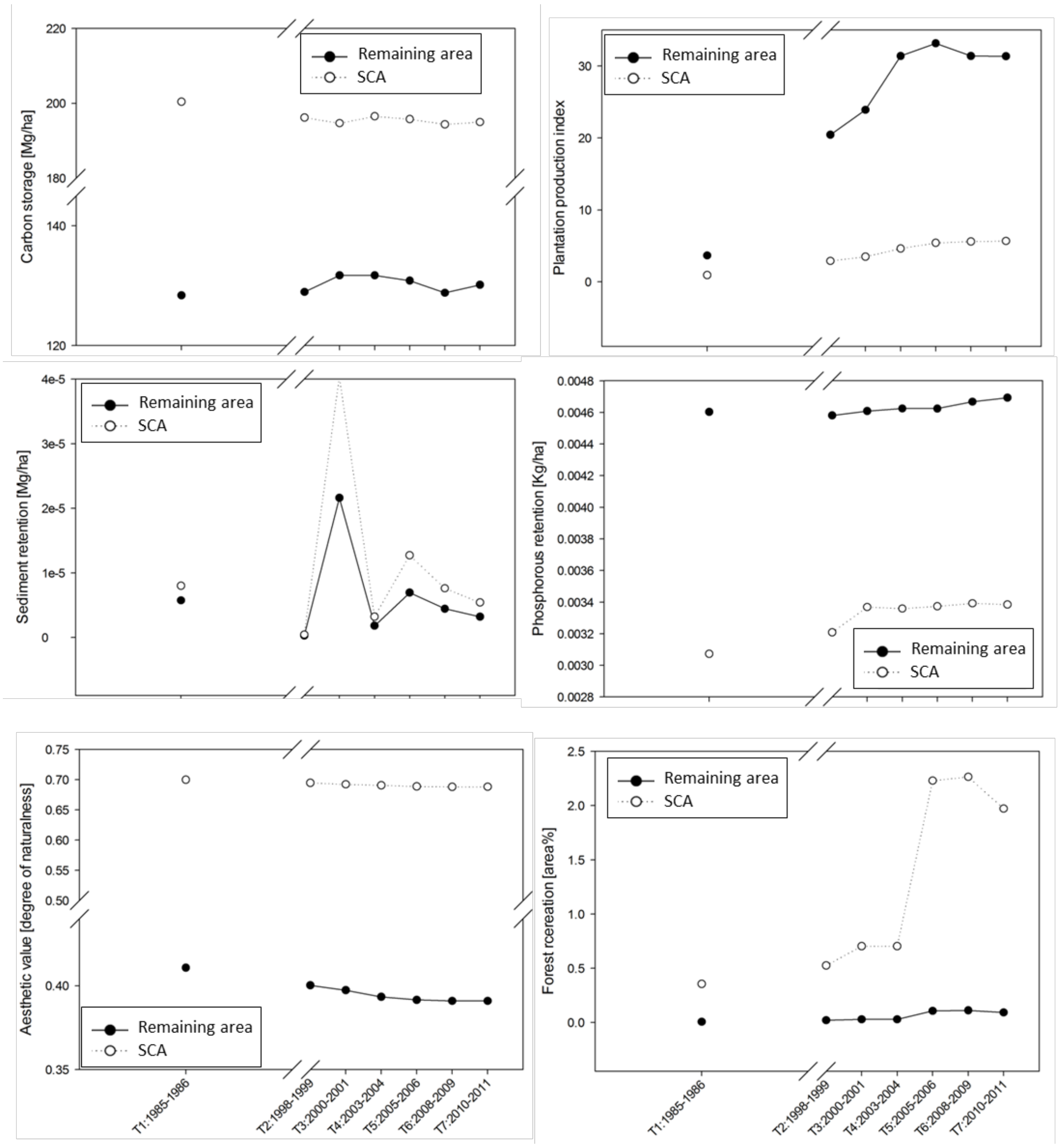

Figure 5.3 Ecosystem services supply trajectory over time in the SCA and remaining area: a) carbon storage, b) plantation productivity index, c) sediment retention, d) phosphorous retention, e) aesthetic value, f) forest recreation. The values represent arithmetic mean values for each area.

About 60% of the carbon storage service was provided by the SCA, with a slight fluctuation over time (Figure 5.3a). In contrast, the variation in the remaining areas was larger - likely due to the increasing of exotic forest plantations in the area.

Plantation production index differed strongly between both areas, with more than 80% of the exotic forest plantations service allocated in the remaining areas (Figure 5.3b). Exotic plantations and the services provided by them increased strongly over time in the remaining areas, especially from 1985 to 2006. This trend was also noticeable in the SCA, although at a much lower rate.

The supply of water quality regulation services (sediment and phosphorous retention) varied substantially in different areas (Figure 5.3c and d). Sediment retention revealed higher peaks in the years 2000-2001 (T3) and 2005-2006 (T5), with higher values in the SCA than in the remaining areas; with values that reached 60% of the total supply of sediment retention for the SCA. Phosphorous retention showed higher values in the remaining areas (approx. 60% of the total supply), with an increase over time for both areas.

For the cultural services aesthetic value and forest recreation the highest supply occurred in the SCA (Figure 5.3e and f). For aesthetic value, the areas with the greater provisioning values (approx. 63%) were allocated in the SCA with a slightly decreasing trend over time. Forest recreational areas were primarily located in the SCA and coincided with the location of the national protected areas (see methods section 2.2.2). Over time forest recreation service increased largely, especially in the period between 2003 and 2006.

5.4.3 Spatial allocation of ecosystem services beneficiaries

The distribution of ecosystem services beneficiaries differed between the different areas and for each ecosystems service (Figure 5.4). Carbon storage beneficiaries were mainly concentrated in the Central Valley, except for the densely populated northern part of the study area near the seaside) (Figure 5.4a). The mountainous regions showed a lower number of service beneficiaries.

Beneficiaries of water regulation (drinking water) and provisioning services were mainly located in the central-western part of the study area (Figure 5.4b). The coastal area could not be included in the analysis due to a lack of official data regarding rural drinking water sources. However, given the low population density of this area, it is unlikely that their inclusion would have changed the overall pattern.

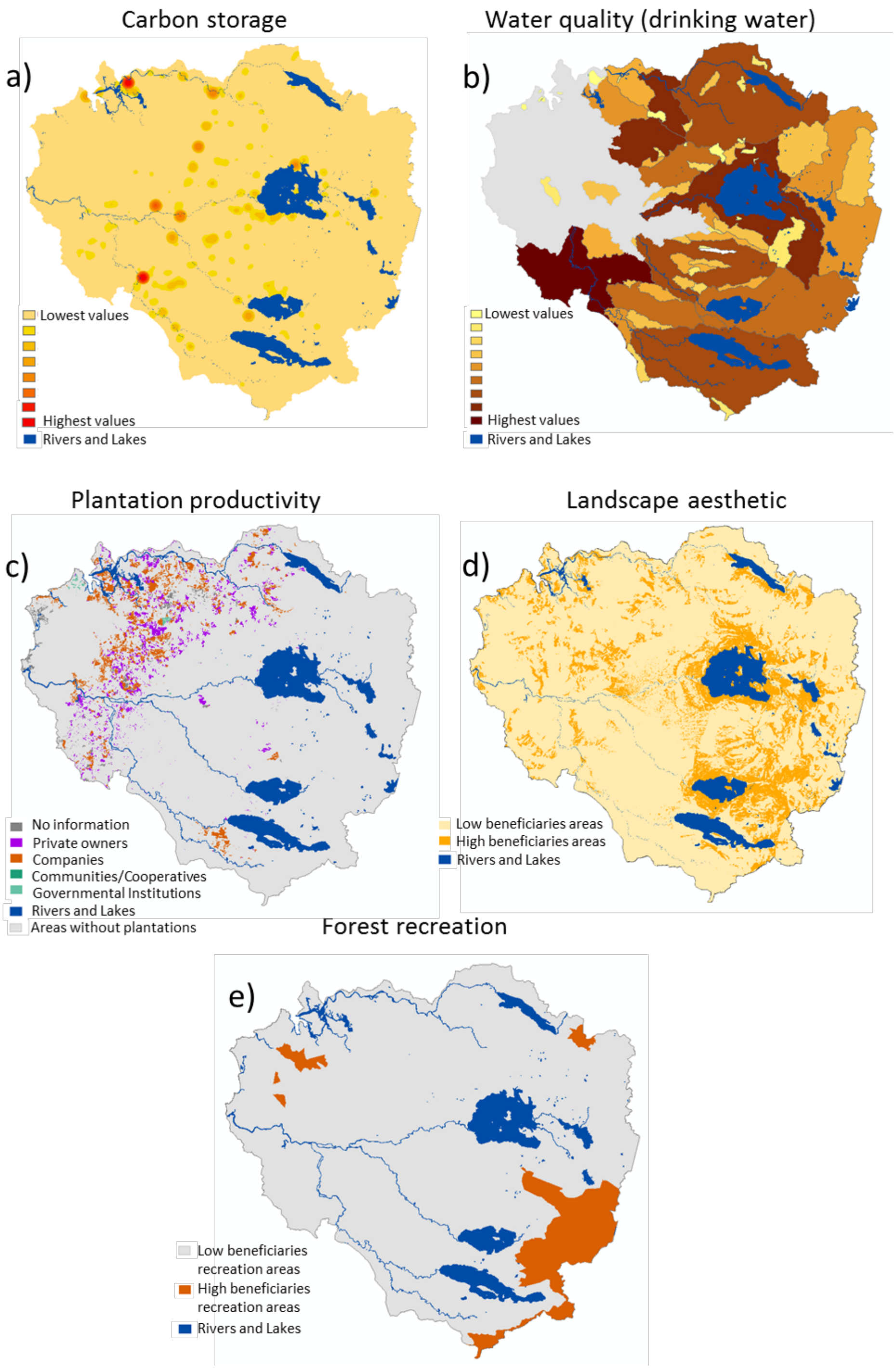

Figure 5.4 Spatial distribution of beneficiaries for the selected ecosystem services in the study area: a) carbon storage beneficiaries, b) drinking water beneficiaries, c) plantation productivity beneficiaries, d) landscape aesthetic beneficiaries, e) forest recreation beneficiaries. For carbon storage, drinking water beneficiaries, and cultural services high values indicate areas with high densities of beneficiaries of that services, and low values low density of beneficiaries. In the case of plantation productivity, the colors also show the different landholding ownership.

Plantation beneficiaries were almost exclusively concentrated in the Central Valley. However, some beneficiaries were also located in the proximity of the Coastal range, especially in the central-northern part of the study area. The resulting maps also reveal that forestry companies and private owners were the most frequent type of exotic forest plantations beneficiaries (Figure 5.4c). Beneficiaries of landscape aesthetics were highly concentrated in proximity to water bodies (lakes, rivers, see), mountains (especially volcanoes), and in areas in the central valley close to the main roads (Figure 5.4d). Forest recreational beneficiaries were spatially concordant to the national parks (Figure 5.4e) – mainly due to the fact that it is illegal to access private areas for recreational purposes in Chile.

5.4.4 SCA and beneficiaries

Similarly, to our ecosystem services supply results, ecosystem services beneficiaries were unevenly distributed across SCA and the remaining areas. Also, spatial patterns of service supply and related beneficiaries differed clearly. For carbon storage beneficiaries, the mean population density was slightly higher in the SCA (27.3 inhab/km^2) compared to the remaining areas (26.4 inhab/km^2). However, SCA accounted for a much smaller fraction of the total study area with only 40% of the total. Figure 5.5 depicts that the areas in the SCA most likely to face land use change due to urban and forest plantation/agricultural expansion also showed a high number of beneficiaries. These areas were distributed mainly in the North-West and South-West of the region. Other areas likely to be transformed were allocated in the central areas and the proximity of the big lakes.

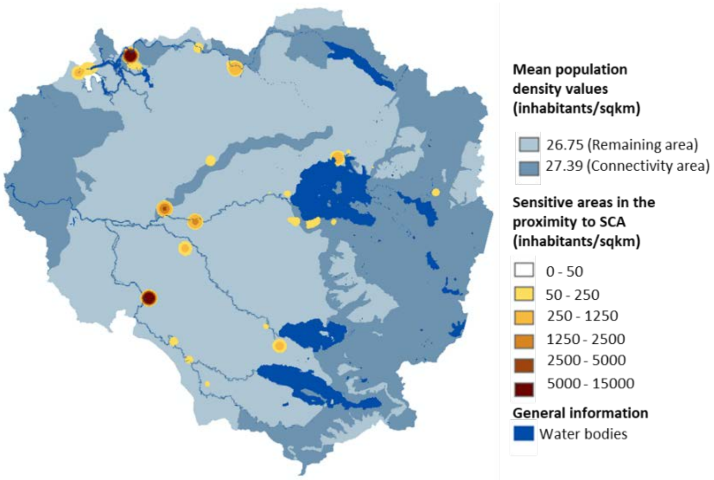

Figure 5.5 Distribution of the carbon storage beneficiaries in the SCA and remaining areas, the values represent the mean values obtained from the population density values for urban and rural areas. The sensitive areas located in the proximity of the SCA are shown.

The location of forest exotic plantation beneficiaries coincided with the location of high supply of the services (long distance beneficiaries where not include it in this study). In the SCA, the forestry companies owned most of the plantation areas, followed by the private owners and regions in which the owner could not be identified (Figure 5.6). These results were similar for the remaining areas in which the forestry companies owned more than 50% of the total plantation areas, followed by the private owners (38 %). Areas with a high density of beneficiaries in the sensitive areas were mainly located in the North-West area near to the seaside.

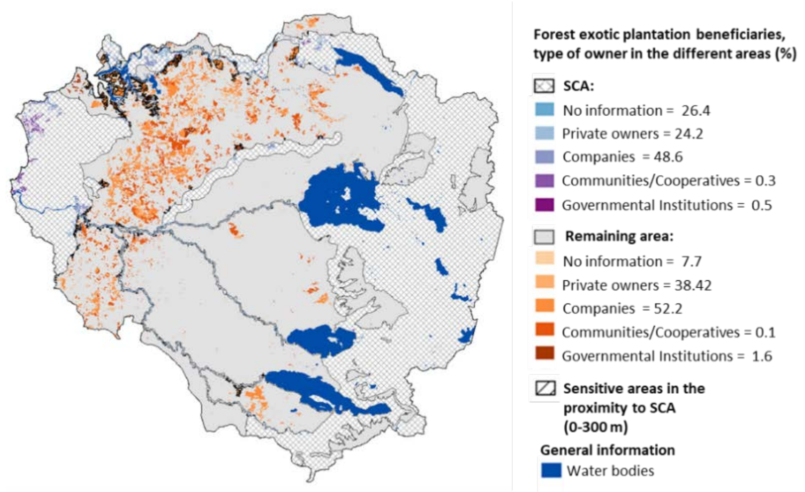

Figure 5.6 Distribution of the forest exotic plantation beneficiaries in the SCA and remaining areas, the values represent the percentage of types of owners in each area. The sensitive areas located in the proximity of the SCA are shown (% of the remaining areas).

Drinking water beneficiaries showed a different spatial pattern than the supply the regulation of ecosystem services (Figure 5.3). In the SCA, the beneficiaries of the drinking water services were mainly located in a small portion of the Central-East study area and the riparian zones in the Central Valley. For the remaining areas, the areas with more beneficiaries were located in the Central Valley, especially in the South-West and in the surrounding of the riparian zones that belong to the SCA (Figure 5.7). Areas with a high density of beneficiaries in the proximities of the SCA were mainly concentrated in the riparian zones located through the Central Valley and in the North-West side of the study area.

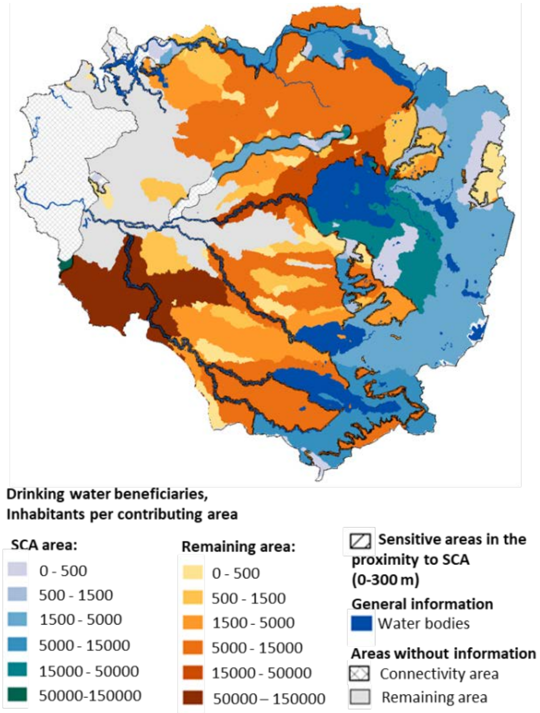

Figure 5.7 Distribution of the drinking water beneficiaries in the SCA and remaining areas, the values represent the mean population density values in each area. The sensitive areas located in the proximity of the SCA are also shown.

The beneficiaries of aesthetic value services were mainly distributed in the remaining areas, close to populated zones in areas with high landscape beauty. Also, a high number of beneficiaries were located near water bodies (lakes, rivers and see), volcanoes and mountains, especially in the proximity to the Andes mountain range (Figure 5.8). However, some of these beneficiaries were also located in the SCA, especially in the central valley in the proximity of the riparian zones. For this service, sensitivity areas were distributed mainly in three areas: in the surrounding of the Andes Mountains (close to the lakes), the Central Valley in the proximity of to the SCA and the northwestern part of the study area.

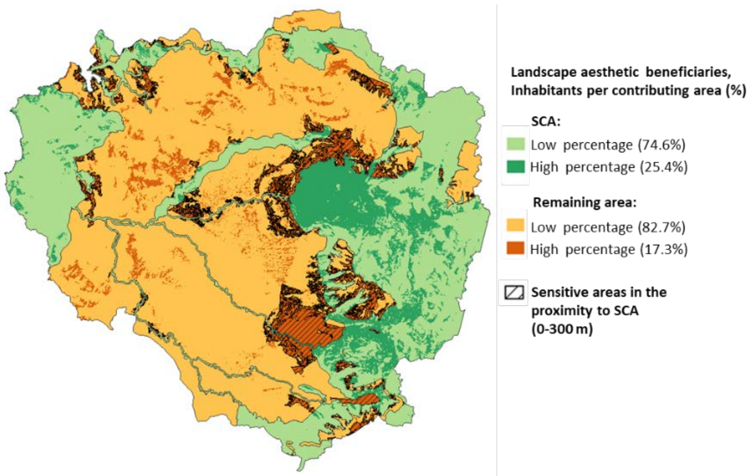

Figure 5.8 Distribution of the landscape aesthetic beneficiaries in the SCA and remaining areas, the values represent the percentage of inhabitants in each area. The sensitive areas located in the proximity of the SCA are also shown.

For forest recreation beneficiaries, Figure 5.9 shows the number of people that accessed the protected areas during the year 2011. The highest number of beneficiaries of forest recreation service accessed the regions located in the Andes mountain range, considering national and foreigner visitors (Figure 5.9a). Concerning the rates of changes, the two more visited protected areas - P.N Vicente Perez Rosales and P.N. Puyehue - showed a different trend over time, especially during the last periods (Figure 5.9b). Although both protected areas experienced an abrupt decrease in the number of visitors after 2009, in the Vicente Perez Rosales P.N the number of visitors increased after 2010 in contrast to P.N. Puyehue that showed a steady decrease over time.

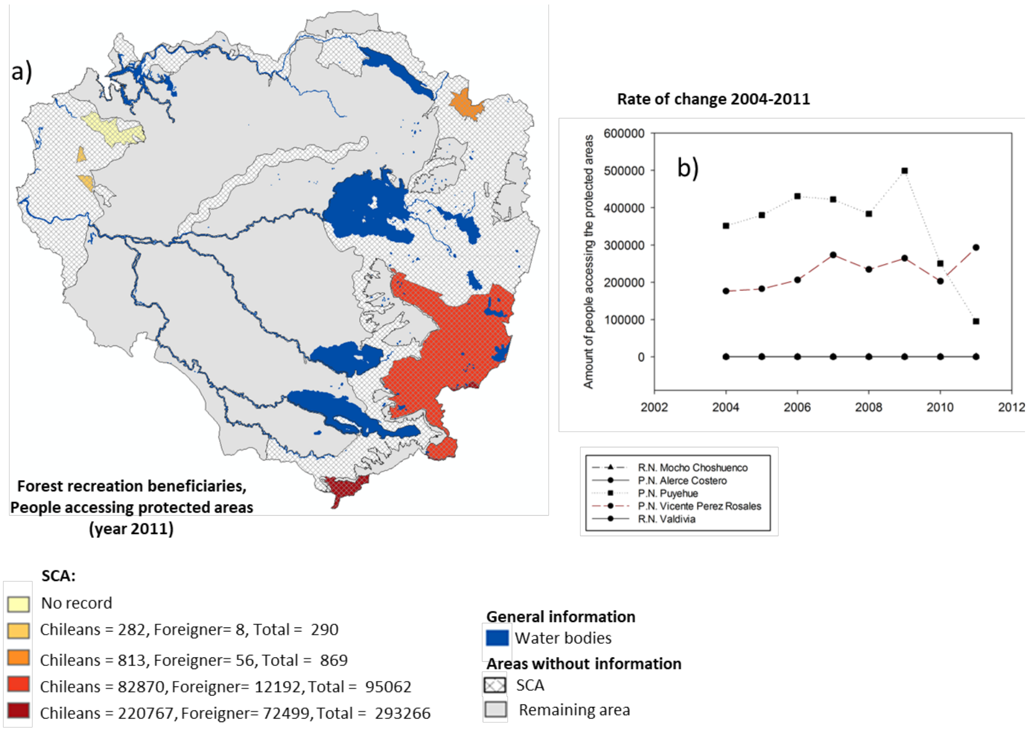

Figure 5.9 Distribution of the forest recreation beneficiaries in the SCA and remaining areas, the values represent the number of people which have had accessed these areas. The rate of change of SCA over time is also shown (the areas that are not shown in the graph had no record of visitors as well as not change over time). The different colors also showed the different national parks, dark red: P.N. Vicente Perez Rosales, red: P.N. Puyehue, dark orange: R.N. Choshuenco, orange: P.N. Alerce Costero, yellow: R.N. Valdivia.

5.4.5 Ecosystem services supply and beneficiaries

The spatial distribution of areas with a high supply of ecosystem services and a high density of beneficiaries differed between the geomorphological units (Figure 5.10). In the Andes mountain range, we identified large continuous areas with higher values in comparison to the Central Valley. In contrast to the Coastal range in which areas with higher values were more dispersed (Figure 5.10). Regions with a higher supply of ecosystem services and a high number of beneficiaries were more frequent in the SCA (54%) than in the remaining areas (46%). Areas with high supply and high beneficiaries were mainly located in the Andes and Coastal ranges. Other areas with higher values were located in the riparian zones, especially in the southern and northern part of the study area.

In the remaining areas, hotspots of service supply and beneficiaries were concentrated mainly in patches in the northern and western regions of the Central Valley. Areas with medium values were identified in the southern (Central Valley) and the eastern (Andes range) part.

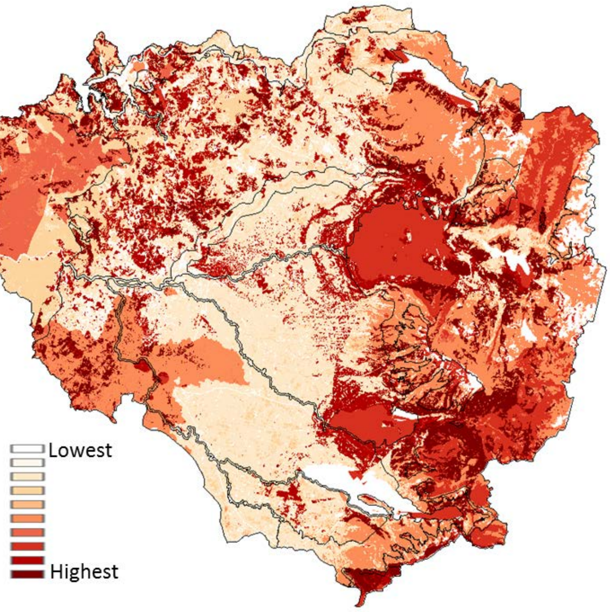

Figure 5.10 Map of the spatial distribution of areas with high values for ecosystem services supply and a high number of beneficiaries across the study area. High values areas indicate maximum values for the supply of six ecosystem services and the highest density of its beneficiaries in the upper 20th percentile, after their normalization.

5.5 Discussion

5.5.1 Ecosystem services supply and its beneficiaries

The spatially explicit identification of ecosystems services beneficiaries at the local scale has been recognized as a critical element to understanding socioecological systems (Bagstad et al., 2014; Bennett et al., 2015; Syrbe and Walz, 2012). However, evident gaps persist in the process of mapping, essential metrics definition, and assessment of the interconnection among ecosystem services and their beneficiaries at the landscape scale (Rieb et al., 2017). Our approach using spatially explicit data and simple GIS-based models attempt to uncover the interlinkages among ecosystems services and its beneficiaries in the area. The integration of different types of approaches and data allowed us to understand the low data availability area but is associated with challenges such as higher data uncertainty.

As the validity of the results widely depends on the quality of the data used for the beneficiaries' assessment, we cross-checked information from different agencies and performed an exhaustive literature review to ensure the consistency of our findings (National Water Agency, DGA; National Statistical Institute, INE; National Tourism Service, SERNATUR). In the case of ecosystem services supply the data was already validated in Locher-Krause et al., 2017a. The results of the beneficiaries' assessment were in line with socioecological information available for the area (DGA, 2017; INE Instituto Nacional de Estadística, 2012, 2007; SERNATUR and GORE Region de los Rios, 2014). The most important challenge to identify the different beneficiaries of the services was the low temporal scale of socioeconomic data in Chile. Although the national population census and socioeconomic statistics are updated every ten years, we used information of the population census from 2002 (for rural and urban areas) as the last reliable census of the country. The release of the recently conducted census (2017) in 2018 will provide additional data. For drinking water beneficiaries missing information / poor-quality data from the Coastal range limited our results to the other geomorphological units.

Future research activities should integrate socio-cultural techniques such as interviews or questioners as an additional source of information to expand the understanding of the demand side of the system (Harrison et al., 2017). To better link management strategies and the interest/values of the different stakeholders, participatory approaches are a promising option (Gómez-Baggethun and Naredo, 2015; Hein et al., 2006; Jax et al., 2013).

The results showed a substantial spatial variation and mismatch between ecosystem services beneficiaries and supply areas. This supply-demand mismatch reveals to be particularly crucial for regulating ecosystem services in the area — mainly carbon storage and drinking water. In the case of the drinking water and forest plantation beneficiaries, both were concentrated primarily on the Central Valley depending directly and indirectly on the water resources coming from the Andes range. In our study area, several studies have reported continuous and abrupt changes in water availability, especially during the summer season (Hervé-Fernández et al., 2016; Huber et al., 2008; Little et al., 2009; Oyarzun et al., 2007). These changes in water availability are especially alarming when we consider that the most substantial number of drinking water beneficiaries overlap with or are located in the proximities of areas used for forest exotic plantation production, which highly increased over time (Figure 5.2). The rise of trade-offs among drinking water demands and utilization for exotic forest plantations production on the other side have been widely reported as a severe problem in various countries and ecosystems (Comte et al., 2012; Grau et al., 2013; Wallace et al., 2003; Young et al., 2005). For instance, Zhang et al., 2015 report a decreasing water availability in dryland areas caused by afforestation restoration strategies. The different spatial patterns of supply and demand for drinking water provisioning shows that the valuation of ecosystems has to consider distant beneficiaries. The uneven spatial distribution of beneficiaries across the landscape indicates the relevance of ecosystem services assessments to identify potential conflicts caused by locally high demands for specific services (Balvanera et al., 2012; Bennett et al., 2015).

While our results showed that the most cultural ecosystem services beneficiaries were located in the Central Valley (Figure 5.4), its supply was provided by the SCA that were mainly distributed in the Andes and Coastal range (Figure 5.8, Figure 5.9). This spatial mismatch corroborates the importance and big potential of planning strategies oriented to restore degraded riparian areas (SCA among the two ranges) in zones with a high density of beneficiaries. Furthermore, Martinez-Harms et al., 2018 emphasized inequalities with regard to access to cultural services in Chile. They provide evidence that wealthier citizens travel longer distances to protected areas while people with lower incomes tend to visit protected areas closer to their homes. This suggests that integrating riparian zones as SCA would provide access to recreational sides for lower-income beneficiaries.

5.5.2 The role of SCA for landscape planning and management

Our analysis showed that SCA supplied more than half of the ecosystem services compared to the remaining areas –particularly for regulating and cultural services– while it accounts only 40% of the total study area (Figure 5.3). Likewise, beneficiaries of the services were highly concentrated nearby or in the SCA (Figure 5.4). The high supply of ecosystem services together with the high concentration of population in the SCA highlight the relevance of a suitable landscape planning strategy to avoid natural resources deterioration. The protection of the SCA seems, therefore, a promising landscape management option to maintain ecosystem services supply and potentially also biodiversity. As has been shown in conservation areas in Costa Rica, this strategy helps not only by upsizing protected areas but also to increase the flow of material and organisms through the landscape (Pringle 2017).

In our case, protecting SCA would also reconnect the two largest relicts of old growth forest, located in the Coastal and the Andes range. Reconnecting these two relicts offer the opportunity to maintain and enhance areas that deliver a high amount of ecosystem services, especially regulating services —carbon and water-related services— but also critical cultural services. In addition, the protection of SCA in the Central Valley with a different protection status —focused on the restoration of the riparian areas— offers the potential to connect a highly fragmented landscape characterized by patches of secondary forest distributed across the landscape (Locher-Krause et al., 2017b; Zamorano-Elgueta et al., 2015).

Based on the current need to maintain and enhance ecosystem services supply and at the same time ensure the best balance among conservation and nature resources utilization, questions such as where and how to prioritize investments are crucial. Landscape planning strategies that start by identifying current protected areas and the potential for connecting them across the landscape should be a priority. Likewise, the definition of areas in which intensive productive activities area restricted is crucial for the maintenance of the flow of ecosystem services to their beneficiaries across the landscape. Furthermore, watershed/soil protection programs that limit inappropriate management activities in vulnerable areas (i.e., riparian areas, slopes) primarily in the upstream watershed areas improve and sustain the drinking water quality and quantity (Guerry et al., 2015).

5.6 Conclusions

The current study has shown that ecosystem services provision can be enhanced by the integration of SCA, especially if we consider that this area contributes to more than half of the total ecosystem services of the area and account only for 40% of the area. Furthermore, the beneficiaries' assessment also highlights how important SCA are for the population. Our study shows that 41% of the ecosystem services beneficiaries depend directly on these SCA to obtain benefits from the selected forest ecosystem services. This percentage rise if we consider services like drinking water which depends on conservation areas located in the Andes range. This crucial information is frequently missing in current ecosystem services assessment, in which the interlinkage among the ecosystem services supply and its beneficiaries it is rarely included (Bagstad et al., 2014; Syrbe and Grunewald, 2017; Wei et al., 2017).

In our case study, SCA were of high importance from the perspective of the supply of services and the perspective of the density of beneficiaries. Nevertheless, the identification of the mismatch between the location of service supply and related beneficiaries is necessary for future landscape planning, especially in cases when the beneficiaries are in a different location as the services supply as in the case of drinking water services.

From our research, it is also possible to conclude that landscape strategies oriented to improve conservation in the region could benefit from using our approach. This approach of enhancing landscape connectivity by starting with the already protected areas and integrating currently degraded landscape elements such as riparian corridors could provide essential benefits to improve the vulnerable state of temperate rainforest in the area. Restauration initiatives as well as the implementation of conservation categories that allow a less intense and more sustainable use of the degraded riparian corridors located in the Central Valley will help to ensure ecosystem services supply in the area.

While our results help to address the gap of the interconnection between ecosystem services supply and its beneficiaries these results are not transferable to other regions and other scales. Consequently, we want to highlight the relevance of our SCA as a landscape planning strategy especially under the ecosystem services approach to understand the interconnection between supply and its beneficiaries better as well as maintain the flow of services across the landscape.

5.7 Acknowledgments

We would like to thank Patricio Romero, Prof. Dr. Sandoval, and Veronica Oyarzun for valuable information. Karla Locher-Krause was supported by a Becas Chile-DAAD pre-doctoral fellowship (Government of Chile and the German Academic Exchange service – A1086435). Sven Lautenbach acknowledges support by the Klaus Tschira Stiftung.

6 Conclusions and Outlook

6.1 Dissertation contribution

In this research high temporal resolution datasets of land cover dynamics and ecosystems services (supply and its beneficiaries) have been produced for a hotspot of biodiversity area in Southern Chile. This information allowed the identification of the trajectory and the effects of the human impact on ecosystem services and biodiversity in this region. The integrated approach that combines remote sensing, modeling techniques and methods from different disciplines such as social and natural sciences captured the land cover dynamic, ecosystem services supply trajectory and its beneficiaries across the landscape. Results revealed that land use and cover change in the area is highly dynamic and indicate a human appropriation at different spatio-temporal scales. For example, the highly productive areas in the central Valley showed a dynamic transformation to exotic forest plantation areas, with an increase in provisioning services but a decline in the supply of regulating ecosystem services. This situation can impact the availability of the system to maintain ecosystem services functions in the future.

Main past and ongoing trends of deforestation, afforestation, fragmentation and ecosystem services supply together with its beneficiaries were successfully identified and analyzed using a step by step approach that is summarized below. The first step of the procedure started is a landscape analysis at an expanded temporal resolution (Chapter 3) and continued with the ecosystem services modeling assessment (supply and its beneficiaries) over time and across the landscape (Chapters 4 and 5). This base information allowed the development and evaluation of a recommended landscape planning strategy seeking for a comprehensive investigation of the area (Chapter 5).

The conclusions of the related work are discussed in the following subsections.

Expand the temporal resolution of landscape transformations in a biodiversity hotspot area.

Chapter 3 uncovered the land cover dynamics in the area by expanding its temporal scale. This extended temporal resolution —with seven periods, from 1985-2011, using all Landsat scenes available for the area— revealed a highly dynamic and disturbed area, dominated by processes such as clear-cuts of exotic forest plantations, regrowth of secondary forest, afforestation with exotic species, and deforestation of native forest. The hotspot of biodiversity area evidenced a dynamic spatio-temporal transformation of both land cover composition and configuration. According to the random forest and remote sensing analysis (Chapter 3) landscape composition shows a clear and drastic increase of exotic forest plantations in the area, especially from 1985 until 1999 with the highest net change for a single class of 706% in 26 years. All land cover classes in the area transitioned in some proportion to exotic forest plantation. However secondary forest, shrublands, grassland/arable land are the ones that had the highest area loss, followed in lower proportion from the old growth forest. Old growth forest showed a continuous decrease over time, with the highest deforestation rate of 1.2% (net loss) between 1985 and 1999. This deforestation rate tended to slow down in the last study period. Land cover classes as secondary forest showed an increase over time especially coming from shrublands areas, which is part of the natural process of succession. However secondary forest also gained area from old growth forest, a clear sign of forest degradation. Shrublands showed a more mixed trend mainly due to the fact that these areas can be easily changed to more productive land use classes. Moreover, the procedure for shrublands classification shows higher uncertainties due to the difficulty to differentiate between closer successional stages as is the case of shrublands and secondary forest. According to the calculated landscape indices, fragmentation increased over time, with a decrease of large patches and an increase of the total edge length especially in the period from 1985 -1999. After that, the old growth forest patches remain more continuous, but fragmentation at the edges of smaller forest patches is observed, leading to a continuous loss of the core area crucial to maintaining ecological functions (i.e., regulation of microclimatic conditions) and species habitat. In the case of secondary forest, even when the results of Chapter 3 revealed an increase of this land cover class, landscape indices showed a high fragmentation with smaller and disconnected patches. These secondary forest patches are mainly located in the central valley, an area with high pressure due to human-influenced activities and where most of the settlements are located (Chapter 5). Furthermore, results showed a highly fragmented secondary forest located mainly in areas used for intensive production. This situation uncovered the stage of degradation of this type of forest which has crucial relevance to maintain forest ecosystems in the area. To summarize, the data analysis revealed an irregular, but intense deforestation and fragmentation process linked both to the old growth forest and the secondary forest. The results of this chapter highlight the advantage of expanding the temporal resolution of landscape studies, allowing the identification of internal trajectories of land use dynamic and providing baseline data for the following step.

Spatio-temporal assessment to understand the dynamics of multiple ecosystems services supply across the landscape.

On the base of the work done in Chapter 3, Chapter 4 documented the biophysical and socioeconomic data analysis revealing the dynamic and modification of the ecosystems services supply in the area. The model and proxies used confirmed the diverse pattern of ecosystem services supply depending on the specific temporal (7 periods), and spatial scale (geomorphological units) assessed. The differentiation between the four main geomorphological units, Coastal mountain range, Central Valley, Pre-Andean and Andes mountain range allowed the uncovering of contrasting patterns within the study area. In the case of carbon stocks, there are contrasting trends in the different geomorphological units, with higher stocks in the Andes and the Coastal mountain range.

Despite its higher stocks, the supply decreased in these areas (the Andes and Coastal range) over time in contrast to the clear increase of the provision of this service in the Central Valley, mainly linked to deforestation and afforestation processes identified in Chapter 3. Likewise, the calculation of the plantation productivity index, which represents a provisioning service, revealed a strong increase with a relative change of 370 and 900% (1985 to 2011) in the Central Valley and Coastal range units. The quantification of the irregular trends reported mainly for regulation ecosystem services as such sediment and phosphorous retention underline the importance of vegetation in the riparian areas, especially forest. The forested areas identified in chapter 3 showed higher values of regulating services and play a crucial role to maintain the supply in the area, particularly in the Central Valley (chapter 4). The cultural ecosystem services modeled in chapter 4 showed an increase in forest recreation services supply over time. However, these areas are isolated and mainly located in regions with low accessibility in the Andes and the Coastal range. In addition, aesthetic supply areas decreased over time; if we consider that this service is based on the degree of naturalness of the area, this result indicated a higher impact of human activities and impairments due to economic pressure.

In summary, the land use change and fragmentation information generated in chapter 3 set the base for modeling of the trajectory of ecosystem services supply produced in chapter 4. Furthermore, the ecosystem services assessment provided clear guidelines for the development of land use planning recommendations to protect and enhance areas recognized as vulnerable (Chapter 4). These areas are mainly riparian buffers close to streams and also areas in both mountain range which showed a high amount of services but that are clearly disconnected and isolated in the landscape. Recommendations such as the reconnection of this highly fragmented landscape to improve the management of multiple ecosystem services in the area is a topic discussed of the next step (chapter 5).

Ecosystem services beneficiary's assessment integrating landscape management strategies to reconnect fragmented ecosystems.

Chapter 5 assessed the recommendations derived from chapter 4, by analyzing the potential contribution of a landscape planning strategy — areas with differentiated protection status oriented to reconnect the landscape— to ecosystem services supply and its beneficiaries. These structural connectivity areas (SCA) were defined as the integration of linear (riparian corridors) and patchy (national conservation units) landscape elements. The combination of socioeconomic data and proxies together with the ecosystem services supply data modeled in chapter 4 showed a clear spatial mismatch between ecosystem services and its beneficiaries across the landscape. This mismatch uncovered the relevance of identifying potentially threatened areas caused by high demands of services such as provisioning and regulating services, which lead to a decrease in the supply of the service. Results from chapter 5 confirmed that SCA have the potential to improve ecosystem services supply and biodiversity conservation in the area. SCA showed higher values for the supply of regulation (60%) and cultural ecosystem services (63%) in comparison to the remaining landscape, even when only account for 40% of the total study area.

Additionally, local data used to identify ecosystem services beneficiaries also showed that in or in the proximities of the SCA is where the highest number of the beneficiaries are located.

The trends identified in chapter 5 emphasize the findings towards the relevance of the SCA as an area that supplies a high amount of services but is under high pressure which could impair future ecosystem functioning.

On the other hand, the results showed that provisioning ecosystem services are "offered" mainly by the remaining areas instead of being in the SCA. Moreover, SCA are also showed to be an effective planning strategy oriented to reconnect the two main old-growth forests in the area (in the Andes and the Coastal range) to ensure genetic flow between them. Hence, the data analyzed in chapter 5 suggest that planning initiatives oriented to assign different protection categories to promote restoration of native forest (old growth and secondary forest) in riparian areas could help to maintain the flow and supply of ecosystem services in the area. Furthermore, the results of this dissertation confirm the assumption that conservation initiatives aiming at enlarging existing conservation units highly contribute to conservation minimizing the socioeconomic cost of creating new units.

6.2 General conclusion and final remarks

The main objective of this dissertation was to improve the understanding of the impacts and effects of human activities in a biodiversity hotspot area by expanding the temporal resolution of land cover and ecosystem services assessments. The study used a step-by-step approach integrating different types of data and methodologies benefiting from the current technological advances (i.e., the opening of the Landsat archive, improved computational techniques) seeking to close knowledge gaps in an area highly relevant for conservation. Furthermore, the increasing availability of big data such as earth observation products (chapter 3) together with extended biophysical and socioeconomic databases (chapter 4 and 5) increase the potential to support the design of sustainable policies (chapter 5). However, the related advantages and big opportunities also come with new challenges. For instance, uncertainties arising from the quantity and quality of local data especially regarding hydrological monitoring were one of the difficulties for mapping water-related ecosystem services. Likewise, the low temporal resolution of local socioeconomic data decreased the timeframe to understand historical processes, dismissing the projection of future scenarios. Future policy strategies oriented to the implementation of continuous and more detailed monitoring programs to expand the spatio-temporal resolution of the local data such as hydrological databases, nutrients, species, etc. will play an essential role in estimating the impact of uncertainties on the accuracy of environmental impact analysis. Such monitoring systems should focus not only on

biophysical data but also on socioeconomic information to enhance the knowledge of their interconnection. This type of information is particularly relevant to balance society needs and its impacts in biogeographical areas described as biodiversity reservoirs as the Valdivian temperate rainforest.
Additionally, landscape planning strategies designed to manage the territory in a more integrated way with the commitment of the different agencies working on it together is urgently needed, especially in terms of protecting ecologically relevant areas. Ecologically relevant areas such as riparian corridors, steep slopes, and areas with a high amount of endangered species among others should have stricter, legally bounding protection standards, limiting unsuitable types of management (i.e., large clear cuts) to avoid environmental impacts. Moreover, the landscape planning recommendations as the one assessed in chapter 5 could greatly benefit not only from implementing the ecosystem services concept but also from expanding existing conservation units such as national parks, by connecting the riparian areas that are vulnerable to human impact considering different protection categories.

One challenge that came clear from this research was the uncertainties associated with some of the model simplifications, especially for those models simulating biophysical processes (see chapter 4). These simplifications were related to the type of the model selected for this study (InVEST) that requires a moderate amount of input data, appropriate for regions with data scarcity. Further improvements for future research may be achieved by using more intensive data-driven models once data availability has been improved in the area. Although this dissertation shows the magnitude and trajectory of human impacts in the region and provides landscape planning recommendations, the study does not include a more extensive selection of ecosystem services. Moreover, the selection of the modeled ecosystem services was based on literature reviews, expert consultations and the personal knowledge of the researcher. Further research should involve policymakers and stakeholders already in the process of selection of ecosystems and their services, expanding the type of services and data to be considered. The involvement of different types of stakeholders could contribute to integrate different types of knowledge such as traditional knowledge and different cosmovision as the one from Mapuches, one of the indigenous population from southern Chile. Moreover, as is reported in chapter 4 and 5 ecosystem services mapping and quantification could support the design of mechanisms towards an integrated landscape planning, considering the multifunctionality of these systems under current climatic scenarios.

A List of Articles prepared during the preparation period of the thesis

During the preparation period of this thesis, the following scientific papers were prepared and partially published.

Locher-Krause, K. E., Volk, M., Waske, B., Thonfeld, F., & Lautenbach, S. 2017. Expanding temporal resolution in landscape transformations: Insights from a Landsat-based case study in Southern Chile. *Ecological Indicators*, 75*, 132-144.

Locher-Krause, K. E., Lautenbach, S., & Volk, M. 2017. Spatio-temporal change of ecosystem services as a key to understand natural resource utilization in Southern Chile. *Regional Environmental Change,*
1-17.

Cord, A.F., Bartkowski, B., Beckmann, M., Dittrich, A., Hermans-Neumann, K., Kaim, A., Lienhoop, N., Locher-Krause, K., Priess, J., Schröter-Schlaack, C. and Schwarz, N., 2017. Towards systematic analyses of ecosystem service trade-offs and synergies: Main concepts, methods and the road ahead. *Ecosystem services*, 28*, pp.264-272.

Locher-Krause, K., Volk, .M & Lautenbach, S. *In Review*. Spatio-temporal change of ecosystem services as a key to understand natural resource utilization in Southern Chile. *Ecosystem services*

B Presentations of research related to this thesis at scientific conferences

Oral presentations

Locher-Krause, K. E., Volk, M & Lautenbach, S. 2014. Spatial/temporal analysis of forest ecosystem services in the south of Chile. XXIV IUFRO World Congress, October 5- 11 2014, Salt Lake City, USA

Locher-Krause, K. E., Volk, M & Lautenbach, S. 2014. Using a continuous dataset to understand the dynamic transformation of native ecosystems over time in the South of Chile. 9th IALE World Congress, July 5-10, Portland, Oregon, USA.

Locher-Krause, K. E., Volk, M & Lautenbach, S. 2016a. Reconnecting landscapes: how to connectivity areas impact ecosystem services supply and its beneficiaries in Southern Chile. GEO BON Open Science Conference & GEO BON All Hands Meeting, July 4-8, Leipzig, Germany

Locher-Krause, K. E., Volk, M & Lautenbach, S. 2016b. Spatio-temporal change of ecosystem services: a key to understand natural resources utilization towards balance human well-being and nature. Second IALE Latin-American Conference IUFRO Landscape Ecology Latin-American, 28 November-2 December Temuco, Chile.

Locher-Krause, K. E., Volk, M & Lautenbach, S. 2017. Quantifying and understanding spatio-temporal ES interaction in transformed landscapes: insights from a Patagonian biodiversity hotspot. IUFRO 125th Anniversary Congress, Interconnecting Forest, Science and People, 18-22 September 2017, Freiburg, Germany.

Locher-Krause, K. E., Volk, M & Lautenbach, S. 2018b. ES supply and beneficiary's assessment: the role of SCA in a highly transformed Patagonian landscape. ESP European Regional Conference, Ecosystem services Partnership, October 15- 19, San Sebastian, Spain.

Poster Presentations

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References

A, J.G., C, J.G., 2005. Contenido de carbono en la biomasa aérea de bosques nativos en Chile * 26, 33–38.

Agosta, S.J., Hulshof, C.M., Staats, E.G., 2017. Organismal responses to habitat change: herbivore performance, climate and leaf traits in regenerating tropical dry forests. J. Anim. Ecol. 86, 590–604. https://doi.org/10.1111/1365-2656.12647

Albaugh, J.M., Dye, P.J., King, J.S., 2013. Eucalyptus and Water Use in South Africa Eucalyptus and Water Use in South Africa. Int. J. For. Res. 2013. https://doi.org/10.1155/2013/852540

Altamirano, A., Aplin, P., Miranda, A., Cayuela, L., Algar, A.C., Field, R., 2013. High rates of forest loss and turnover obscured by classical landscape measures. Appl. Geogr. 40, 199–211. https://doi.org/10.1016/j.apgeog.2013.03.003

Altamirano, A., Lara, A., 2010. Deforestación en ecosistemas templados de la precordillera andina del centro-sur de Chile Deforestation in temperate ecosystems of pre-Andean range of south-central Chile. Imagine 31, 53–64.

Alvarez-Cobelas, M., Sánchez-Carrillo, S., Angeler, D.G., Sánchez-Andrés, R., 2009. Phosphorus export from catchments: a global view. J. North Am. Benthol. Soc. 28, 805–820. https://doi.org/10.1899/09-073.1

Andersson, K., Lawrence, D., Zavaleta, J., Guariguata, M.R., 2016. More Trees, More Poverty? The Socioeconomic Effects of Tree Plantations in Chile, 2001–2011. Environ. Manage. 57, 123–136. https://doi.org/10.1007/s00267-015-0594-x

Andren, H., 1994. Effects of habitat fragmentation on birds and mammals of suitable habitat : a review. Oikos 71, 355–366. https://doi.org/10.2307/3545823

Andrew, M.E., Wulder, M.A., Nelson, T.A., Coops, N.C., 2015. Spatial data, analysis approaches, and information needs for spatial ecosystem service assessments: a review. GIScience Remote Sens. 52, 344–373. https://doi.org/10.1080/15481603.2015.1033809

Andrieu, E., Vialatte, A., Sirami, C., 2015. Misconceptions of Fragmentation's Effects on Ecosystem Services: A Response to Mitchell et al. Trends Ecol. Evol. 30, 633–634. https://doi.org/10.1016/j.tree.2015.09.003

Armesto, J., Rozzi, R., Smith-Ramirez, C., Arroyo, M.T.., 1998. Conservation Targets in South american Temperate Forests. Science (80-.). 282, 1271–1272. https://doi.org/10.1126/science.282.5392.1271

Armesto, J.J., Manuschevich, D., Mora, A., Smith-Ramirez, C., Rozzi, R., Abarzúa, A.M., Marquet, P. a., Smith-Ramírez, C., Rozzi, R., Abarzúa, A.M., Marquet, P. a., Smith-Ramirez, C., Rozzi, R., Abarzúa, A.M., Marquet, P. a., 2010. From the Holocene to the Anthropocene: A historical framework for land cover change in southwestern South America in the past 15,000 years. Land use policy 27, 148–160. https://doi.org/10.1016/j.landusepol.2009.07.006

Armesto, J.J., Rozzi, R., Smith-Ramírez, C., Arroyo, M.K., 1998. Conservation Targets in South American Temperate Forest. Science (80-.).

Armesto, J.J., Villagran, C., Arroyo, M.K., 1996. Ecologia de los Bosques Nativos de Chile. Ecol. los Bosques Nativ. Chile.

Arroyo, M.T.K., Donoso, C., Murua, R., Pisano, E., Schlatter, R., Serey, I., 1996. Toward an ecologically sustainable forestry project: concepts, analysis and recommendations, Universida. ed.

Bagstad, K.J., Johnson, G.W., Voigt, B., Villa, F., 2013a. Spatial dynamics of ecosystem service flows: A comprehensive approach to quantifying actual services. Ecosyst. Serv. 4, 117–125. https://doi.org/10.1016/j.ecoser.2012.07.012

Bagstad, K.J., Semmens, D.J., Waage, S., Winthrop, R., 2013b. A comparative assessment of decision-support tools for ecosystem services quantification and valuation. Ecosyst. Serv. 5, 27–39. https://doi.org/10.1016/j.ecoser.2013.07.004

Bagstad, K.J., Villa, F., Batker, D., Harrison-Cox, J., Voigt, B., Johnson, G.W., 2014. From theoretical to actual ecosystem services: Mapping beneficiaries and spatial flows in ecosystem service assessments. Ecol. Soc. 19. https://doi.org/10.5751/ES-06523-190264

Balvanera, P., Daw, T.M., Gardner, T., Martin-Lopez, B., Norstrom, A. V, Speranza, C.I., Speirenburg, M., Bennett, E.M., Farfan, M., Hamann, M., Kittinger, J.N., Luthe, T., Maass, M., Peterson, G., Perez-Verdin, G., 2015. Key features for more successful place-based sustainability research on social-ecological systems. Ecol. Soc. 22, 14. https://doi.org/10.5751/ES-08826-220114

Balvanera, P., Uriarte, M., Almeida-Leñero, L., Altesor, A., DeClerck, F., Gardner, T., Hall, J., Lara, A., Laterra, P., Peña-Claros, M., Silva Matos, D.M., Vogl, A.L., Romero-Duque, L.P., Arreola, L.F., Caro-Borrero, Á.P., Gallego, F., Jain, M., Little, C., de Oliveira Xavier, R., Paruelo, J.M., Peinado, J.E., Poorter, L., Ascarrunz, N., Correa, F., Cunha-Santino, M.B., Hernández-Sánchez, A.P., Vallejos, M., 2012. Ecosystem services research in Latin America: The state of the art. Ecosyst. Serv. 2, 56–70. https://doi.org/10.1016/j.ecoser.2012.09.006

Barnett, A., Fargione, J., Smith, M.P., 2016. Mapping Trade-Offs in Ecosystem Services from Reforestation in the Mississippi Alluvial Valley. Bioscience 66, 223–237. https://doi.org/10.1093/biosci/biv181

Bathurst, J.C., Iroumé, A., Cisneros, F., Fallas, J., Iturraspe, R., Novillo, M.G., Urciuolo, A., Bièvre, B. de, Borges, V.G., Coello, C., Cisneros, P., Gayoso, J., Miranda, M., Ramírez, M., 2011. Forest impact on floods due to extreme rainfall and snowmelt in four Latin American environments 1: Field data analysis. J. Hydrol. 400, 281–291. https://doi.org/10.1016/j.jhydrol.2010.11.044

Beier, P., Noss, R.F., 1998. Do Habitat Corridors Provide Connectivity? Conserv. Biol. 12, 1241–1252. https://doi.org/10.1111/j.1523-1739.1998.98036.x

Bennett, E.M., Chaplin-Kramer, R., 2016. Science for the sustainable use of ecosystem services. F1000Research 5, 2622. https://doi.org/10.12688/f1000research.9470.1

Bennett, E.M., Cramer, W., Begossi, A., Cundill, G., Díaz, S., Egoh, B.N., Geijzendorffer, I.R., Krug, C.B., Lavorel, S., Lazos, E., Lebel, L., Martín-López, B., Meyfroidt, P., Mooney, H.A., Nel, J.L., Pascual, U., Payet, K., Harguindeguy, N.P., Peterson, G.D., Prieur-Richard, A.H., Reyers, B., Roebeling, P., Seppelt, R., Solan, M., Tschakert, P., Tscharntke, T., Turner, B.L., Verburg, P.H., Viglizzo, E.F., White, P.C.L., Woodward, G., 2015. Linking biodiversity, ecosystem services, and human well-being: three challenges for designing research for sustainability. Curr. Opin. Environ. Sustain. 14, 76–85. https://doi.org/10.1016/j.cosust.2015.03.007

Bennett, E.M., Peterson, G.D., Gordon, L.J., 2009. Understanding relationships among multiple ecosystem services. Ecol. Lett. 12, 1394–1404. https://doi.org/10.1111/j.1461-0248.2009.01387.x

Birkhofer, K., Diehl, E., Andersson, J., Ekroos, J., FrÃ¼h-MÃ¼ller, A., Machnikowski, F., Mader, V.L., Nilsson, L., Sasaki, K., RundlÃ¶f, M., Wolters, V., Smith, H.G., 2015. Ecosystem servicesâ€"current challenges and opportunities for ecological research. Front. Ecol. Evol. 2, 1–12. https://doi.org/10.3389/fevo.2014.00087

Bonilla, C. a, Reyes, J.L., Magri, A., 2010. Water Erosion Prediction Using the Revised Universal Soil Loss Equation (RUSLE) in a GIS Framework, Central Chile. Chil. J. Agric. Res. 70, 159–169. https://doi.org/10.4067/S0718-58392010000100017

Bonilla, C.A., Johnson, O.I., 2012. Soil erodibility mapping and its correlation with soil properties in Central Chile. Geoderma 189–190, 116–123. https://doi.org/10.1016/j.geoderma.2012.05.005

Brauman, K.A., Daily, G.C., Duarte, T.K., Mooney, H.A., 2007. The Nature and Value of Ecosystem Services: An Overview Highlighting Hydrologic Services. Annu. Rev. Environ. Resour. 32, 67–98. https://doi.org/10.1146/annurev.energy.32.031306.102758

Breiman, L., 2001. Random forests. Mach. Learn. 5–32. https://doi.org/10.1023/A:1010933404324

Brockerhoff, E.G., Jactel, H., Parrotta, J.A., Quine, C.P., Sayer, J., 2008. Plantation forests and biodiversity: Oxymoron or opportunity? Biodivers. Conserv. 17, 925–951. https://doi.org/10.1007/s10531-008-9380-x

Burkhard, B., Kandziora, M., Hou, Y., Müller, F., 2014. Ecosystem service potentials, flows and demands-concepts for spatial localisation, indication and quantification. Landsc. Online 34, 1–32. https://doi.org/10.3097/LO.201434

Bustamante, R.O., Simonetti, J.A., 2005. Is Pinus radiata invading the native vegetation in central Chile? Demographic responses in a fragmented forest. Biol. Invasions 7, 243–249. https://doi.org/10.1007/s10530-004-0740-5

Butchart, S.H.M., Walpole, M., Colle, B., Strien, A. van, Scharlemann, J.P.W., Almond, R.E.A., Baillie, J.E.M., Bomhard, B., Brown, C., Bruno, J., Carpenter, K.E., Carr, G.M., Chanson, J., Anna M., J.-C.V., Watson, R., 2010. Global Biodiversity : Indicators of Recent Declines. Science (80-.). 10, 1164–1169. https://doi.org/10.1126/science.1187512

Canty, M.J., Nielsen, A. a., 2008. Automatic radiometric normalization of multitemporal satellite imagery with the iteratively re-weighted MAD transformation. Remote Sens. Environ. 112, 1025–1036. https://doi.org/10.1016/j.rse.2007.07.013

Carmona, A., Nahuelhual, L., Echeverría, C., Báez, a., 2010. Linking farming systems to landscape change: An empirical and spatially explicit study in southern Chile. Agric. Ecosyst. Environ. 139, 40–50. https://doi.org/10.1016/j.agee.2010.06.015

Central Bank of Chile, 2016. Central Bank of Chile Statistics Database [WWW Document].

Centro de Estudios Agrarios y Ambientales, C., 2010. Identificacion de areas potenciales para establecer reservas destinadas a proteger la fauna nativa de especies hidrobiologicas de agua dulce. Valdivia.

CIREN, 1994. Descripciones de suelos, materiales y simbolos. Estudio agrologico. Santiago.

Clec'h, S. Le, Oszwald, J., Decaens, T., Desjardins, T., Dufour, S., Grimaldi, M., Jegou, N., Lavelle, P., 2016. Mapping multiple ecosystem services indicators: Toward an objective-oriented approach. Ecol. Indic. 69, 508–521. https://doi.org/10.1016/j.ecolind.2016.05.021

Cole, L.J., Brocklehurst, S., Robertson, D., Harrison, W., McCracken, D.I., 2015. Riparian buffer strips: Their role in the conservation of insect pollinators in intensive grassland systems. Agric. Ecosyst. Environ. 211, 207–220. https://doi.org/10.1016/j.agee.2015.06.012

Comte, I., Colin, F., Whalen, J.K., Grünberger, O., Caliman, J.P., 2012. Agricultural Practices in Oil Palm Plantations and Their Impact on Hydrological Changes, Nutrient Fluxes and Water Quality in Indonesia. A Review., Advances in Agronomy. Elsevier Inc. https://doi.org/10.1016/B978-0-12-394277-7.00003-8

CONAF-CONAMA-BIRF, 1999. Catastro y evaluacion de los recursos vegetacionales nativos de Chile. CONAF-CONAMA. Santiago.

Cord, A.F., Bartkowski, B., Beckmann, M., Dittrich, A., Hermans-Neumann, K., Kaim, A., Lienhoop, N., Locher-Krause, K., Priess, J., Schröter-Schlaack, C., Schwarz, N., Seppelt, R., Strauch, M., Václavík, T., Volk, M., 2017a. Towards systematic analyses of ecosystem service trade-offs and synergies: Main concepts, methods and the road ahead. Ecosyst. Serv. 28, 264–272. https://doi.org/10.1016/j.ecoser.2017.07.012

Cord, A.F., Brauman, K.A., Chaplin-Kramer, R., Huth, A., Ziv, G., Seppelt, R., 2017b. Priorities to Advance Monitoring of Ecosystem Services Using Earth Observation. Trends Ecol. Evol. 32, 416–428. https://doi.org/10.1016/j.tree.2017.03.003

Costanza, R., d'Arge, R., de Groot, R.S., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R. V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M., 1997. The value of the world's ecosystem services and natural capital. Nature. https://doi.org/10.1038/387253a0

Costanza, R., de Groot, R., Braat, L., Kubiszewski, I., Fioramonti, L., Sutton, P., Farber, S., Grasso, M., 2017. Twenty years of ecosystem services: How far have we come and how far do we still need to go? Ecosyst. Serv. 28, 1–16. https://doi.org/10.1016/j.ecoser.2017.09.008

Cuevas, J., Huertas, J., Leiva, C., Paulino, L., Dörner, J., Arumí, J., 2014. Nutrient retention in a microcatchment with low levels of anthropogenic pollution. Bosque 35, 75–88. https://doi.org/10.4067/S0717-92002014000100008

Cuevas, J.G., Soto, D., Arismendi, I., Pino, M., Lara, A., Oyarzún, C., 2006. Relating land cover to stream properties in southern Chilean watersheds: Trade-off between geographic scale, sample size, and explicative power. Biogeochemistry 81, 313–329. https://doi.org/10.1007/s10533-006-9043-5

Cunningham, S.C., Mac Nally, R., Baker, P.J., Cavagnaro, T.R., Beringer, J., Thomson, J.R., Thompson, R.M., 2015. Balancing the environmental benefits of reforestation in agricultural regions. Perspect. Plant Ecol. Evol. Syst. 17, 301–317. https://doi.org/10.1016/j.ppees.2015.06.001

Cushman, S.A., 2006. Effects of habitat loss and fragmentation on amphibians: A review and prospectus. Biol. Conserv. 128, 231–240. https://doi.org/10.1016/j.biocon.2005.09.031

Daily, G.C., Polasky, S., Goldstein, J., Kareiva, P.M., Mooney, H.A., Pejchar, L., Ricketts, T.H., Salzman, J., Shallenberger, R., 2009. Ecosystem services in decision making: Time to deliver. Front. Ecol. Environ. 7, 21–28. https://doi.org/10.1890/080025

Dallimer, M., Davies, Z.G., Diaz-Porras, D.F., Irvine, K.N., Maltby, L., Warren, P.H., Armsworth, P.R., Gaston, K.J., 2015. Historical influences on the current provision of multiple ecosystem services. Glob. Environ. Chang. 31, 307–317. https://doi.org/10.1016/j.gloenvcha.2015.01.015

De Groot, R., Brander, L., van der Ploeg, S., Costanza, R., Bernard, F., Braat, L., Christie, M., Crossman, N., Ghermandi, A., Hein, L., Hussain, S., Kumar, P., McVittie, A., Portela, R., Rodriguez, L.C., ten Brink, P., van Beukering, P., 2012. Global estimates of the value of ecosystems and their services in monetary units. Ecosyst. Serv. 1, 50–61. https://doi.org/10.1016/j.ecoser.2012.07.005

De Groot, R.S., Alkemade, R., Braat, L., Hein, L., Willemen, L., 2010. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecol. Complex. 7, 260–272. https://doi.org/10.1016/j.ecocom.2009.10.006

DGA, D.N. de A., 2017. Estadísticas estaciones DGA. Santiago.

Di Castri, F., Hajek, E.R., 1976. Bioclimatologia de Chile. Universidad Catolica de Chile, Santiago.

Díaz, G.I., Nahuelhual, L., Echeverría, C., Marín, S., 2011. Drivers of land abandonment in Southern Chile and implications for landscape planning. Landsc. Urban Plan. 99, 207–217. https://doi.org/10.1016/j.landurbplan.2010.11.005

Díaz, S., Fargione, J., Chapin, F.S., Tilman, D., 2006. Biodiversity loss threatens human well-being. PLoS Biol. 4, 1300–1305. https://doi.org/10.1371/journal.pbio.0040277

Diaz, S., Lavorel, S., Bello, F. De, Quetier, F., Grigulis, K., Robson, T.M., 2007. Incorporating plant functional diversity effects in ecosystem service assessments. Proc. Natl. Acad. Sci. 104, 20684–20689. https://doi.org/10.1073/pnas.0704716104

Doktor, D., Lausch, A., Spengler, D., Thurner, M., 2014. Extraction of Plant Physiological Status from Hyperspectral Signatures Using Machine Learning Methods. Remote Sens. 6, 12247–12274. https://doi.org/10.3390/rs61212247

Donoso, C., 1993. Bosques templados de Chile y Argentina. Variacion, estructura y dinamica, Segunda ed. ed. Editorial Universitaria, Santiago, Chile.

Donoso, C., Lara, A., 1999. Silvicultura de los Bosques nativos de Chile, Ecologia de los Bosques nativos de Chile. Editorial Universitaria, Santiago, Chile.

Durán, A.P., Casalegno, S., Marquet, P.A., Gaston, K.J., 2013. Representation of ecosystem services by terrestrial protected areas: Chile as a case study. PLoS One 8, 1–8. https://doi.org/10.1371/journal.pone.0082643

Eastman, J.., 2012. IDRISI Selva (Worcester, MA: Clark University).

Echeverria, C., Coomes, D. a., Hall, M., Newton, A.C., 2008. Spatially explicit models to analyze forest loss and fragmentation between 1976 and 2020 in southern Chile. Ecol. Modell. 212, 439–449. https://doi.org/10.1016/j.ecolmodel.2007.10.045

Echeverria, C., Coomes, D., Salas, J., Rey-Benayas, J.M., Lara, A., Newton, A., 2006a. Rapid deforestation and fragmentation of Chilean Temperate Forests. Biol. Conserv. 130, 481–494. https://doi.org/10.1016/j.biocon.2006.01.017

Echeverría, C., Newton, A., Nahuelhual, L., Coomes, D., Rey-Benayas, J.M., 2012. How landscapes change: Integration of spatial patterns and human processes in temperate landscapes of southern Chile. Appl. Geogr. 32, 822–831. https://doi.org/10.1016/j.apgeog.2011.08.014

Echeverría, C., Newton, A.C., Lara, A., Benayas, J.M.R., Coomes, D. a., 2007. Impacts of forest fragmentation on species composition and forest structure in the temperate landscape of southern Chile. Glob. Ecol. Biogeogr. 16, 426–439. https://doi.org/10.1111/j.1466-8238.2007.00311.x

Eigenbrod, F., Armsworth, P.R., Anderson, B.J., Heinemeyer, A., Gillings, S., Roy, D.B., Thomas, C.D., Gaston, K.J., 2010. The impact of proxy-based methods on mapping the distribution of ecosystem services. J. Appl. Ecol. 47, 377–385. https://doi.org/10.1111/j.1365-2664.2010.01777.x

Ellis, E.C., Goldewijk, K.K., Siebert, S., Lightman, D., Ramankutty, N., 2010. Anthropogenic transformation of the biomes, 1700 to 2000. Glob. Ecol. Biogeogr. 19, 589–606. https://doi.org/10.1111/j.1466-8238.2010.00540.x

Ellis, E.C., Kaplan, J.O., Fuller, D.Q., Vavrus, S., Klein Goldewijk, K., Verburg, P.H., 2013. Used planet: a global history. Proc. Natl. Acad. Sci. U. S. A. 110, 7978–85. https://doi.org/10.1073/pnas.1217241110

Ellis, E.C., Ramankutty, N., 2008. Putting people in the map: Anthropogenic biomes of the world. Front. Ecol. Environ. 6, 439–447. https://doi.org/10.1890/070062

ESRI, 2011. ArcGIS Desktop: Release 10. Redlands, CA: Environmental Systems Research Institute.

FAO/IIASA/ISRIC/ISSCAS/JRC, 2012. Harmonized World Soil Database (version 1.2).

Farley, K.A., Jobbágy, E.G., Jackson, R.B., 2005. Effects of afforestation on water yield: A global synthesis with implications for policy. Glob. Chang. Biol. 11, 1565–1576. https://doi.org/10.1111/j.1365-2486.2005.01011.x

Fischer, J., Lindenmayer, D.B., 2007. Landscape modification and habitat fragmentation: A synthesis. Glob. Ecol. Biogeogr. 16, 265–280. https://doi.org/10.1111/j.1466-8238.2007.00287.x

Fisher, B., Turner, R.K., Morling, P., 2009. Defining and classifying ecosystem services for decision making. Ecol. Econ. 68, 643–653. https://doi.org/10.1016/j.ecolecon.2008.09.014

Foley, J. a, Ramankutty, N., Brauman, K. a, Cassidy, E.S., Gerber, J.S., Johnston, M., Mueller, N.D., O'Connell, C., Ray, D.K., West, P.C., Balzer, C., Bennett, E.M., Carpenter, S.R., Hill, J., Monfreda, C., Polasky, S., Rockström, J., Sheehan, J., Siebert, S., Tilman, D., Zaks, D.P.M., 2011. Solutions for a cultivated planet. Nature 478, 337–42. https://doi.org/10.1038/nature10452

Foley, J.A., Defries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K., Helkowski, J.H., Holloway, T., Howard, E.A., Kucharik, C.J., Monfreda, C., Patz, J. a, Prentice, I.C., Ramankutty, N., Snyder, P.K., 2005. Global consequences of land use. Science 309, 570–4. https://doi.org/10.1126/science.1111772

Folke, C., 2006. Resilience: The emergence of a perspective for social–ecological systems analyses. Glob. Environ. Chang. 16, 253–267. https://doi.org/10.1016/j.gloenvcha.2006.04.002

Foody, G.M., 2002. Status of land cover classification accuracy assessment. Remote Sens. Environ. 80, 185–201. https://doi.org/10.1016/S0034-4257(01)00295-4

Friis, C., Nielsen, J.Ø., Otero, I., Haberl, H., Niewöhner, J., Hostert, P., 2015. From teleconnection to telecoupling: taking stock of an emerging framework in land system science. J. Land Use Sci. 4248, 1–23. https://doi.org/10.1080/1747423X.2015.1096423

Gajardo, R., 1994. La vegetacion natural de Chile: clasificacion y distribucion geografica. Editorial Universitaria, Santiago.

Gao, F., Masek, J., 2008. Developing Consistent and Continuous Moderate Resolution Data Products from Landsat and Landsat - like Data „ Location consistent.

García-Nieto, A.P., García-Llorente, M., Iniesta-Arandia, I., Martín-López, B., 2013. Mapping forest ecosystem services: From providing units to beneficiaries. Ecosyst. Serv. 4, 126–138. https://doi.org/10.1016/j.ecoser.2013.03.003

Garcia, O., 1995. Índices De Sitio Preliminares Para Eucalipto.

García, O., 1970. Indices de sitio para pino insigne en Chile. Instituto Forestal. Serie de Investigacion. Santiado, Chile.

Garcia, X., Benages-Albert, M., Pavón, D., Ribas, A., Garcia-Aymerich, J., Vall-Casas, P., 2017. Public participation GIS for assessing landscape values and improvement preferences in urban stream corridors. Appl. Geogr. 87, 184–196. https://doi.org/10.1016/j.apgeog.2017.08.009

Gayoso, J., 2001. Medicion de la capacidad de captura de carbono en bosques nativos y plantaciones de Chile. Trab. Present. en Taller Secuestro Carbono. Mérida, Venez. 2001 1–22.

Gayoso, J., Schlegel, B., 2003. Estudio Linea Base Carbono: Carbono en Bosques Nativos, Matorrales, y Praderas de la Decima Region de Chile.

Gergel, S.E., Turner, M.G., 2002. Learning Lanscap Ecology, a practical guide to concepts and techniques.

Gillies, C.S., St. Clair, C.C., 2008. Riparian corridors enhance movement of a forest specialist bird in fragmented tropical forest. Proc. Natl. Acad. Sci. 105, 19774–19779. https://doi.org/10.1073/pnas.0803530105

Gislason, P.O., Benediktsson, J.A., Sveinsson, J.R., 2006. Random Forests for land cover classification. Pattern Recognit. Lett. 27, 294–300. https://doi.org/10.1016/j.patrec.2005.08.011

Goldstein, J.H., Caldarone, G., Colvin, C., Duarte, T.K., Ennaanay, D., Fronda, K., Hannahs, N., Mckenzie, E., Mendoza, G., Smith, K., Woodside, U., Daily, G.C., 2008. The Natural Capital Project , Kamehameha Schools , and InVEST : Integrating Ecosystem Services into Land-Use Planning in Hawai ` i.

Goldstein, J.H., Caldarone, G., Duarte, T.K., Ennaanay, D., Hannahs, N., Mendoza, G., Polasky, S., Wolny, S., Daily, G.C., 2012. Integrating ecosystem-service tradeoffs into land-use decisions. Proc. Natl. Acad. Sci. 109, 7565–7570. https://doi.org/10.1073/pnas.1201040109

Gómez-Baggethun, E., Naredo, J.M., 2015. In search of lost time: the rise and fall of limits to growth in international sustainability policy. Sustain. Sci. 10, 385–395. https://doi.org/10.1007/s11625-015-0308-6

Gonzalez, P., Gayoso, J., 2005. Comparison of Three Methods to Project Future Baseline Carbon Emissions in Temperate Rainforest , Curiñanco , Chile 1–3.

Goodwin, N.R., Collett, L.J., Denham, R.J., Flood, N., Tindall, D., 2013. Cloud and cloud shadow screening across Queensland, Australia: An automated method for Landsat TM/ETM+ time series. Remote Sens. Environ. 134, 50–65. https://doi.org/10.1016/j.rse.2013.02.019

Grau, H.R., Kuemmerle, T., Macchi, L., 2013. Beyond ‘land sparing versus land sharing’: environmental heterogeneity, globalization and the balance between agricultural production and nature conservation. Curr. Opin. Environ. Sustain. 5, 477–483. https://doi.org/10.1016/j.cosust.2013.06.001

Grêt-Regamey, A., Weibel, B., Bagstad, K.J., Ferrari, M., Geneletti, D., Klug, H., Schirpke, U., Tappeiner, U., 2014. On the effects of scale for ecosystem services mapping. PLoS One 9, 1–26. https://doi.org/10.1371/journal.pone.0112601

Griffiths, P., Kuemmerle, T., Baumann, M., Radeloff, V.C., Abrudan, I. V., Lieskovsky, J., Munteanu, C., Ostapowicz, K., Hostert, P., 2013. Forest disturbances, forest recovery, and changes in forest types across the carpathian ecoregion from 1985 to 2010 based on landsat image composites. Remote Sens. Environ. 151, 72–88. https://doi.org/10.1016/j.rse.2013.04.022

Guerry, A.D., Polasky, S., Lubchenco, J., Chaplin-Kramer, R., Daily, G.C., Griffin, R., Ruckelshaus, M., Bateman, I.J., Duraiappah, A., Elmqvist, T., Feldman, M.W., Folke, C., Hoekstra, J., Kareiva, P.M., Keeler, B.L., Li, S., McKenzie, E., Ouyang, Z., Reyers, B., Ricketts, T.H., Rockström, J., Tallis, H., Vira, B., 2015. Natural capital and ecosystem services informing decisions: From promise to practice. Proc. Natl. Acad. Sci. 112, 7348–7355. https://doi.org/10.1073/pnas.1503751112

Gustafson, E.J., Parker, G.R., 1992. Relationship between land cover proportion and indices of spatial pattern. Landsc. Ecol. 7, 101–110.

Gutiérrez, A.G., Armesto, J.J., Aravena, J.-C., Carmona, M., Carrasco, N. V., Christie, D. a., Peña, M.-P., Pérez, C., Huth, A., 2009. Structural and environmental characterization of old-growth temperate rainforests of northern Chiloé Island, Chile: Regional and global relevance. For. Ecol. Manage. 258, 376–388. https://doi.org/10.1016/j.foreco.2009.03.011

Haddad, N.M., Brudvig, L.A., Clobert, J., Davies, K.F., Gonzalez, A., Holt, R.D., Lovejoy, T.E., Sexton, J.O., Austin, M.P., Collins, C.D., Cook, W.M., Damschen, E.I., Ewers, R.M., Foster, B.L., Jenkins, C.N., King, A.J., Laurance, W.F., Levey, D.J., Margules, C.R., Melbourne, B.A., Nicholls, A.O., Orrock, J.L., Song, D., Townshend, J.R., 2015. Habitat fragmentation and its lasting impact on Earth ’ s ecosystems. Sci. Adv. 1–9.

Haines-Young, R., 2009. Land use and biodiversity relationships. Land use policy 26, 178–186. https://doi.org/10.1016/j.landusepol.2009.08.009

Haines-Young, R., Potschin, M., 2018. CICES (Common International Classification of Ecosystem Services) Guidance on the Application of the Revised Structure.

Hamel, P., Chaplin-Kramer, R., Sim, S., Mueller, C., 2015. A new approach to modeling the sediment retention service (InVEST 3.0): Case study of the Cape Fear catchment, North

Carolina, USA. Sci. Total Environ. 524–525, 166–177. https://doi.org/10.1016/j.scitotenv.2015.04.027

Hansen, J., Sato, M., Ruedy, R., Lacis, A., Oinas, V., 2000. Global warming in the twenty-first century : An alternative scenario. Proc. Natl. Acad. Sci. 97, 9875–9880.

Hansen, M.C., Loveland, T.R., 2012. A review of large area monitoring of land cover change using Landsat data. Remote Sens. Environ. 122, 66–74. https://doi.org/10.1016/j.rse.2011.08.024

Harrison, P.A., Dunford, R., Barton, D.N., Kelemen, E., Martín-López, B., Norton, L., Termansen, M., Saarikoski, H., Hendriks, K., Gómez-Baggethun, E., Czúcz, B., García-Llorente, M., Howard, D., Jacobs, S., Karlsen, M., Kopperoinen, L., Madsen, A., Rusch, G., van Eupen, M., Verweij, P., Smith, R., Tuomasjukka, D., Zulian, G., 2017. Selecting methods for ecosystem service assessment: A decision tree approach. Ecosyst. Serv. https://doi.org/10.1016/j.ecoser.2017.09.016

Hastie, T., Tibshirani, R., Friedman, J., 2009. The Elements of Statistical Learning - Data Mining, Inference, and Prediction. Springer.

Hein, L., van Koppen, K., de Groot, R.S., van Ierland, E.C., 2006. Spatial scales, stakeholders and the valuation of ecosystem services. Ecol. Econ. 57, 209–228. https://doi.org/10.1016/j.ecolecon.2005.04.005

Hersperger, A.M., Gennaio, M., Verburg, P.H., Bürgi, M., 2012. Linking Land Change with Driving Forces and Actors : Four Conceptual Models 15, 1–14.

Hervé-Fernández, P., Oyarzún, C., Brumbt, C., Huygens, D., Bodé, S., Verhoest, N.E.C., Boeckx, P., 2016. Assessing the ‘two water worlds’ hypothesis and water sources for native and exotic evergreen species in south-central Chile. Hydrol. Process. 30, 4227–4241. https://doi.org/10.1002/hyp.10984

Hess, R.H., Fisher, R. a, 2001. Communicating clearly about conservation corridors. Landsc. Urban Plan. 55, 195–208.

Hicks, C.C., Levine, A., Agrawal, A., Basurto, X., Breslow, S.J., Carothers, C., Charnley, S., Coulthard, S., Dolsak, N., Donatuto, J., Garcia-Quijano, C., Mascia, M.B., Norman, K., Poe, M.R., Satterfield, T., Martin, K. St., Levin, P.S., 2016. Engage key social concepts for sustainability. Science (80-.). 352, 38–40. https://doi.org/10.1126/science.aad4977

Hietel, E., Waldhardt, R., Otte, A., 2004. Analysing land-cover changes in relation to environmental variables in Hesse, Germany. Landsc. Ecol. 19, 473–489. https://doi.org/10.1023/B:LAND.0000036138.82213.80

Hijmans, R.J., Cameron, S.E., Parra, J.L., Jones, P.G., Jarvis, A., 2005. Very high resolution interpolated climate surfaces for global land areas. Int. J. Climatol. 25, 1965–1978.

Hilker, T., Wulder, M.A., Coops, N.C., Seitz, N., White, J.C., Gao, F., Masek, J.G., Stenhouse, G., 2009. Generation of dense time series synthetic Landsat data through data blending with MODIS using a spatial and temporal adaptive reflectance fusion model. Remote Sens. Environ. 113, 1988–1999. https://doi.org/10.1016/j.rse.2009.05.011

Hobbs, R.J., 1992. The role of corridors in conservation: Solution or bandwagons. Trends Ecol. Evol. 7, 389–392.

Hobbs, R.J., Arico, S., Aronson, J., Baron, J.S., Bridgewater, P., Cramer, V.A., Epstein, P.R., Ewel, J.J., Klink, C.A., Lugo, A.E., Norton, D., Ojima, D., Richardson, D.M., Sanderson, E.W., Valladares, F., Vilà, M., Zamora, R., Zobel, M., 2006. Novel ecosystems: Theoretical and management aspects of the new ecological world order. Glob. Ecol. Biogeogr. 15, 1–7. https://doi.org/10.1111/j.1466-822X.2006.00212.x

Hofman, M.P.G., Hayward, M.W., Kelly, M.J., Balkenhol, N., 2018. Enhancing conservation network design with graph-theory and a measure of protected area effectiveness: Refining wildlife corridors in Belize, Central America. Landsc. Urban Plan. 178, 51–59. https://doi.org/10.1016/j.landurbplan.2018.05.013

Holland, R.A., Eigenbrod, F., Armsworth, P.R., Anderson, B.J., Thomas, C.D., Gaston, K.J., 2011. The influence of temporal variation on relationships between ecosystem services. Biodivers. Conserv. 20, 3285–3294. https://doi.org/10.1007/s10531-011-0113-1

Huang, C., Goward, S.N., Masek, J.G., Thomas, N., Zhu, Z., Vogelmann, J.E., 2010. An automated approach for reconstructing recent forest disturbance history using dense Landsat time series stacks. Remote Sens. Environ. 114, 183–198. https://doi.org/10.1016/j.rse.2009.08.017

Huber, A., Iroumé, A., Bathurst, J.C., 2008. Effect of Pinus radiata plantations on water balance in Chile. Hydrol. Process. 22, 142–148. https://doi.org/10.1002/hyp.6582

Huygens, D., Roobroeck, D., Cosyn, L., Salazar, F., Godoy, R., Boeckx, P., 2011. Microbial nitrogen dynamics in south central Chilean agricultural and forest ecosystems located on an Andisol. Nutr. Cycle. Agroecosystems 89, 175–187. https://doi.org/10.1007/s10705-010-9386-0

INE Instituto Nacional de Estadística, 2012. Compendio estadistico. Santiago.

INE Instituto Nacional de Estadística, 2007. Censo Agropecuario y Forestal 2007. Santiago.

INFOR, 1999. Disponibilidad de madera de plantaciones de pino radiata en Chile 1998-2027. Santiado, Chile.

IPCC, Barker, T., 2007. Climate Change 2007 : An Assessment of the Intergovernmental Panel on Climate Change, Change. Cambridge University Press. https://doi.org/10.1256/004316502320517344

Iroume, A., 2003. Transporte de sedimentos en una cuenca de montaña en la Cordillera de los Andes de la Novena Región de Chile. Bosque (Valdivia) 24, 125–135. https://doi.org/10.4067/S0717-92002003000100010

Izquierdo, A.E., Grau, H.R., 2009. Agriculture adjustment, land-use transition and protected areas in Northwestern Argentina. J. Environ. Manage. 90, 858–865. https://doi.org/10.1016/j.jenvman.2008.02.013

Jackson, R.B., Canadell, J., Ehleringer, J.R., Mooney, H.A., Sala, O.E., Schulze, E.D., 1996. A global analysis of root distributions for terrestrial biomes. Oecologia 108, 389–411. https://doi.org/10.1007/BF00333714

Jackson, R.B., Jobbagy, E.G., Avissar, R., Roy, S.B., Barrett, D.J., Cook, C.W., Farley, K.A., le Maitre, D.C., McCarl, B.A., Murray, B.C., Jackson, R.B., Jobba, E.G., Roy, S.B., Barrett, D.J., Cook, C.W., Farley, K.A., Maitre, D.C., McCarl, B.A., Murray, B.C., 2005. Trading water for Carbon with biological Carbon sequestration. Science (80-.). 310, 1944–1947. https://doi.org/10.1126/science.1119282

James, G.M., Witten, D., Hastie, T., Tibshirani, R., 2013. An Introduction to Statistical Learning: with Applications in R. Springer Berlin Heidelberg. https://doi.org/10.1007/978-1-4614-7138-7 1

Jax, K., Barton, D.N., Chan, K.M.A., de Groot, R., Doyle, U., Eser, U., Görg, C., Gómez-Baggethun, E., Griewald, Y., Haber, W., Haines-Young, R., Heink, U., Jahn, T., Joosten, H., Kerschbaumer, L., Korn, H., Luck, G.W., Matzdorf, B., Muraca, B., Neßhöver, C., Norton, B., Ott, K., Potschin, M., Rauschmayer, F., von Haaren, C., Wichmann, S., 2013. Ecosystem services and ethics. Ecol. Econ. 93, 260–268. https://doi.org/10.1016/j.ecolecon.2013.06.008

Jiang, P., Cheng, L., Li, M., Zhao, R., Huang, Q., 2014. Analysis of landscape fragmentation processes and driving forces in wetlands in arid areas: A case study of the middle reaches of the Heihe River, China. Ecol. Indic. 46, 240–252. https://doi.org/10.1016/j.ecolind.2014.06.026

Joppa, L.N., Boyd, J.W., Duke, Clifford S. Hampton, S., Jackson, Stephen T. Jacobs, K.L., Kassam, K.-A.S., Mooney, H.A., Ogden, L.A., Ruckelshaus, Mary Shogren, J.F., 2016. Government : Plan for ecosystem services 351, 1036–1037. https://doi.org/10.1126/science.351.6277.1037-a

Ju, J., Roy, D.P., 2008. The availability of cloud-free Landsat ETM+ data over the conterminous United States and globally. Remote Sens. Environ. 112, 1196–1211. https://doi.org/10.1016/j.rse.2007.08.011

Ju, J., Roy, D.P., Vermote, E., Masek, J., Kovalskyy, V., 2012. Continental-scale validation of MODIS-based and LEDAPS Landsat ETM+ atmospheric correction methods. Remote Sens. Environ. 122, 175–184. https://doi.org/10.1016/j.rse.2011.12.025

Kaufman, Y.J., Tanré, D., Remer, L.A., Vermote, E.F., Chu, A., Holben, B.N., 1997. Operational remote sensing of tropospheric aerosol over land from EOS moderate resolution imaging spectroradiometer. J. Geophys. Res. Atmos. 102, 17051–17067.

Kawasaki, K., Shigesada, N., Iinuma, M., 2017. Effects of long-range taxis and population pressure on the range expansion of invasive species in heterogeneous environments. Theor. Ecol. 10, 269–286. https://doi.org/10.1007/s12080-017-0328-1

Keith, H., Mackey, B.G., Lindenmayer, D.B., 2009. Re-evaluation of forest biomass carbon stocks and lessons from the world ' s most carbon-dense forests. Proc. Natl. Acad. Sci. U. S. A. 106, 11635–11640. https://doi.org/10.1073/pnas.0901970106

Kennedy, R.E., Townsend, P. a., Gross, J.E., Cohen, W.B., Bolstad, P., Wang, Y.Q., Adams, P., 2009. Remote sensing change detection tools for natural resource managers: Understanding concepts and tradeoffs in the design of landscape monitoring projects. Remote Sens. Environ. 113, 1382–1396. https://doi.org/10.1016/j.rse.2008.07.018

Kennedy, R.E., Yang, Z., Cohen, W.B., 2010. Detecting trends in forest disturbance and recovery using yearly Landsat time series: 1. LandTrendr - Temporal segmentation algorithms. Remote Sens. Environ. 114, 2897–2910. https://doi.org/10.1016/j.rse.2010.07.008

Klein Goldewijk, K., Ramankutty, N., 2004. Land cover change over the last three centuries due to human activities: The availability of new global data sets. GeoJournal 61, 335–344. https://doi.org/10.1007/s10708-004-5050-z

Lambin, E.F., Geist, H.J., 2006. Land-Use and Land-Cover Change, Local Processes and Global Impacts. Springer Berlin Heidelberg New York.

Lambin, E.F., Geist, H.J., Lepers, E., 2003. Dynamics of Land-use and Land-cover change in Tropical Regions. Annu. Rev. Environ. Resour. 28, 205–241. https://doi.org/10.1146/annurev.energy.28.050302.105459

Lambin, E.F., Meyfroidt, P., 2011. Global land use change, economic globalization, and the looming land scarcity. Proc. Natl. Acad. Sci. U. S. A. 108, 3465–72. https://doi.org/10.1073/pnas.1100480108

Lambin, E.F., Meyfroidt, P., 2010. Land use transitions: Socio-ecological feedback versus socio-economic change. Land use policy 27, 108–118. https://doi.org/10.1016/j.landusepol.2009.09.003

Lambin, E.F., Turner, B.L., Geist, H.J., Agbola, S.B., Angelsen, A., Bruce, J.W., Coomes, O.T., Dirzo, R., Fischer, G., Folke, C., George, P.S., Homewood, K., Imbernon, J., Leemans, R., Li, X., Moran, E.F., Mortimore, M., Ramakrishnan, P.S., Richards, J.F., Skånes, H., Steffen, W., Stone, G.D., Svedin, U., Veldkamp, T. a., Vogel, C., Xu, J., 2001. The causes of land-use and land-cover change: moving beyond the myths. Glob. Environ. Chang. 11, 261–269. https://doi.org/10.1016/S0959-3780(01)00007-3

Lamy, T., Liss, K.N., Gonzalez, A., Bennett, E.M., 2016. Landscape structure affects the provision of multiple ecosystem services. Environ. Res. Lett. 11, 1–9. https://doi.org/10.1088/1748-9326/11/12/124017

Lara, A., Little, C., Nahuelhual, L., Urrutia, R., Díaz, I.A., 2011. Lessons, challenges and policy recommendations for the management, conservation and restoration of native forests in Chile, in: Biodiversity Conservation in the Americas: Lessons and Policy Recommendations. Santiago, Chile, pp. 259–299.

Lara, A., Little, C., Urrutia, R., McPhee, J., Álvarez-Garretón, C., Oyarzún, C., Soto, D., Donoso, P., Nahuelhual, L., Pino, M., Arismendi, I., 2009. Assessment of ecosystem services as an opportunity for the conservation and management of native forests in Chile. For. Ecol. Manage. 258, 415–424. https://doi.org/10.1016/j.foreco.2009.01.004

Lara, A., Veblen, T.T., 1993. Forest plantations in Chile: a successful model?, in: Afforestation Policies, Planning and Progress. London, pp. 118–139.

Laurance, W.F., Camargo, J.L.C., Fearnside, P.M., Lovejoy, T.E., Williamson, G.B., Mesquita, R.C.G., Meyer, C.F.J., Bobrowiec, P.E.D., Laurance, S.G.W., 2018. An Amazonian rainforest and its fragments as a laboratory of global change. Biol. Rev. 93, 223–247. https://doi.org/10.1111/brv.12343

Lautenbach, S., Kugel, C., Lausch, A., Seppelt, R., 2011. Analysis of historic changes in regional ecosystem service provisioning using land use data. Ecol. Indic. 11, 676–687. https://doi.org/10.1016/j.ecolind.2010.09.007

Lehner, B., Verdin, K., Jarvis, A., 2008. New global hydrography derived from spaceborne elevation data. Eos, Trans. AGU 10.

Lele, S., Springate-Bagnski, O., Lakerveld, R., Deb, D., Dash, P., 2013. Ecosystem Services: Origins, Contributions, Pitfalls, and Alternatives. Conserv. Soc. 11, 343–358.

León-Muñoz, J., Echeverría, C., Marcé, R., Riss, W., Sherman, B., Iriarte, J.L., 2013. The combined impact of land use change and aquaculture on sediment and water quality in oligotrophic Lake Rupanco (North Patagonia, Chile, 40.8°S). J. Environ. Manage. 128, 283–291. https://doi.org/10.1016/j.jenvman.2013.05.008

Liaw, A., Wiener, M., 2002. Classification and Regression by randomForest. R news 2, 18–22. https://doi.org/10.1177/154405910408300516

Lindenmayer, D.B., Franklin, J.F., Fischer, J., 2006. General management principles and a checklist of strategies to guide forest biodiversity conservation. Biol. Conserv. 131, 433–445. https://doi.org/10.1016/j.biocon.2006.02.019

Little, C., Cuevas, J.G., Lara, A., Pino, M., Schoenholtz, S., 2014. Buffer effects of streamside native forests on water provision in watersheds dominated by exotic forest plantations. Ecohydrology n/a-n/a. https://doi.org/10.1002/eco.1575

Little, C., Lara, A., McPhee, J., Urrutia, R., 2009. Revealing the impact of forest exotic plantations on water yield in large scale watersheds in South-Central Chile. J. Hydrol. 374, 162–170. https://doi.org/10.1016/j.jhydrol.2009.06.011

Liu, S., Costanza, R., Farber, S., Troy, A., 2010. Valuing ecosystem services, Valuing ecosystem services-Theory practice and the need for a transdisciplinary synthesis. Ann. N. Y. Acad. Sci. 1185, 54–78. https://doi.org/10.1111/j.1749-6632.2009.05167.x

Locher-Krause, K.E., Lautenbach, S., Volk, M., 2017a. Spatio-temporal change of ecosystem services as a key to understand natural resource utilization in Southern Chile. Reg. Environ. Chang. 17, 2477–2493. https://doi.org/10.1007/s10113-017-1180-y

Locher-Krause, K.E., Volk, M., Waske, B., Thonfeld, F., Lautenbach, S., 2017b. Expanding temporal resolution in landscape transformations: Insights from a landsat-based case study in Southern Chile. Ecol. Indic. 75, 132–144. https://doi.org/10.1016/j.ecolind.2016.12.036

López-Vicente, M., Poesen, J., Navas, A., Gaspar, L., 2013. Predicting runoff and sediment connectivity and soil erosion by water for different land use scenarios in the Spanish Pre-Pyrenees. Catena 102, 62–73. https://doi.org/10.1016/j.catena.2011.01.001

Loveland, T.E., Reed, B.C., Brown, J.F., Ohlen, D.O., Zhu, Z., Yang, L., Merchant, J.W., 2000. Development of a global land cover characteristics database and IGBP DISCover from 1 km AVHRR data. Int. J. Remote Sens. 21, 1303–1330. https://doi.org/10.1080/014311600210191

Lunetta, R.S., Knight, J.F., Ediriwickrema, J., Lyon, J.G., Worthy, L.D., 2006. Land-cover change detection using multi-temporal MODIS NDVI data. Remote Sens. Environ. 105, 142–154. https://doi.org/10.1016/j.rse.2006.06.018

Lyon, J., Gross, N.M., 2005. Patterns of plant diversity and plant-environmental relationships across three riparian corridors. For. Ecol. Manage. 204, 267–278. https://doi.org/10.1016/j.foreco.2004.09.019

Mach, M.E., Martone, R.G., Chan, K.M.A., 2015. Human impacts and ecosystem services: Insufficient research for trade-off evaluation. Ecosyst. Serv. 16, 112–120. https://doi.org/10.1016/j.ecoser.2015.10.018

Machado, A., 2004. An index of naturalness. J. Nat. Conserv. 12, 95–110. https://doi.org/10.1016/j.jnc.2003.12.002

Malinga, R., Gordon, L.J., Jewitt, G., Lindborg, R., 2015. Mapping ecosystem services across scales and continents - A review. Ecosyst. Serv. 13, 57–63. https://doi.org/10.1016/j.ecoser.2015.01.006

Manuschevich, D., 2016. Neoliberalization of forestry discourses in Chile. For. Policy Econ. 69, 21–30. https://doi.org/10.1016/j.forpol.2016.03.006

Martinez-Harms, M.J., Bryan, B.A., Wood, S.A., Fisher, D.M., Law, E., Rhodes, J.R., Dobbs, C., Biggs, D., Wilson, K.A., 2018. Science of the Total Environment Inequality in access to cultural ecosystem services from protected areas in the Chilean biodiversity hotspot. Sci. Total Environ. 636, 1128–1138. https://doi.org/10.1016/j.scitotenv.2018.04.353

Martínez Pastur, G., Peri, P.L., Lencinas, M. V., García-Llorente, M., Martín-López, B., 2016. Spatial patterns of cultural ecosystem services provision in Southern Patagonia. Landsc. Ecol. 31, 383–399. https://doi.org/10.1007/s10980-015-0254-9

Masek, J., Wolfe, R., Hall, F., Gsfc, N., Huang, C., Goward, S., Healey, S., Powell, S., 2006. LEDAPS Overview and Status.

Masek, J.G., Vermote, E.F., Saleous, N.E., Wolfe, R., Hall, F.G., Huemmrich, K.F., Gao, F., Kutler,

J., Lim, T., 2006. A Landsat Surface Reflectance Dataset 3, 68–72.
Mcgarigal, K., 2014. Fragstats help. Dep. Environ. Conserv. Univ. Massachusetts, Amherst 1–182. https://doi.org/10.1093/ntr/nts298
Mcgarigal, K., 2001. Introduction to Landscape Ecology. Landsc. Ecol. 2001, 1–52. https://doi.org/10.1007/BF02275260
Meyfroidt, P., Lambin, E.F., 2011. Global Forest Transition: Prospects for an End to Deforestation, Annual Review of Environment and Resources. https://doi.org/10.1146/annurev-environ-090710-143732
Meyfroidt, P., Rudel, T.K., Lambin, E.F., 2010. Forest transitions, trade, and the global displacement of land use. Proc. Natl. Acad. Sci. U. S. A. 107, 20917–20922. https://doi.org/10.1073/pnas.1014773107
Millennium Ecosystem Assessment, 2005. Ecosystems and Human Well-being: Synthesis. WashingtonDC Isl. Press, Millenium Assessments 5, 1–100. https://doi.org/10.1196/annals.1439.003
Miranda, A., Altamirano, A., Cayuela, L., Lara, A., González, M., 2017. Native forest loss in the Chilean biodiversity hotspot: revealing the evidence. Reg. Environ. Chang. 17, 285–297. https://doi.org/10.1007/s10113-016-1010-7
Miranda, A., Altamirano, A., Cayuela, L., Pincheira, F., Lara, A., 2015. Different times, same story: Native forest loss and landscape homogenization in three physiographical areas of south-central of Chile. Appl. Geogr. 60, 20–28. https://doi.org/10.1016/j.apgeog.2015.02.016
Mitchell, M.G.E., Bennett, E.M., Gonzalez, A., 2014. Forest fragments modulate the provision of multiple ecosystem services. J. Appl. Ecol. 51, 909–918. https://doi.org/10.1111/1365-2664.12241
Mitchell, M.G.E., Bennett, E.M., Gonzalez, A., 2013. Linking Landscape Connectivity and Ecosystem Service Provision: Current Knowledge and Research Gaps. Ecosystems 16, 894–908. https://doi.org/10.1007/s10021-013-9647-2
Mitchell, M.G.E., Suarez-Castro, A.F., Martinez-Harms, M.J., Maron, M., McAlpine, C., Gaston, K.J., Johansen, K., Rhodes, J.R., 2015. Reframing landscape fragmentation’s effects on ecosystem services. Trends Ecol. Evol. 30, 190–198. https://doi.org/10.1016/j.tree.2015.01.011
Mouchet, M.A., Lamarque, P., Martín-López, B., Crouzat, E., Gos, P., Byczek, C., Lavorel, S., 2015. An interdisciplinary methodological guide for quantifying associations between ecosystem services. Glob. Environ. Chang. https://doi.org/10.1016/j.gloenvcha.2014.07.012
Mu, Q., Ann, H.F., Maosheng, Z., Running, S.W., 2007. Regional evaporation estimates from flux tower and MODIS satellite data. Remote Sens. Environ. 106, 285–304. https://doi.org/10.1016/j.rse.2006.07.007
Mu, Q., Zhao, M., Running, S.W., 2011. Improvements to a MODIS global terrestrial evapotranspiration algorithm. Remote Sens. Environ. 115, 1781–1800. https://doi.org/10.1016/j.rse.2011.02.019
Müller, F., de Groot, R., Willemen, L., 2010. Ecosystem services at the landscape scale: The need for integrative approaches. Landsc. Online 23, 1–11. https://doi.org/10.3097/LO.201023
Myers, N., Mittermeier, R.A., Mittermeier, C.G., da Fonseca, G.A.B., Kent, J., 2000. Biodiversity hotspots for conservation priorities. Nature 403, 853–858. https://doi.org/10.1038/35002501
Nahuelhual, L., Carmona, a., Aguayo, M., Echeverria, C., 2013. Land use change and ecosystem services provision: a case study of recreation and ecotourism opportunities in southern Chile. Landsc. Ecol. 329–344. https://doi.org/10.1007/s10980-013-9958-x
Nahuelhual, L., Carmona, A., Lara, A., Echeverría, C., González, M.E., 2012. Land-cover change to forest plantations: Proximate causes and implications for the landscape in south-central Chile. Landsc. Urban Plan. 107, 12–20. https://doi.org/10.1016/j.landurbplan.2012.04.006
Nahuelhual, L., Carmona, A., Lozada, P., Jaramillo, A., Aguayo, M., 2013b. Mapping recreation and ecotourism as a cultural ecosystem service: An application at the local level in Southern Chile. Appl. Geogr. 40, 71–82. https://doi.org/10.1016/j.apgeog.2012.12.004
Naidoo, R., Kilian, J.W., Du Preez, P., Beytell, P., Aschenborn, O., Taylor, R.D., Stuart-Hill, G., 2018. Evaluating the effectiveness of local- and regional-scale wildlife corridors using quantitative metrics of functional connectivity. Biol. Conserv. 217, 96–103.

https://doi.org/10.1016/j.biocon.2017.10.037
Naiman, R.J., Decamps, H., Pollock, M., 1993. The Role of Riparian Corridors in Maintaining Regional Biodiversity. Ecol. Appl. 3, 209–212. https://doi.org/10.2307/1941822
Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, D.R., Chan, K.M. a, Daily, G.C., Goldstein, J., Kareiva, P.M., Lonsdorf, E., Naidoo, R., Ricketts, T.H., Shaw, M.R., 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Front. Ecol. Environ. 7, 4–11. https://doi.org/10.1890/080023
Newmark, W.D., Jenkins, C.N., Pimm, S.L., McNeally, P.B., Halley, J.M., 2017. Targeted habitat restoration can reduce extinction rates in fragmented forests. Proc. Natl. Acad. Sci. 114, 9635–9640. https://doi.org/10.1073/pnas.1705834114
Niklitschek, M.E., 2007. Trade Liberalization and Land Use Changes : Explaining the Expansion of Afforested Land in Chile 53, 385–394.
Núñez, D., Nahuelhual, L., Oyarzún, C., 2006. Forests and water: The value of native temperate forests in supplying water for human consumption. Ecol. Econ. 58, 606–616. https://doi.org/10.1016/j.ecolecon.2005.08.010
Nyland, R.D., 2002. Silviculture: Concepts and Applications, 2nd Editio. ed. Waveland Press, Illinois.
Olivares, B., Verbist, K., Lobo, D., Vargas, R., Silva, O., 2011. Evaluation of the Usle Model To Estimate Water Erosion in an Alfisol. J. soil Sci. plant Nutr. 11, 72–85. https://doi.org/10.4067/S0718-95162011000200007
Olson, D.M., Dinerstein, E., Wikramanayake, E.D., Burgess, N.D., Powell, G.V.N., Underwood, E.C., D'amico, J.A., Itoua, I., Strand, H.E., Morrison, J.C., Loucks, C.J., Allnutt, T.F., Ricketts, T.H., Kura, Y., Lamoreux, J.F., Wettengel, W.W., Hedao, P., Kassem, K.R., 2001. Terrestrial Ecoregions of the World: A New Map of Life on Earth. Bioscience 51, 933. https://doi.org/10.1641/0006-3568(2001)051[0933:TEOTWA]2.0.CO;2
Olsson, P., Gunderson, L.H., Carpenter, S.R., Ryan, P., Lebel, L., Folke, C., Holling, C.S., 2006. Shooting the Rapids : Navigating Transitions to Adaptive Governance of Social-Ecological Systems 11.
Otero, L.A., 2006. La huella del fuego. Historia de los bosques nativos poblamientos y cambios en el paisaje del sur de Chile. Pehuen, Santiago.
Oyarzún, C.E., Aracena, C., Rutherford, P., Godoy, R., Deschrijver, A., Oyarzun, C., Aracena, C., Rutherford, P., Godoy, R., Deschrijver, A., Oyarzún, C.E., Aracena, C., Rutherford, P., Godoy, R., Deschrijver, A., 2007. Effects of land use conversion from native forests to exotic plantations on nitrogen and phosphorus retention in catchments of southern Chile. Water. Air. Soil Pollut. 179, 341–350. https://doi.org/10.1007/s11270-006-9237-4
Oyarzun, C.E., Campos, H., Huber, A., 1997. Exportación de nutrientes en microcuencas con distinto uso del suelo en el sur de Chile (Lago Rupanco , X Región). Rev. Chil. Hist. Nat. 70, 507–519.
Oyarzún, C.E., Godoy, R., De Schrijver, A., Staelens, J., Lust, N., 2004. Water chemistry and nutrient budgets in an undisturbed evergreen rainforest of southern Chile. Biogeochemistry 71, 107–123. https://doi.org/10.1007/s10533-004-4107-x
Oyarzún, C.E., Hervé-Fernandez, P., 2015. Ecohydrology and Nutrient Fluxes in Forest Ecosystems of Southern Chile. Biodivers. Ecosyst. - Link. Struct. Funct. 581–600. https://doi.org/10.5772/59016
Oyarzun, C.E., Hervé-fernández, P., Huygens, D., Boeckx, P., 2015. Hydrological Controls on Nutrient Exportation from Old-Growth Evergreen Rainforests and Eucalyptus nitens Plantation in Headwater Catchments at Southern Chile 19–31. https://doi.org/10.4236/ojmh.2015.52003
Pal, M., 2005. Random forest classifier for remote sensing classification. Int. J. Remote Sens. 26, 217–222. https://doi.org/10.1080/01431160412331269698
Patterson, M.W., Hoalst-Pullen, N., 2011. Dynamic equifinality: The case of south-central Chile's evolving forest landscape. Appl. Geogr. 31, 641–649. https://doi.org/10.1016/j.apgeog.2010.12.004
Pauchard, A., Alaback, P.B., 2004. Influence of Elevation, Land Use, and Landscape Context on Patterns of Alien Plant Invasions along Roadsides in Protected Areas of South-Central Chile

Published by : Wiley for Society for Conservation Biology Stable URL. Conserv. Biol. 18, 238–248. https://doi.org/10.1111/j.1523-1739.2004.00300.x

Pflugmacher, D., Cohen, W.B., E. Kennedy, R., 2012a. Using Landsat-derived disturbance history (1972–2010) to predict current forest structure. Remote Sens. Environ. 122, 146–165. https://doi.org/10.1016/j.rse.2011.09.025

Pinilla S., J.., 1998. Antecedentes zonas de isocrecimiento potencial para Eucalyptus. Santiago, Chile.

Pontius, R.G., Millones, M., Pontius, Robert, Gilmore, J., Millones, M., Pontius, R.G., Millones, M., 2011. Death to Kappa: Birth of quantity disagreement and allocation disagreement for accuracy assessment. Int. J. Remote Sens. 32, 4407–4429. https://doi.org/10.1080/01431161.2011.552923

Pontius, R.G., Shusas, E., McEachern, M., 2004. Detecting important categorical land changes while accounting for persistence. Agric. Ecosyst. Environ. 101, 251–268. https://doi.org/10.1016/j.agee.2003.09.008

Pringle, R.M., 2017. Upgrading protected areas to conserve wild biodiversity. Nature 546, 91–99. https://doi.org/10.1038/nature22902

Prugh, L.R., Hodges, K.E., Sinclair, A.R.E., Brashares, J.S., 2008. Effect of habitat area and isolation on fragmented animal populations. Proc. Natl. Acad. Sci. U. S. A. 105, 20770–5. https://doi.org/10.1073/pnas.0806080105

Puyravaud, J., 2003. Standardizing the calculation of the annual rate of deforestation. For. Ecol. Manage. 177, 593–596.

Qiu, J., Turner, M.G., 2013. Spatial interactions among ecosystem services in an urbanizing agricultural watershed. Proc. Natl. Acad. Sci. U. S. A. 110, 12149–54. https://doi.org/10.1073/pnas.1310539110

R Development Core Team, 2013. R: A Language and Environment for Statistical computing.

Raudsepp-Hearne, C., Peterson, G.D., 2016. Scale and ecosystem services: How do observation, management, and analysis shift with scale—lessons from Quï¿½bec. Ecol. Soc. 21. https://doi.org/10.5751/ES-08605-210316

Raudsepp-Hearne, C., Peterson, G.D., Bennett, E.M., 2010. Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. Proc. Natl. Acad. Sci. U. S. A. 107, 5242–7. https://doi.org/10.1073/pnas.0907284107

Renard, D., Rhemtulla, J.M., Bennett, E.M., 2015. Historical dynamics in ecosystem service bundles. Proc. Natl. Acad. Sci. 112, 13411–13416. https://doi.org/10.1073/pnas.1502565112

Renard, K.G., Freimund, J.R., 1994. Using monthy precipitation data to estimate R-factor in the revised USLE. J. Hydrol.

Reyers, B., Biggs, R., Cumming, G.S., Elmqvist, T., Hejnowicz, A.P., Polasky, S., 2013. Getting the measure of ecosystem services: A social-ecological approach. Front. Ecol. Environ. 11, 268–273. https://doi.org/10.1890/120144

Rieb, J.T., Chaplin-Kramer, R., Daily, G.C., Armsworth, P.R., Böhning-Gaese, K., Bonn, A., Cumming, G.S., Eigenbrod, F., Grimm, V., Jackson, B.M., Marques, A., Pattanayak, S.K., Pereira, H.M., Peterson, G.D., Ricketts, T.H., Robinson, B.E., Schröter, M., Schulte, L.A., Seppelt, R., Turner, M.G., Bennett, E.M., 2017. When, Where, and How Nature Matters for Ecosystem Services: Challenges for the Next Generation of Ecosystem Service Models. Bioscience 67, 820–833. https://doi.org/10.1093/biosci/bix075

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin, F.S.I.I.I., Lambin, E., Lenton, T.M., Scheffer, M., Folke, C., Schellnhuber, H.J., Nykvist, B., Wit, C.A. De, Hughes, T., Leeuw, S. Van Der, Rodhe, H., Sörlin, S., Snyder, P.K., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R.W., Fabry, V.J., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P., Foley, J., 2009. Planetary Boundaries : Exploring the Safe Operating Space for Humanity.

Rodriguez-Galiano, V.F., Ghimire, B., Rogan, J., Chica-Olmo, M., Rigol-Sanchez, J.P., 2012. An assessment of the effectiveness of a random forest classifier for land-cover classification. ISPRS J. Photogramm. Remote Sens. 67, 93–104. https://doi.org/10.1016/j.isprsjprs.2011.11.002

Rodríguez, J.P., Beard, T.D., Bennett, E.M., Cumming, G.S., Cork, S.J., Agard, J., Dobson, A.P., Peterson, G.D., 2006. Trade-offs across Space, Time, and Ecosystem Services. Ecol. Soc. 11, 28.

Roy, D.P., Wulder, M.A., Loveland, T.R., C.E., W., Allen, R.G., Anderson, M.C., Helder, D., Irons, J.R., Johnson, D.M., Kennedy, R., Scambos, T.A., Schaaf, C.B., Schott, J.R., Sheng, Y., Vermote, E.F., Belward, A.S., Bindschadler, R., Cohen, W.B., Gao, F., Hipple, J.D., Hostert, P., Huntington, J., Justice, C.O., Kilic, A., Kovalskyy, V., Lee, Z.P., Lymburner, L., Masek, J.G., McCorkel, J., Shuai, Y., Trezza, R., Vogelmann, J., Wynne, R.H., Zhu, Z., 2014. Landsat-8: Science and product vision for terrestrial global change research. Remote Sens. Environ. 145, 154–172. https://doi.org/10.1016/j.rse.2014.02.001

Roy Haines-Young, Potschin, M., 2013. Common International Classification of Ecosystem Services (CICES, Version 4 . 3). Rep. to Eur. Environ. Agency 1–17. https://doi.org/10.1016/B978-0-12-419964-4.00001-9

Rozzi, R., Armesto, J.J., Figueroa, J., 1994. Biodiversidad y conservación de los bosques nativos de Chile : una aproximación jerárquica 15, 55–64.

Runkel, R.L., Crawford, C.G., Timothy A. Cohn, 2004. Load estimator (LOADEST): a FORTRAN program for estimating constituent loads in streams and rivers.

Sala, O.E., Chapin, F.S., Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A., Leemans, R., Lodge, D.M., Mooney, H. a, Oesterheld, M., Poff, N.L., Sykes, M.T., Walker, B.H., Walker, M., Wall, D.H., 2000. Global biodiversity scenarios for the year 2100. Science 287, 1770–4.

Salas, C., Donoso, P.J., Vargas, R., Arriagada, C.A., Pedraza, R., Soto, D.P., 2016. The Forest Sector in Chile: An Overview and Current Challenges. J. For. 114, 562–571. https://doi.org/10.5849/jof.14-062

Salazar, A., Baldi, G., Hirota, M., Syktus, J., McAlpine, C., 2015. Land use and land cover change impacts on the regional climate of non-Amazonian South America: A review. Glob. Planet. Change 128, 103–119. https://doi.org/10.1016/j.gloplacha.2015.02.009

Schlatter, J. V., Gerding, V., 1995. Método de clasificación de sitios para la producción forestal, ejemplo en Chile. Bosque 16, 13–20.

Schlegel, B., 2001. Estimación de la biomasa y carbono en bosques del tipo forestal siempreverde. Simp. Int. Medición y Monit. la captura carbono en ecosistemas For. 14, 1–13.

Schlegel, B.C., Donoso, P.J., 2008. Effects of forest type and stand structure on coarse woody debris in old-growth rainforests in the Valdivian Andes, south-central Chile. For. Ecol. Manage. 255, 1906–1914. https://doi.org/10.1016/j.foreco.2007.12.013

Scholes, R.J., Reyers, B., Biggs, R., Spierenburg, M.J., Duriappah, A., 2013. Multi-scale and cross-scale assessments of social-ecological systems and their ecosystem services. Curr. Opin. Environ. Sustain. 5, 16–25. https://doi.org/10.1016/j.cosust.2013.01.004

Schröter, M., Remme, R.P., Sumarga, E., Barton, D.N., Hein, L., 2015. Lessons learned for spatial modelling of ecosystem services in support of ecosystem accounting. Ecosyst. Serv. 13, 64–69. https://doi.org/10.1016/j.ecoser.2014.07.003

Schulz, J.J., Cayuela, L., Echeverria, C., Salas, J., Rey Benayas, J.M., 2010. Monitoring land cover change of the dryland forest landscape of Central Chile (1975–2008). Appl. Geogr. 30, 436–447. https://doi.org/10.1016/j.apgeog.2009.12.003

Senf, C., Leitão, P.J., Pflugmacher, D., van der Linden, S., Hostert, P., 2015. Mapping land cover in complex Mediterranean landscapes using Landsat: Improved classification accuracies from integrating multi-seasonal and synthetic imagery. Remote Sens. Environ. 156, 527–536. https://doi.org/10.1016/j.rse.2014.10.018

Seppelt, R., Dormann, C.F., Eppink, F. V., Lautenbach, S., Schmidt, S., 2011. A quantitative review of ecosystem service studies: approaches, shortcomings and the road ahead. J. Appl. Ecol. 48, 630–636. https://doi.org/10.1111/j.1365-2664.2010.01952.x

Seppelt, R., Lautenbach, S., Volk, M., 2013. Identifying trade-offs between ecosystem services, land use, and biodiversity: a plea for combining scenario analysis and optimization on different spatial scales. Curr. Opin. Environ. Sustain. 5, 458–463. https://doi.org/10.1016/j.cosust.2013.05.002

SERNATUR, GORE Region de los Rios, 2014. Plan de Accion Region de los Rios, sector turismo 2014-2018 56.

Sharp, R., Tallis, H., Ricketts, T., Guerry, A.D., Wood, S.A., Chaplin-Kramer, R., Nelson, E., Ennaanay, D., Wolny, S., Olwero, N., Vigerstol, K., Pennington, D., Mendoza, G., Aukema, J., Foster, J., Forrest, J., Cameron, D., Arkema, K., Lonsdorf, E., Kennedy, C., Verutes, G., Kim, C.K., Guannel, G., Papenfus, M., Toft, J., Marsik, M., Bernhardt, J., Griffin, R., Glowinski, K., Chaumont, N., Perelman, A., Lacayo, M., Mandle, L., Hamel, P., Vogl, A.L., Rogers, L., Bierbower, W., 2015. InVEST 3.2.0 User's Guide. The Natural Capital Project. Stanford University, University of Minnesota, The Nature Conservancy, and World Wildlife Fund.

Siebert, S.F., 2003. Beyond Malthus and perverse incentives: economic globalization forest conversion and habitat fragmentation., in: Bradshaw, G.A, Marquet, P.. (Ed.), How Landscapes Change. Human Disturbance and Ecosystem Fragmentation in the Americas. Berlin, pp. 19–32.

SINIA, M. de M.A., 2015. Sistema de Información Ambiental Geográfica.

Skovsgaard, J.P., Vanclay, J.K., 2008. Forest site productivity: A review of the evolution of dendrometric concepts for even-aged stands. Forestry 81, 13–31. https://doi.org/10.1093/forestry/cpm041

Spake, R., Lasseur, R., Crouzat, E., Bullock, J.M., Lavorel, S., Parks, K.E., Schaafsma, M., Bennett, E.M., Maes, J., Mulligan, M., Mouchet, M., Peterson, G.D., Schulp, C.J.E., Thuiller, W., Turner, M.G., Verburg, P.H., Eigenbrod, F., 2017. Unpacking ecosystem service bundles: Towards predictive mapping of synergies and trade-offs between ecosystem services. Glob. Environ. Chang. 47, 37–50. https://doi.org/10.1016/j.gloenvcha.2017.08.004

Stow, D. a, Hope, A., McGuire, D., Verbyla, D., Gamon, J., Huemmrich, F., Houston, S., Racine, C., Sturm, M., Tape, K., Hinzman, L., Yoshikawa, K., Tweedie, C., Noyle, B., Silapaswan, C., Douglas, D., Griffith, B., Jia, G., Epstein, H., Walker, D., Daeschner, S., Petersen, A., Zhou, L., Myneni, R., 2004. Remote sensing of vegetation and land-cover change in Arctic Tundra Ecosystems. Remote Sens. Environ. 89, 281–308. https://doi.org/10.1016/j.rse.2003.10.018

Syrbe, R.-U., Grunewald, K., 2017. Ecosystem service supply and demand – the challenge to balance spatial mismatches. Int. J. Biodivers. Sci. Ecosyst. Serv. Manag. 13, 148–161. https://doi.org/10.1080/21513732.2017.1407362

Syrbe, R.-U.U., Walz, U., 2012. Spatial indicators for the assessment of ecosystem services: Providing, benefiting and connecting areas and landscape metrics. Ecol. Indic. 21, 80–88. https://doi.org/10.1016/j.ecolind.2012.02.013

Taylor, P., Farig, L., Henein, K., Merriam, G., 1993. Connectivity is a vital element of landscape structure. Oikos 68, 571–573.

Tomscha, S.A., Sutherland, I.J., Renard, D., Gergel, S.E., Rhemtulla, J.M., Bennett, E.M., Daniels, L.D., Eddy, I.M., Clark, E.E., 2016. A guide to historical data sets for reconstructing ecosystem services over time. Bioscience XX, 1–16. https://doi.org/10.1093/biosci/biw086

Torres, R., Azócar, G., Rojas, J., Montecinos, A., Paredes, P., 2015. Vulnerability and resistance to neoliberal environmental changes: An assessment of agriculture and forestry in the Biobio region of Chile (1974-2014). Geoforum 60, 107–122. https://doi.org/10.1016/j.geoforum.2014.12.013

Turner, B.L., Lambin, E.F., Reenberg, A., 2007. The emergence of land change science for global environmental change and sustainability. Proc. Natl. Acad. Sci. 105, 20690–20695.

Turner, B.L.L., Lambin, E.F., Reenberg, A., B. L. Turner, E.F.L. and A.R., Turner, B.L.L., Lambin, E.F., Reenberg, A., 2007. The emergence of land change science for global environmental change and sustainability. Proc. Natl. Acad. Sci. 105, 20690–20695.

Turner, M.G., 1989a. Landscape Ecology : The Effect Pattern on Process. Annu. Rev. Ecol. Syst. 20, 171–197.

Turner, M.G., Donato, D.C., Romme, W.H., 2012. Consequences of spatial heterogeneity for ecosystem services in changing forest landscapes: priorities for future research. Landsc. Ecol. 28, 1081–1097. https://doi.org/10.1007/s10980-012-9741-4

Turner, R.K., Daily, G.C., 2008. The ecosystem services framework and natural capital conservation. Environ. Resour. Econ. 39, 25–35. https://doi.org/10.1007/s10640-007-9176-6

UNEP, 2010. Environment Outlook: Latin America and the Caribian -GEO LAC 3, United Nation

Environmental Programme.

Veblen, T.T., Schlegel, F.M., Oltremari, J. V, 1983. Temperate broad-leaved evergreen forests of South America, in: JD, O. (Ed.), Temperate Broad-Leaved Evergreen Forests. Elsevier, Amsterdam, pp. 5–31.

Verburg, P.H., van de Steeg, J., Veldkamp, a Willemen, L., 2009. From land cover change to land function dynamics: a major challenge to improve land characterization. J. Environ. Manage. 90, 1327–35. https://doi.org/10.1016/j.jenvman.2008.08.005

Vitousek, P.M., Mooney, H.A., Lubchenco, J., Melillo, J.M., 1997. Human Domination of Earth ' s Ecosystems. Science (80-.). 277, 494–499. https://doi.org/10.1126/science.277.5325.494

Vogelmann, J.E., Xian, G., Homer, C., Tolk, B., 2012. Monitoring gradual ecosystem change using Landsat time series analyses: Case studies in selected forest and rangeland ecosystems. Remote Sens. Environ. 122, 92–105. https://doi.org/10.1016/j.rse.2011.06.027

Volk, M., 2013. Modelling ecosystem services – Challenges and promising future directions. Sustain. Water Qual. Ecol. 1–2, 3–9. https://doi.org/10.1016/j.swaqe.2014.05.003

Walker, B.H., Anderies, J.M., Kinzig, A.P., Ryan, P., 2006. Exploring Resilience in Social-Ecological Systems Through Comparative Studies and Theory Development : Introduction to the Special Issue 11.

Wallace, J.., Acreman, M.C., Sullivan, C.., 2003. The sharing of water between society and ecosystems: from conflict to catchment–based co–management. Philos. Trans. R. Soc. Lond. B. Biol. Sci. https://doi.org/10.1098/rstb.2003.1383

Walz, U., 2015. Indicators to monitor the structural diversity of landscapes. Ecol. Modell. 295, 88–106. https://doi.org/10.1016/j.ecolmodel.2014.07.011

Walz, U., Stein, C., 2014. Indicators of Hemeroby for Land Use Monitoring in Germany. J. Nat. Conserv. 22, 279–289.

Waske, B., Braun, M., 2009. Classifier ensembles for land cover mapping using multitemporal SAR imagery. ISPRS J. Photogramm. Remote Sens. 64, 450–457. https://doi.org/10.1016/j.isprsjprs.2009.01.003

Waske, B., van der Linden, S., Oldenburg, C., Jakimow, B., Rabe, A., Hostert, P., 2012. imageRF – A user-oriented implementation for remote sensing image analysis with Random Forests. Environ. Model. Softw. 35, 192–193. https://doi.org/10.1016/j.envsoft.2012.01.014

Watson, J.E.M., Dudley, N., Segan, D.B., Hockings, M., 2014. The performance and potential of protected areas. Nature 515, 67–73. https://doi.org/10.1038/nature13947

Wei, H., Fan, W., Wang, X., Lu, N., Dong, X., Zhao, Y., Ya, X., Zhao, Y., 2017. Integrating supply and social demand in ecosystem services assessment: A review. Ecosyst. Serv. 25, 15–27. https://doi.org/10.1016/j.ecoser.2017.03.017

Wischmeier, W.H., Smith, D.D., 1978. Predicting rainfall erosion losses. Agric. Handb. no. 537 285–291. https://doi.org/10.1029/TR039i002p00285

Wong, C.P., Jiang, B., Kinzig, A.P., Lee, K.N., Ouyang, Z., 2015. Linking ecosystem characteristics to final ecosystem services for public policy. Ecol. Lett. 18, 108–118. https://doi.org/10.1111/ele.12389

Wulder, M. a., White, J.C., Goward, S.N., Masek, J.G., Irons, J.R., Herold, M., Cohen, W.B., Loveland, T.R., Woodcock, C.E., 2008. Landsat continuity: Issues and opportunities for land cover monitoring. Remote Sens. Environ. 112, 955–969. https://doi.org/10.1016/j.rse.2007.07.004

Wulder, M.A., Masek, J.G., Cohen, W.B., Loveland, T.R., Woodcock, C.E., 2012. Opening the archive: How free data has enabled the science and monitoring promise of Landsat. Remote Sens. Environ. 122, 2–10. https://doi.org/10.1016/j.rse.2012.01.010

Xin, Q., Olofsson, P., Zhu, Z., Tan, B., Woodcock, C.E., 2013. Toward near real-time monitoring of forest disturbance by fusion of MODIS and Landsat data. Remote Sens. Environ. 135, 234–247. https://doi.org/10.1016/j.rse.2013.04.002

Yates, E.D., Levia, D.F., Williams, C.L., 2004. Recruitment of three non-native invasive plants into a fragmented forest in southern Illinois. For. Ecol. Manage. 190, 119–130. https://doi.org/10.1016/j.foreco.2003.11.008

Young, J., Watt, A., Nowicki, P., Alard, D., Clitherow, J., Klaus, H., Johnson, R., Laczko, E., McCracken, D., Matouch, S., Niemela, J., Richadars, C., 2005. Towards sustainable land use:

identifying and managing the conflicts between human activities and biodiversity conservation in Europe. Biodivers. Conserv. 14, 1641–1661. https://doi.org/https://doi.org/10.1007/s10531-004-0536-z

Zamorano-Elgueta, C., Rey Benayas, J.M., Cayuela, L., Hantson, S., Armenteras, D., 2015. Native forest replacement by exotic plantations in southern Chile (1985-2011) and partial compensation by natural regeneration. For. Ecol. Manage. 345, 10–20. https://doi.org/10.1016/j.foreco.2015.02.025

Zhang, L., Podlasly, C., Feger, K.H., Wang, Y., Schwärzel, K., 2015. Different land management measures and climate change impacts on the runoff - A simple empirical method derived in a mesoscale catchment on the Loess Plateau. J. Arid Environ. 120, 42–50. https://doi.org/10.1016/j.jaridenv.2015.04.005

Zhu, Z., Woodcock, C.E., 2014. Continuous change detection and classification of land cover using all available Landsat data. Remote Sens. Environ. 144, 152–171. https://doi.org/10.1016/j.rse.2014.01.011

Zhu, Z., Woodcock, C.E., 2012. Object-based cloud and cloud shadow detection in Landsat imagery. Remote Sens. Environ. 118, 83–94. https://doi.org/10.1016/j.rse.2011.10.028

Zhu, Z., Woodcock, C.E., Olofsson, P., 2012. Continuous monitoring of forest disturbance using all available Landsat imagery. Remote Sens. Environ. 122, 75–91. https://doi.org/10.1016/j.rse.2011.10.030

Zomer, R.J., Trabucco, A., Bossio, D.A., Verchot, L. V., 2008. Climate change mitigation: A spatial analysis of global land suitability for clean development mechanism afforestation and reforestation. Agric. Ecosyst. Environ. 126, 67–80. https://doi.org/10.1016/j.agee.2008.01.014

Zomer, R.J., Trabucco, A., Straaten, O. Van, Bossio, D.A., 2007. Carbon, Land and Water: A Global Analysis of the Hidrologic Dimensions of Climate Change Mitigation through Afforestation/Reforestation.

Declaration of the candidate

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